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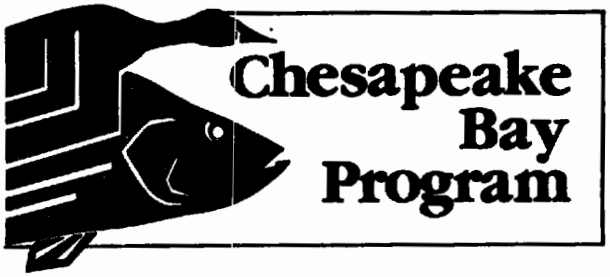
Understanding the Estuary: Advances in Chesapeake Bay Research

Proceedings of a Conference March 29–31, 1988



Understanding the Estuary: Advances in Chesapeake Bay Research

Proceedings of a Conference
29-31 March 1988
Baltimore, Maryland

Maurice P. Lynch and Elizabeth C. Krome
Editors

August 1988

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FOREWORD

A conference entitled "Understanding the Estuary: Advances in Chesapeake Bay Research" was held 29-31 March 1988 in Baltimore, MD. The conference was primarily oriented towards scientists engaged in research on fundamental estuarine processes in Chesapeake Bay and secondarily oriented to managers with scientific and technical backgrounds. Scientifically and technically conversant citizens were also encouraged to attend.

The conference examined recent research findings in several areas that are relevant to a wide range of estuarine processes. Conveners attempted to provide a context for assessing the relevance of these scientific findings to the long-term efforts to protect and restore the Chesapeake watershed.

Estuarine research is coming of age in terms of scientific credibility. For years, many scientists engaged in oceanographic or limnological studies considered estuarine research as a spinoff or splinter effort of blue-water or freshwater work. Now, the realization is growing that estuaries are unique areas, worthy of focused attention. This recognition comes none too soon. Coastal demographic pressures and readily visible degradation of habitats, living resources, and water quality, have created a strong public outcry to "do something" about coastal and estuarine pollution.

The Chesapeake Bay area is fortunate in that estuarine research has been nurtured and encouraged by the regional states, which established laboratories dedicated to estuarine studies of the Bay as far back as the 1920's. In the past two decades, interaction of Bay scientists, resource managers, and the public has both nurtured and been nurtured by major federally funded Chesapeake Bay studies. The most recent of these studies was funded through the U.S. Environmental Protection Agency and resulted in a coordinated federal, Maryland, Virginia, Pennsylvania, and District of Columbia Chesapeake Bay Restoration and Protection Program. This program is considered a model for estuarine management both nationally and internationally.

The emergence of the Chesapeake Bay Program as a national and international model of estuarine management is based to no small extent on accumulated information developed by the Bay research community. Maintaining Chesapeake Bay Program's momentum will depend on two things. The Bay scientific and technical community must improve understanding of the vital estuarine processes. Equally important, this understanding must be communicated to the managers and the public. It is hoped that this conference contributed to this improvement in understanding and communication.

We wish to thank the individuals who contributed to these Proceedings and the conference by attending the sessions, presenting the papers, and participating in the discussions. We hope to continue these exchanges through similar conferences and activities in the future.

Maurice P. Lynch
Joseph A. Mihursky
Conference Co-chairs

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The Johns Hopkins University
Baltimore, Maryland 21211

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Benedict Estuarine Research Laboratory
Benedict, Maryland 20612

Genetics and Species Conservation

Robert W. Chapman
The Johns Hopkins University
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Jonathan Garber
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Joseph A. Mihursky
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Uncertainty in Estuarine Research

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My question today is why there is any uncertainty left in estuarine research. After all, AERS began looking at estuarine research in 1947. A monumental research symposium was held in 1964, and the Estuarine Research Federation was formed shortly thereafter. Within this time we have also seen the establishment of the U. S. Office of Naval Research, the National Science Foundation, the Environmental Protection Agency, the National Oceanographic and Atmospheric Administration, Sea Grant, and the Department of Energy. With all this activity, why are there any uncertainties? We deal in facts, unlike the soft sciences. For example, in the physical sciences, everything has been reduced to equations. Tide tables are available for dates far in the future—there is no uncertainty about them. The National Estuarine Inventory Data Analysis contains detailed information on the James and other rivers. The chemical facts of water are undisputed—water contains two atoms of hydrogen and one of oxygen. The biological sciences do still have a few areas that are not as determined as the chemical and physical sciences. Nevertheless, biologists have made major strides in recent years — for example, in DNA, RNA, and genetic manipulation, and the buzzword now in Washington is marine biotechnology.

Questions do remain. In North Carolina, for example, there is the question of the striped bass. There are not many left, and people ask why. There is that gift from Florida, the red tide. Blame for these problems has been placed, in some quarters at least, on the Russians flushing their bilges or on a vicious plot by developers.

Outside the research laboratories, in the real world, are we dealing with certainty or uncertainty? Some things are certain. For example, we can look at an aerial view of an entrepreneur's effort to make waterfront property available to more of mankind—a canal system. It is certain that this enterprise will lead to more people, which will lead to more sewerage. Inevitably this enterprise leads to sewage pollution. Must we deliberate about whether an algal bloom will follow, and then a fish kill? Lawyers and administrators and multitudes of scientists are not necessary to verify that this sequence of events is a certainty.

Another example is the building of ports. Where there are ports, there are ships, and where there are ships, there is oil, which sooner or later spills. The consequence is the death of wildlife and the deterioration of marshes.

In the watershed we can look at the "improvement" of streams, which amounts to straightening, with the cutting down of streamside vegetation. The runoff carrries the consequence downstream, where the result will be the posting of a sign announcing "Closed to shellfishing."

So if all these things are certain, what are the uncertainties of estuarine research?

What do we really know? What do we need to know as we approach the end of the 20th century? The situation is really one of ignorance in a sea of knowledge.

An example of determining what we really know is a report drafted by Kirby Smith and me reviewing 4,693 reports on the Newport River system in North Carolina. These reports represent over 190 collective years of research at a number of labs, and what they tell us is summarized in 80 pages of lay language. This document should furnish some guidance to decision-makers about what we know and where we should go from here.

There has been plenty of deliberation in the past about where we should be going. Past publications have outlined what needs to be done next. One thing we must do next is adopt an interdisciplinary approach. Interdisciplinary research has arrived for the blue-water oceanographers, and perhaps one day this will happen to the shallow-water researchers as well. Are we as scientists practicing what we are preaching? A useful document along this line is "A Good Bay Agreement and Ways to Make it Better." Another is a draft report for the first year of the Albemarle Pamlico project, entitled "Citizens Guide to Coastal Resource Management."

Another area of uncertainty is this: To what degree

are we successful and dedicated in getting citizens involved in the effort? If we don't know what needs to be done, it will be hard to persuade citizens. We should include legislators and local governments in our thinking. Albemarle-Pamlico, for instance, is surrounded by 20 coastal counties, each of which has a board of supervisors, usually five men. These are lay people who earn their living at something besides marine science, and they may have little understanding of what these scientific documents mean. (The EPA Journal may be a little more understandable.)

Implementation of sound management practices is the largest uncertainty. We can see this problem in a hypothetical situation, perhaps described as a faraway place, where, unlike our respective home states, a myriad of agencies are presumed to be responsible for a myriad of carefully delineated problems. The scenario might concern stormwater runoff. The rain goes downhill, into an estuary. Imagine a large estuarine system. Over the course of 400 years it has evolved a tradition of fishing, first commercial and then recreational. The state has established a mechanism for identifying water quality status: a rating of SA indicates fewer than 14 fecal coliforms per 100 cc of water; SB and SC follow as higher fecal coliform counts indicate water of progressively poorer quality. Shellfish are taken only from SA waters. There is an absolute: if the level goes over 14/100 cc, one agency calls another, and the water is closed to shellfishing until the count goes down again. Over 320,000 acres have been closed over the last 40 years. Motels, shopping malls, and condominiums have arrived, bringing with them dollars, profits, and accelerated runoff including fecal coliforms. Will laws and regulatory agencies take care of this degradation? The Coastal Improvement Agency issues permits only for development above high water. The Environmental Quality Agency is responsible for water quality at the water molecule level but has no responsibility for the watershed or its resources. Another group (the Resource Agency) is responsible for living resources. Citizens trying to unravel a problem get very frustrated trying to find someone who has authority to deal with it. These different agencies rarely meet, and they report to different cabinet officers. The citizens must wonder whether the bureaucrats really care about the environment. Is there a chance, they wonder, that partisan politics and economic gains may be more important?

A final uncertainty is long-term, sustained funding for specific objectives. A new bill, the Chaffee-Mitchell bill on Marine Research Funding, would provide $30 million for research in 10 geographic districts, in environmental issues in coastal waters. We need to look at what this bill would achieve, and whether this bill is the best way to do it.

We need a better understanding of what we know and what we need to know. We need to bring citizens, legislators, and local government into closer cooperation. We need to be sure that good sound management practices evolve from this information. And we need to be sure adequate funds are available for long-term research; we do not need pork-barrel efforts that will come back to haunt us.

Changes in Understanding of the Circulation of the Chesapeake Bay

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We have been aware of the primary circulation of the Chesapeake Bay for about 40 years. The fresh water is introduced by the rivers and is amplified, so that the circulation in the Bay is considerably larger than the flux of the rivers coming into it. The paradigm is tidal action over a rough bottom, mixing salt and fresh water and driving the gravitational circulation. During the last 40 years we have learned about meteorologically forced circulation and topographically controlled circulation. We are challenged by questions such as where nutrients, dissolved oxygen, suspended sediments, and toxics go. There is much we do not understand about transport and mixing. We thought the primary mechanism was primary circulation, but now we realize that something else is also at work, and at a relatively small scale. Our current knowledge is inadequate to describe transport processes, as it principally pertains to large-scale processes.

A number of discomfitting possibilities must be considered:

- Mixing may be non-uniform in space and time.
- Transport may be dominated by transient, episodic events.
- The vertical exchange may be localized.
- Topographic controls may dominate.
- The boundary conditions for mathematical modeling may defy straightforward formulation.

The variability we observe includes tidal variability (time span of a tidal period); quarter-wave seiche (1.7 day period); local forcing (4-10 day period); non-local forcing (4-10 day period), and gravitational circulation. The meteorological response can be determined if the effects of local forcing are removed.

Two new tools are available for research. The first is the bottom-mounted acoustic Doppler profiler, which can measure the horizontal flow of water very precisely. For instance, in the Patuxent River, intrusions of high-salt water across the entrance sill are being studied with this instrumentation. Results are showing that the middle portions of the water column are more quiescent, and that the role of wind coupled with the quarter-wave seiche can be quite important, as important as the diurnal tide. Wind can excite a near-resonance process, with large-amplitude seiches and strong surges and anti-surges.

The other new tool now available both to reveal the physics and to integrate the catalog of processes is numerical modeling. Process modeling is less expensive than simulation modeling, but it is more difficult to compare directly to observations. Both kinds of modeling are needed.

Recommendations for research directions include:

- Long-term (3-5 years) measurements
- More three-dimensional studies of circulation processes
- Boundary and lateral meeting processes
- Wind stress field (data are woefully inadequate)
- Bottom stress field
- Process modeling
- Simulation modeling
- Model-observation comparisons
- Exchange between the estuary and the shelf

QUESTIONS

Q: Are we missing a component in not studying long-term flushing? What is its importance compared to that of the quarter-wave seiche?
A: We have not done quantitative checking on this, but the outflow of a quarter-wave seiche is lost to the system, and the inflow is totally new water. So seiching provides an indication of integrated inflow from the continental shelf.
Q: Internal waves were not mentioned. What is the period of the lateral seiche?
A: The process catalog includes longitudinal, lateral, and mixing processes. The lateral internal seiche time is 8-20 hours. A higher-frequency lateral seiche may be important as well. The surface gravity wave seiche is on the order of minutes.

Opportunities to Improve Understanding of the Circulation of the Chesapeake Bay

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We have a need for future research in estuaries, and we need to consider how to use modern technology. Research is being driven by a strong need for sensible management decisions. The Bay must be dealt with as the large ecosystem that it is. It interacts with itself, its watershed, and its population. Physical circulation is of primary importance, and we need to understand its role.

We understand the circulation qualitatively for the mean flow and for some of the low-frequency fluctuations. Our understanding of short-duration and small-scale variations is still rudimentary.

The processes responsible for the horizontal and vertical transport are understood: advection by the mean flow, and advection by the fluctuating flow terms such as turbulent diffusion.

Is sufficient fundamental research being done to support the decision-making process for Chesapeake Bay restoration and preservation? No, because most Bay questions, while appearing simple on the surface, require a detailed understanding for a satisfactory answer. An example is the "simple" question of how much nutrient loadings should be reduced. This simple question generates others: How much and how fast will it improve the quality of water in the Bay? Does it matter where the reductions are made? These are very difficult questions to answer. Thus it is obvious that we need to do fundamental research on the Bay's physical processes so that we know what makes the system run.

New opportunities for information are available:

- Remote and large-scale sampling of surface currents is possible.
- High-resolution Doppler profiles of the current field and the velocity field can be procured, either from moored instruments or as transects from moving vessels.
- Data from airborne and satellite remote sensing are available (although it must be remembered that use of satellite data is difficult and time-consuming, and resolution is low).
- Measurements can be made of small-scale turbulent mixing processes.

For example, in the Delaware Bay NOAA used data from a radar backscatter system to show a two-dimensional circulation. This kind of monitoring (CODAR) can and should be done on the Chesapeake, as it provides answers to long-term questions.

As another example, Doppler profiles show that circulation in the Chesapeake has three layers: downstream at the top and bottom, and upstream in the middle.

The Bay Program needs a substantial commitment to sampling as well as modeling the circulation. If sampling is inadequate we do not know whether all the processes that are taking place are being reflected in the model.

QUESTIONS

Q: Is the three-layer circulation ephemeral? What are the implications for sediment transport?
A: We don't know yet; longer-term studies, over several days or many tidal cycles, are needed.
Q: What were the velocities on the cross-Bay transect?
A: In November, 10 m/sec.
Q: What about the labor force necessary to perform the research you outline?
A: The number of people required is not that large for CODAR and the bottom-mounted acoustic doppler profiler. The money goes into equipment rather than staff.
Q: How about staff for interpreting the results?
A: This does require staffing. Although support has been available for interpretation in open-ocean research, funding for this activity in estuaries is falling through a crack between the National Science Foundation and the Environmental Protection Agency. It looks "too applied" to NSF and "not applied enough" to EPA. I think we need a serious, peer-reviewed scientific program running on the order of $1 million a year, and we need a long-term commitment.

Stratification Control and Bay-Shelf Exchange: Two Physical Processes and Their Implications for Ecosystem Dynamics

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Much of the interest in estuarine research revolves around ecosystem effects, and it is important to link the results of physical process research to the ecosystem. It is axiomatic that the major advances in estuarine research are interdisciplinary, and it is these interdisciplinary results that are of most interest to management. Two recent examples are the studies of Hurricane Agnes [Chesapeake Research Consortium, 1976] and the investigation of anoxia in the Bay in the early 1980's [Officer et al. 1984]. I will present here two examples of physical processes that have implications for the living resources, with some suggestions for future research directions.

SEASONAL STRATIFICATION

The vertical density distribution has a strong influence on the distribution of nutrients in the water column, and the degree of vertical mixing has been shown to have a strong influence on the seasonal succession of phytoplankton. The natural variability in stratification also affects the seasonal development of the anoxic water mass in the Bay. Seliger and Boggs [1987] have suggested that streamflow-induced change in stratification is the major source of interannual variability in summer volume of anoxic water. This possibility has major implications for nutrient control strategies.

Processes controlling stratification

Four major processes influence stratification in the Chesapeake: runoff, tidal mixing, wind mixing, and the dynamic balance of heating and cooling effects. In an annual cycle (Figure 1), salinity generally drops in early spring, responding to the spring freshet, and reaches a minimum in June. Stable stratification sets up in May and persists through the summer. The first major wind event in the fall brings together salinities in the top and bottom water, and they are mixed intermittently thereafter [Goodrich et al. 1987]. A large wind event can leave the water column completely mixed (Figure 2). The salinity of the lower Bay shows much more variability than does the upper Bay, as the exchange with the ocean is a major factor in the salt balance. The tide is a background source of mixing energy, which is constant in time but spatially variable; the amplitude in the upper Bay is half that in the lower Bay. The condition of stratification is thus a dynamic balance between the buoyancy inputs (runoff and surface heating) and mixing energy inputs (tide, wind, and surface cooling).

Prospects for prediction and hindcasting

The processes controlling stratification are well understood on a large scale, but more work is needed to better define the small-scale processes. A large historical data base exists on the internal behavior of the estuary and on its boundary forcing (including current, salinity, and temperature for the former, and runoff, sea level, and wind for the latter). Large-scale hydrodynamic modeling is in a relatively advanced state. For example, the model developed by Blumberg and Goodrich has simulated the overturn that was observed in September 1983. Prospects are good for determining the natural variability in stratification using a combination of hydrodynamic modeling and historical observations. This natural variability could then be removed as a factor in the interannual variations in the anoxic water mass. Unlike the two previous speakers, I am not convinced that a large-scale field program is necessary to solve this particular problem.

BAY-SHELF EXCHANGE AND BLUE CRAB LARVAL RETENTION

Another example of the impact of physical processes on the ecosystem is found in the exchange between the Bay and the shelf, and its effect on the larval population of blue crabs.

Crab populations are seemingly unaffected by pollution or fishing pressure, but the natural variability

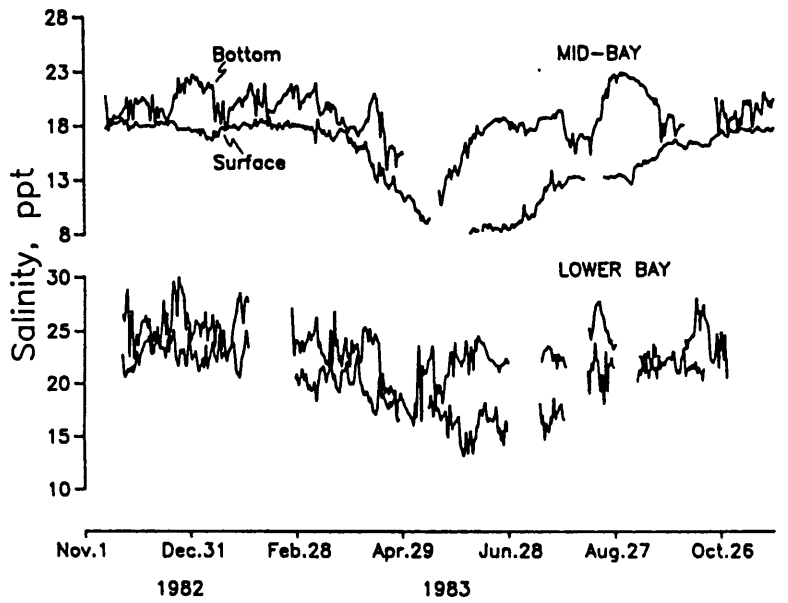

Figure 1. Continuous surface and bottom salinity from two current meter moorings in the mid-Bay region (off Patuxent River entrance) and in the lower Bay (between the Rappahannock and York rivers). Data have been lowpass-filtered to remove variance at tidal frequencies and above. Data from NOAA Chesapeake Bay Circulation Survey.

in stocks is so great that any anthropogenic signal would be difficult to discern. Much of this variability stems from the blue crab's early life history. The larvae develop offshore, and the return of the postlarvae (megalopae) to the Bay depends on physical transport. Current data (Figure 3) indicate that the Eulerian mean flow field at the Bay entrance was relatively stable in the two years of observation. Superimposed on this flow is a strongly variable wind-driven exchange, the magnitude of which can be accurately estimated from sea level records at virtually any time (Figure 4).

Three-year daily records of megalopae collected at the VIMS pier in the York River show an episodic distribution. Thirteen of the 18 peaks observed over the three years correspond to positive anomalies in subtidal volume, including the largest 1985 peak, which occurred during the massive storm surge associated with Hurricane Juan (Figure 5). This correspondence suggests that an important mechanism for the return of megalopae into the Bay is transport via wind-driven exchange events. Analysis of the frequency of these exchange events over a 28-year period indicates that they reach a fall peak in the last two weeks of September, during which time an average of roughly two

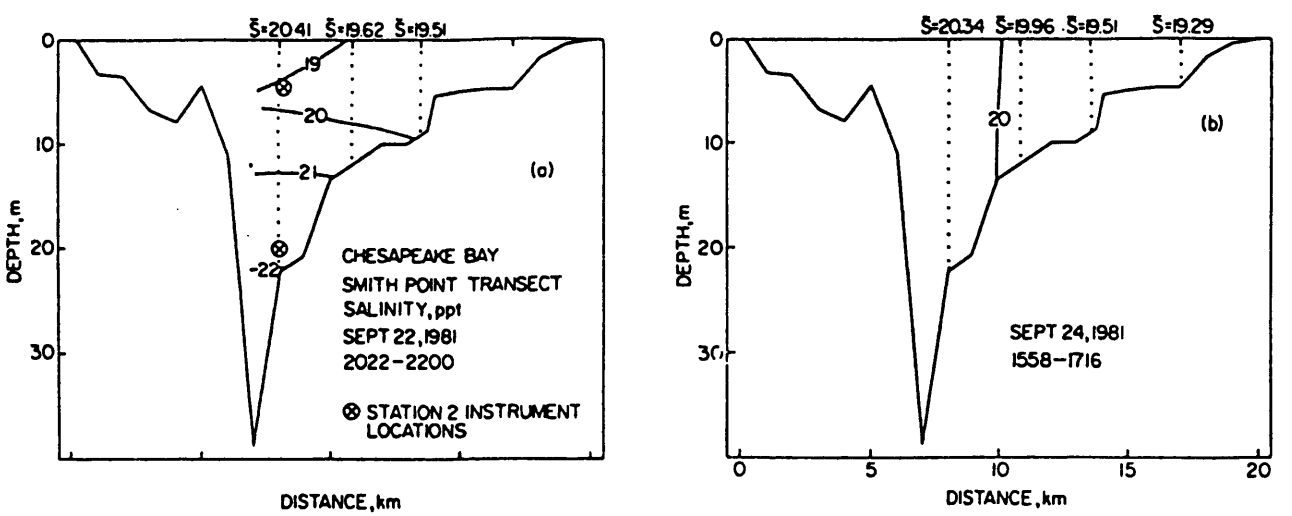

Figure 2. Lateral salinity sections at Smith Point (south of Potomac River entrance) before (a) and after (b) the storm of September 23-24, 1981.

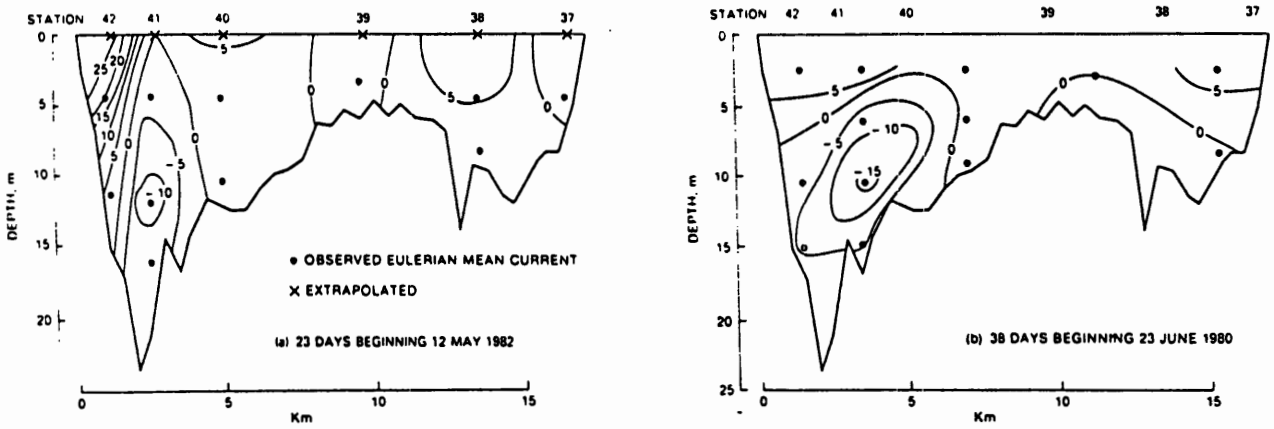

Figure 3. Lateral current structure at the Chesapeake Bay entrance. Velocities are in cm/sec and are normal to the transect. Positive velocities are out of the Bay. The 1982 data are from NOAA, the 1980 data from the Chesapeake Bay Institute.

inflow events greater than a tidal prism can be expected (Figure 6). The 28-year record indicates that an average of ten events of at least this magnitude can be expected during the July–November period when megalopae are present. The number of inflow events increases in the late summer, reaching a maximum around the equinox before leveling off. This is significant because water temperatures decline rapidly after the equinox until the megalopae cannot sustain activity. The main point here is that these transport events are not fortuitous but rather must be considered a stable feature of the flow climatology of the Bay entrance. It seems plausible that the blue crab has evolved a strategy to take advantage of this feature.

The larvae of a number of other commercially important species, such as menhaden and croaker, must also pass through the mouth of the Bay in relatively non-motile forms. If recruitment studies for these organisms are to resolve the dominant transport processes, they should include daily sampling. Weekly or biweekly sampling cruises are likely to miss the wind-associated recruitment events that may dominate the recruitment process.

A climatological approach to physical studies of the Bay can be very productive. As the above example suggests, small changes in the long-term physical behavior of the system can have significant effects on particular species. A climatological analysis of the many available long-term records is needed both for ecological applications and as a benchmark for numerical model simulations.

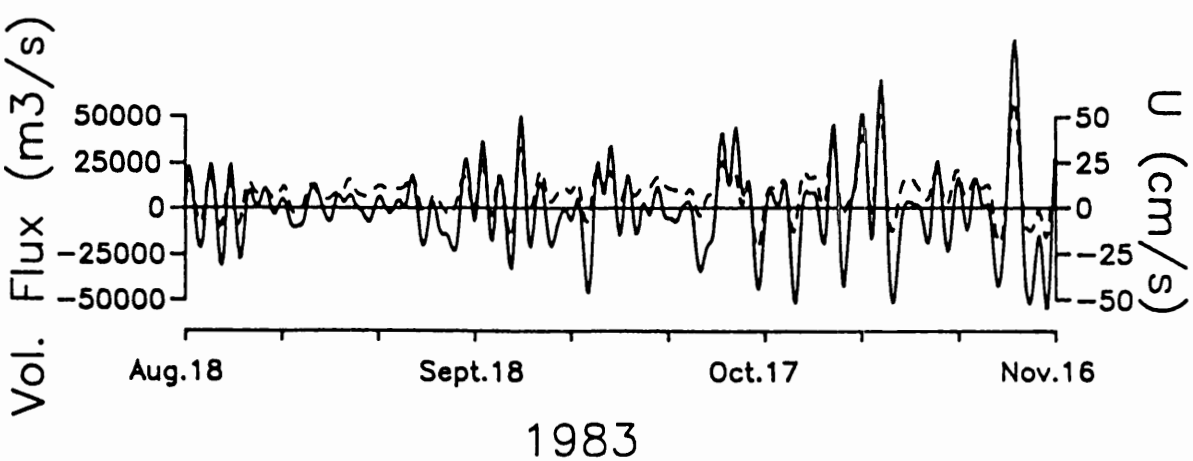

Figure 4. Volume flux and velocity at the Chesapeake Bay entrance. Solid line is the subtidal volume flux as calculated from sea level records at Baltimore, Solomons, and Kiptopeke. Dashed line is the filtered principal axis velocity at the Chesapeake Bay entrance (station 40, surface; see Figure 3 for location).

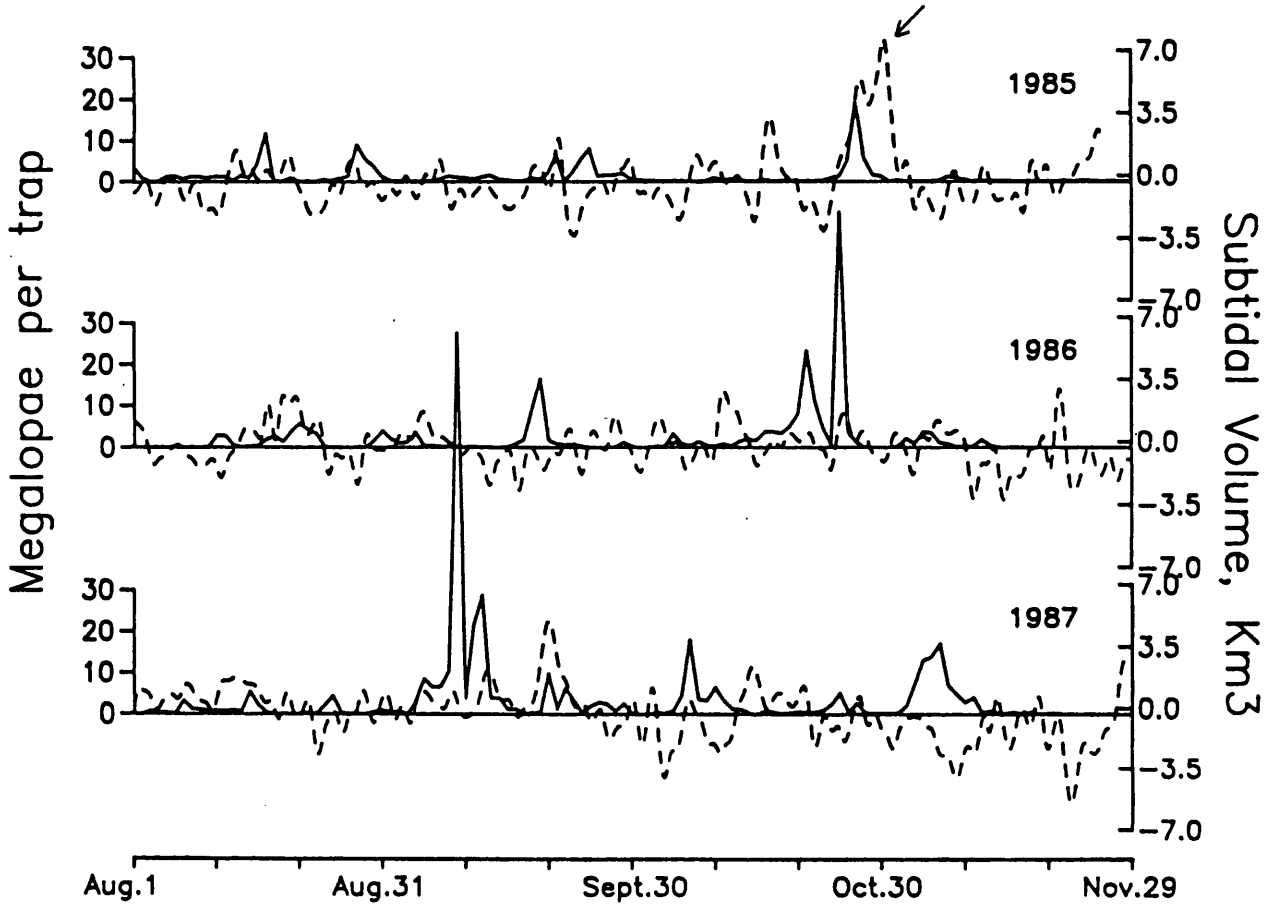

Figure 5. Mean number of megalopae per trap at VIMS pier (solid) and Chesapeake Bay subtidal volume (dashed), 1985-87. Zero on subtidal volume axis represents series mean. Arrow indicates passage of Hurricane Juan in November 1985.

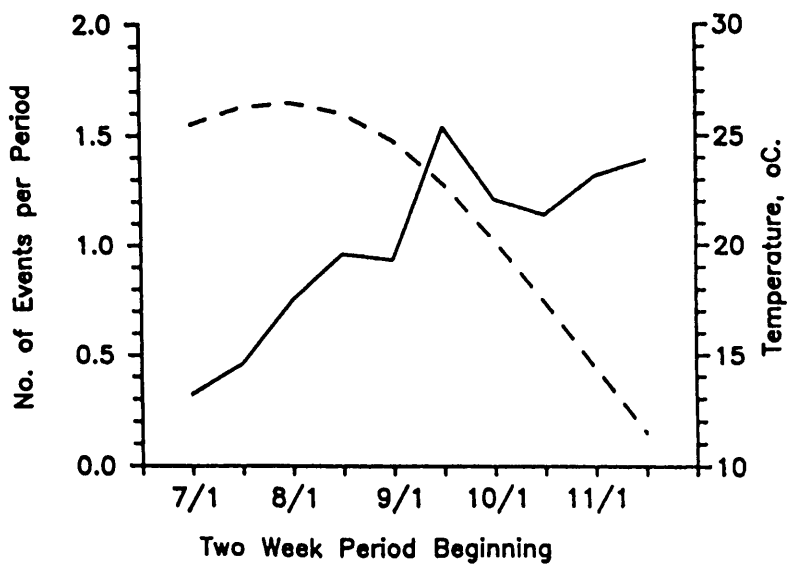

Figure 6. Frequency of inflow events greater than one tidal prism (solid), 1955-86, and monthly mean water temperature at the Chesapeake Bay Bridge Tunnel (dashed) [Dowgiallo et al. 1984].

QUESTIONS

Q: It should be noted that there is a contradiction between your assessment of the suitability of the longer-term data bases and the assessment that Boicourt has made.

REFERENCES

Blumberg, A., D. Goodrich. Modeling of wind-induced destratification in Chesapeake Bay. Submitted to J. Geophys. Res.

Chesapeake Research Consortium, 1976. The effects of tropical storm Agnes on the Chesapeake Bay estuarine system. Johns Hopkins University Press, Baltimore, MD. 639 pp.

Dowgiallo, M., M. Predoehl, S. Green, R. Dennis, 1984. Marine environmental assessment, Chesapeake Bay, 1983 annual summary. NOAA National Environmental Satellite, Data and Information Service, Washington, D. C. 81 pp.

Goodrich, D., W. Boicourt, P. Hamilton, D. Pritchard, 1987. Wind-induced destratification in Chesapeake Bay. J. Phys. Oceanogr. 17:2232-2240.

Officer, C., R. Biggs, J. Taft, L. Cronin, M. Tyler, W. Boynton, 1984. Chesapeake Bay anoxia: origin, development and significance. Science 223:22-27.

Seliger, H., J. Boggs, 1987. Anoxia in the Chesapeake Bay (abstract). EOS Trans. Am. Geophys. Union 68(30), 1692 pp.

Chemical and Physical Processes Influencing Bioavailability of Toxics in Estuaries: An Overview

Within the Chesapeake Bay region, pressure from increased population continues to mount. Population increases bring increases in water use, shipping, recreation, the location of industries on the water—and it is inevitable that the Chesapeake Bay will be subjected to continued loading of toxic substances.

We as a regional society must address the potential impact of contaminants in the Chesapeake Bay. To predict and assess impact, however, we must first understand the processes that control contaminant toxicity and availability. This session is an attempt to present the current view of our knowledge in this area —how contaminants move through the estuary and the coastal zone, and what abiotic and biotic processes control contaminant movement, fate, and bioavailability.

This session will be composed of two speakers; one will generally address inorganic contaminants; the other will address organic substances. The focus will be not only on what we know, but also on what we must still discover. In that regard, I offer the following recommendations for further research. I ask that you keep them in mind as you listen to the following speakers.

- Determine how partitioning affects the toxicity of model compounds. Is complexation a detoxification mechanism? Are particulate-associated contaminants still toxic? How important is concentration in the surface microlayer?
- Understand the role of biota in the transfer of pollutants. Is transport across the sediment/water interface an important process? What about fecal pellet production? What is the fate of contaminants associated with biological tissues?
- Examine the importance of anoxia on transport and availability. Is availability increased or decreased? Is the reservoir capacity of the sediments increased or decreased?
- Evaluate the importance of sediments as a controlling mechanism. Are they a sink or source, and for which contaminants? What controls the flux of carbon and contaminants across the sediment/water interface?
- Understand the role of communities in the transport and transformation of contaminants. Can we move away from laboratory experiments with single species? Can we move toward natural community manipulations and microcosm approaches?

—*J.G.S.*

Factors Affecting the Bioavailability of Toxic Trace Elements to Estuarine Organisms

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INTRODUCTION

The many factors altering the bioavailability of toxic trace elements may be considered in two major groupings. One group consists of processes that somehow change the availability of the contaminant present, but do not alter its total amount or basic chemical form. This group of factors includes organic and inorganic complexation, ionic strength, pH, redox reactions, and competition with similar compounds. These factors are rather general in nature, applying to large groups of organisms and large groups of elements in a wide variety of environments.

The other major group of factors controlling bioavailability comprises processes that control the concentration, distribution, or chemical form of the contaminant within the system. Factors in this group include sources and sinks, such as adsorption, flocculation, sedimentation, and remobilization, redox reactions, and the formation of organometallic compounds. While these factors do operate in most environments, their relative importance in each environment with respect to a particular element is highly variable, and to adequately quantify their effects requires study of a specific site and element.

It is also important to note that organisms are not merely affected by the trace elements in question; their activities in the environment also help determine in large measure the factors that control bioavailability of trace elements.

GENERAL ROUTES OF TRACE ELEMENT UPTAKE

In discussing the bioavailability of trace elements to organisms, it is useful to consider groups of compounds that have similar patterns of availability. A focus of recent research has been the effect of chemical form, or speciation, of trace elements on their availability to aquatic biota. There are three groups of contaminants for which generalizations concerning bioavailability are available: cationic trace elements, anionic trace elements, and organic compounds (Figure 1).

For cationic elements, free ion activity appears to be the major factor determining bioavailability [Sunda and Guillard, 1976; Anderson and Morel 1978; Sunda et al. 1978; Engel and Sunda 1979; Rueter and Morel 1981; Anderson and Morel 1982; Harrison and Morel 1983; Morel and Morel-Laurens 1983; Zamuda 1984; Wright and Zamuda 1987]. For example, the uptake of copper by oysters is largely controlled by the copper free ion concentration of the medium, and to a lesser extent by the copper content of the food [Zamuda 1984]. However, other chemical forms may be available. For example, an uncharged complex may provide a second route of access to the cell. For the grass shrimp *(Palaemonetes pugio)* [Engel et al. 1981] and several species of phytoplankton [Sanders and Abbe 1987a,b], it appears that the calculated activity of the uncharged species AgCl better explains the availability and toxicity of silver than does the calculated free silver ion activity.

For anionic trace elements (e.g., chromate, molybdate, selenate, arsenate, germanate), bioavailability is most often controlled by competition for uptake with a similar nutrient ion of greater abundance (i.e., sulfate, phosphate, or silicate). For those elements whose availability varies inversely with sulfate (chromate [Riedel 1984, 1985a], selenate [Schrift 1974; Wheeler et al. 1982], molybdate [Howarth and Cole 1985; Cole et al. 1986]), salinity plays a dominant role in bioavailability. This is shown in Figure 2 for the effect of salinity and sulfate on chromate uptake by phytoplankton [Riedel 1984]. Arsenate behaves as a phosphate analog [Blum 1966; Planas and Healy 1978; Sanders 1979], and germanate behaves as an analog of silicate [Azam and Volcani 1981], so the availability of these metals varies according to the concentration of their analog, which in turn is highly dependent on biological activity.

A third group of trace elements are those which form lipophilic organo-metallic compounds, such as methyl

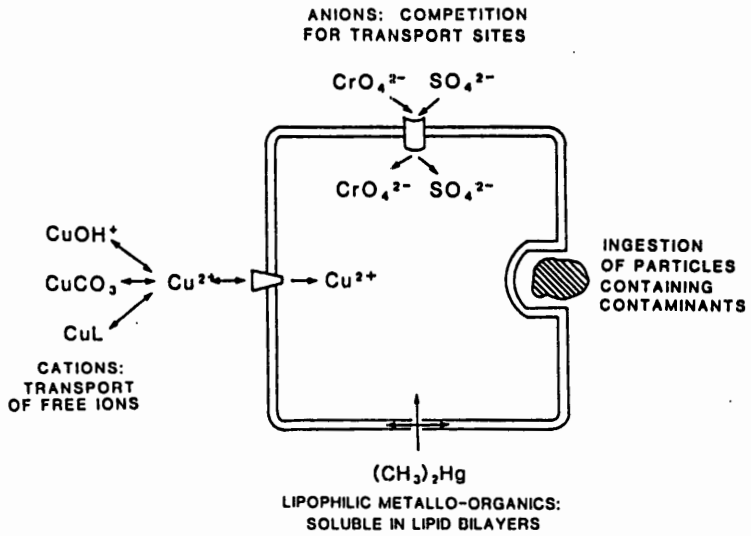

Figure 1. Dominant routes of uptake for three important trace element groups.

and dimethyl mercury, methyl and ethyl lead species, and methyl and butyl tin. There are few direct comparisons of the bioavailability of alkyl compounds and their inorganic precursors, but in general it appears that their uptake and toxicity is enhanced by their greater lipid solubility. For example, in a study of the uptake of inorganic and methylmercury by the diatom *Skeletonema costatum* and the copepod *Acartia clausi*, methylmercury resulted in much greater uptake of total mercury than inorganic mercury [Fujiki 1980]. For inorganic and organo-tin compounds, there is a general correlation between the octanol/water partition coefficient (P_{ow}), an index of the lipophilicity of the compound, and the toxicity to phytoplankton [Wong et al. 1982] and to animals [Zuckerman et al. 1978].

UPTAKE AND INCORPORATION

The ability of marine organisms to accumulate trace elements has been well documented; the literature is replete with studies of pollutant levels in organisms from the Chesapeake Bay [Young et al. 1980; Abbe 1982; Bieri et al. 1982; Di Guilio and Scanlon 1982; Hung 1982; Phelps et al. 1985; Abbe and Sanders 1986; Shigenaka and Calder 1987; Wright and Foster 1987]. Less well known, and currently under study, is the relative role of various uptake processes. For those organisms (primarily animals) that can live in a medium containing a dissolved toxic element, contact sediment with high concentrations of the element, and ingest other organisms that have already incorporated the element, the logical query is which source contributes most to the uptake of the toxic. The answer, of course, varies for different organisms and pollutants; however, in general, water appears to be the most important source of most toxics to the greatest number of organ-

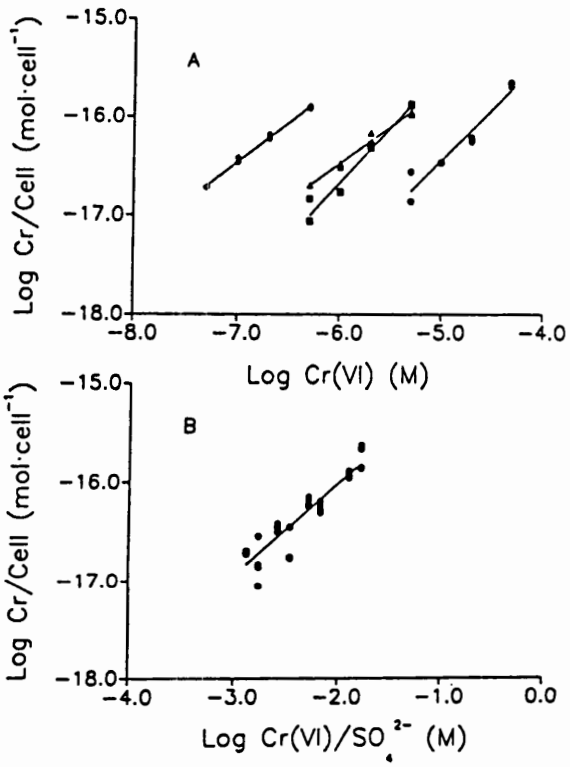

Figure 2. Uptake of chromate by phytoplankton at several combinations of salinity and sulfate concentration. (A) Uptake as a function of total chromate in medium. (B) Uptake as a function of the ration of sulfate to chromate in the medium [from Riedel 1985a].

isms. Much of the metal present in food is not passed across the gut wall. Generally speaking, uptake of contaminants from food is more important for larger animals, for which surface absorption is less important. Animals have been shown to take up large amounts of

metals from contaminated sediments, but it is usually not clear whether the materials have been ingested or absorbed directly from interstitial water.

The uptake of silver by oysters from Chesapeake Bay is solely from silver dissolved in water. Uptake from either algal food or sediment is insignificant in comparison [Sanders and Abbe 1986; Sanders and Abbe 1987c]. However, copper uptake by the same organism is more complex. Uptake of dissolved copper occurs readily, largely controlled by the availability of the free copper ion. Copper is also taken up from food, although to a lesser extent. It appears that colloidal organo-cupric complexes are also available [Zamuda and Sunda 1982; Zamuda 1984; Zamuda et al. 1985].

Through the predator/prey relationship, the potential exists for toxic trace elements to be passed up the food chain from autotrophs to herbivores to carnivores. In past years, such food chain transfer was thought to lead to increased body burdens at each step in the chain, resulting in top carnivores with extremely high concentrations of toxic metals, toxic responses within the population, and even potential toxicity to human consumers. Recent work, however, has shown that most trace elements do not get magnified as they are passed up the food chain [Young 1984]. Biomagnification through food appears to be a significant factor for very few trace elements; methylmercury [Fowler 1982] appears to be a notable exception.

Aside from the concentration and sequestering of trace elements, organisms have the capability to alter the chemical form or partitioning of many trace elements. Such shifts alter biological reactivity and toxicity of these elements and can alter their rate of transport through the estuary as well as their eventual fate. Potential metal/phytoplankton transformations are illustrated in Figure 3.

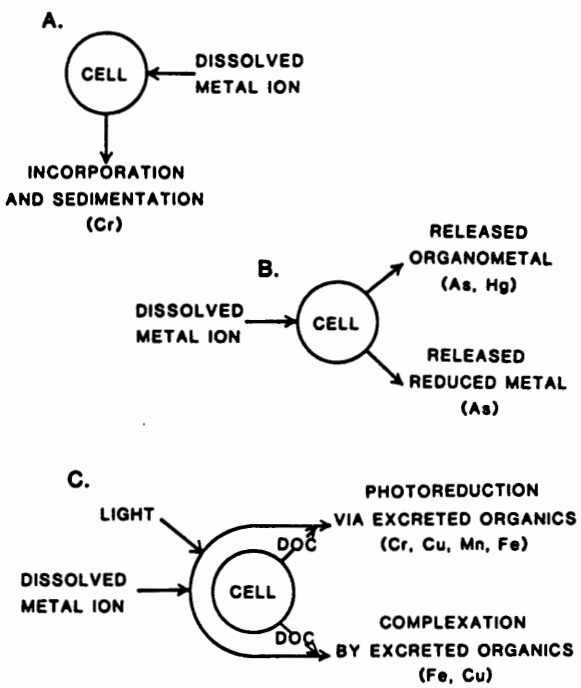

Figure 3. Possible interactions of dissolved metal ions with phytoplankton, including changes in partitioning (A), direct transformation of chemical speciation (B), and indirect facilitation of complexation and photoreduction (C).

FACTORS THAT ALTER BIOAVAILABILITY OF TRACE ELEMENTS AT FIXED CONCENTRATIONS

Ionic strength

In concentrated salt solutions, the activity of an ion is reduced because of the interaction of its charge with other nearby ions. In general, the individual ion activity coefficient ranges from 1.1 for uncharged species to approximately 0.1 for triply charged species in full-strength seawater [Stumm and Morgan 1970]. In practice, changes in bioavailability due to ionic strength alone must be considered in conjunction with changes due to inorganic and organic complexation.

Inorganic complexation

A number of ions are present in seawater in almost constant ratio to one another. Although these ions are not particularly strong complex formers, their concentrations in seawater are high enough that the most predominant chemical species of most trace elements are ion pairs or other complexes with these ions. Other ligands (e.g., sulfide) are of biological origin and have wide variation independent of salinity. Under some circumstances, however, they may be important in determining the chemical speciation of inorganic pollutants.

Inorganic complexation of the various trace elements differs greatly. For example, the inorganic speciation of copper is dominated by complexation by hydroxide and carbonate species; it is only slightly complexed by the very abundant chloride ion [Sunda 1975]. Silver and cadmium, however, have comparatively high binding constants for chloride, and their inorganic speciation is dominated by chloride complexes [Engel et al. 1981; Jenne et al. 1978; Sanders and Abbe 1987b]. The binding of various ions is a highly interdependent process since they compete for common ligands; however, the majority of the important equilibrium constants are well enough known that computer models can readily estimate the binding by inorganic ligands.

Organic complexation

In addition to the inorganic ligands present in aquatic systems, organisms also produce, either through excretion, leakage or decay, a variety of organic compounds that may also complex and reduce the free ion activity of trace elements. Dissolved organic compounds (DOC) range from 0.1 mg l^{-1} in unpolluted freshwater to 1-2 mg l^{-1} in seawater to 10 mg l^{-1} and higher in highly productive or polluted water. Concentrations in the Chesapeake Bay range from 1.9 to 13 mg l^{-1} [Sanders 1982; Sigleo and Helz 1982; Newell 1983; Zamuda 1984; Newell and Sanders 1986]. The composition of the organic matter varies depending on the source, but it contains many functional groups with metal-complexing capabilities. Unlike the situation with inorganic complexing ions, natural organics contain a large number of compounds of varying complexing strengths [Dzombak et al. 1986; Fish et al. 1986]. Therefore, except for a few studies on specific chelators found to be excreted by algae or released into the environment by human activity, the studies of complexation by natural DOC are largely empirical.

Only a few metals (iron, copper, zinc, and mercury, which have the greatest affinity for ligands) are complexed significantly by natural DOC. Copper is organically complexed to a large extent (50-98%) [Hanson and Quinn 1983; Zuelke and Kester 1983; Mills and Quinn 1984; Sunda and Hanson 1987], by a variety of dissolved organic compounds, some of which are relatively labile, and some of which are kinetically inert. In fresh water, where the sources of DOC are highly variable, there is little correlation between DOC and copper complexation capacity between different sites; however, in marine and estuarine waters, including the Chesapeake Bay, where the DOC is largely of autochthonous origin and is more homogeneous on spatial and temporal scales, there is a significant correlation between DOC concentration and copper complexation capacity [Newell and Sanders 1986; van den Berg et al. 1987].

pH

Many of the inorganic compounds important in the complexation of some toxic metals (e.g. carbonate, phosphate, and sulfate), as well as many of the pollutants, have equilibria dependent on pH within the pH range of aquatic systems [Zirino and Yamamoto 1972]. Control of the relative abundance of such species through these equilibria is perhaps the most direct way in which pH determines the bioavailability of pollutants. In addition, the active sites of most organic complexing agents are pH-sensitive groups (e.g., carboxylic acids, amines, and sulfhydryl residues), so that pH has a strong influence on the extent to which organics complex toxic trace elements. Finally, the active sites of enzymes and uptake sites of trace elements contain the same variety of pH-sensitive groups, and the activity of these systems varies with pH as well. Thus, the effect of pH on the availability of a given element to a particular organism in a specific site depends on a variety of effects. Some of these effects (e.g., the effect of pH on inorganic complexation) are rather straightforward to predict, whereas others (the effect of pH on organic complexation and on uptake of the metal) are more difficult.

Algal photosynthesis results in the removal of inorganic carbon from the water column and a resulting increase in pH; during bloom events, such increases can be substantial. In the freshwater section of the Potomac River during 1984, a persistent bloom of *Microcystis* sp. caused substantial variability in pH, with maximum pH of >10 [Seitzinger 1986]. Such a large shift in pH can greatly affect the bioavailability of trace elements.

PROCESSES AFFECTING TRACE ELEMENT CONCENTRATION, DISTRIBUTION, AND FORM

In the previous section we have discussed a number of factors that can affect the bioavailability of a trace element to estuarine organisms at a fixed total concentration. There are also factors that control the total concentration, distribution and form of the element to which organisms are exposed in the Bay. These are discussed below, and are diagrammed in Figure 4.

Sources

There are several sources of toxic trace elements to the Chesapeake Bay (Table 1). One of the most important is runoff of fresh water from the land. Industry, agriculture, and municipal effluents each contributes a significant load of compounds. Many toxic trace elements are present in the air as suspended particles (e.g., copper and zinc) or as vapors (e.g., mercury, lead, selenium, and arsenic) and can be deposited by rain or through dry fall. Aeolian flux of some metals has been estimated for Chesapeake Bay, and although the loadings are not large compared to runoff, they are significant. Aeolian inputs may be a particularly significant source to the surface microlayer where a variety of toxics are concentrated, and to areas remote from industry or other concentrated sources.

Distribution

Estuaries are extremely dynamic systems, moving and changing constantly in response to winds, tides, and runoff, so it is difficult to discuss the distribution of a given trace element in the water and sediment except on a statistical basis. Nevertheless, some general trends in

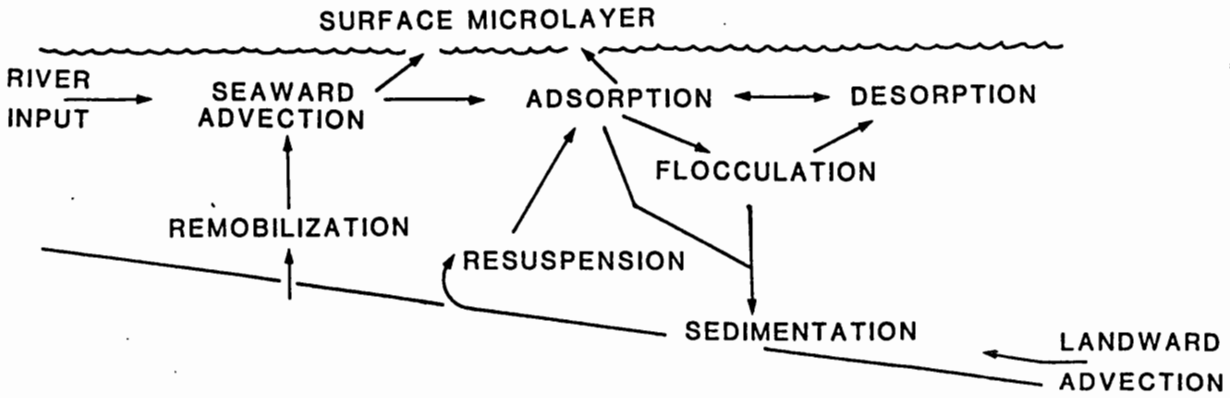

Figure 4. Processes affecting transport and availability of trace elements in estuaries.

trace element distributions can be discussed. Elements whose sources are in fresh water are inversely correlated to salinity; that is, mean concentrations decrease going from the head of the Bay to the ocean, and from tributaries to the main Bay. This pattern has been observed for metals such as cadmium, copper, and lead [Kingston et al. 1982]. Conversely, for those few trace elements whose concentrations are normally higher in seawater than in fresh water, a reverse trend is observed, with bottom water and more saline waters having higher concentrations [Kingston et al. 1982; Sanders 1985]. Elements that are reduced and solubilized in sediments (e.g., iron, manganese, arsenic, and zinc) also tend to be enriched in deep waters of the Bay [Carpenter et al. 1975; Kingston et al. 1982; Troup and Bricker 1975; Sanders 1985].

The phenomenon of seasonal anoxia in the Chesapeake Bay is undoubtedly a dominant factor in the spatial and temporal distributions of redox-reactive elements such as arsenic, chromium, iron, manganese, and selenium. Unfortunately, virtually no published studies on this aspect of the anoxia are available yet. This should be an active area of research in the near future.

The incorporation of a trace element by biota may alter its transport within the estuary. In general, because organisms try to maintain position within a specific environment, incorporation within tissues will lead to the contaminant remaining within the estuary. For example, silver has a high affinity for particle surfaces and is rapidly taken up and incorporated by phytoplankton as well as by suspended sediments. As these particles move down the estuary, the silver is desorbed from the suspended particles and remains in a dissolved state, complexed with chloride. Silver associated with phytoplankton, however, does not desorb, but remains in the Chesapeake Bay and probably recycles [Sanders and Abbe 1987a,b].

Surface films

Trace elements can become concentrated in two areas within water bodies: the sediment and the surface microlayer. The surface microlayer is the upper 50 μm to 1 mm surface film that lies between the water and the atmosphere. Many organic compounds and contaminants concentrate there, due to their hydrophobic nature [Baier et al. 1974; Hardy 1982; Hardy et al. 1986]. The film also contains a variety of particles, including a unique fauna and flora [Hardy et al. 1986]. In addition, the presence of natural organics ensures the presence of trace contaminants that normally bind to organic complexing agents [Garrett and Duce 1980]. Recent studies

Table 1. Estimated sources of toxic trace elements, in metric tons per year, to the Chesapeake Bay. Data taken from Bieri et al. [1982].

Source	Cd	Cr	Cu	Ni	Pb	Zn	Fe	Mn
Major tributaries	75	551	517	402	307	1,444	199,683	19,000
Industry	178	200	190	ND	155	167	2,006	ND
Municipal wastewater	6	200	99	ND	68	284	625	ND
Urban runoff	7	10	9	20	111	63	977	22
Shore erosion	1	83	29	ND	28	3	57,200	ND
Wet fall	3	ND	28	25	34	825	87	5

in the Chesapeake Bay have found high concentrations of many pollutants, including metals, in the surface microlayer [Bellama and Zoller 1983; Gucinski 1986; Hall et al. 1986; Hardy et al. 1987]. Because of the high concentrations and the potential for direct uptake by biota, this layer may represent an efficient and important transfer mechanism for pollutants. If so, this mechanism may be particularly important for periodically emergent organisms, such as sedentary animals in the tidal zone, or emergent vegetation or neuston, including commercially important fish eggs.

Adsorption

Adsorption reactions may be the important controlling factors for the distribution of many toxic trace metals in estuaries, including copper, lead, and zinc [Harris et al. 1975]. These reactions are partly reversible, so that the material can be desorbed from solids by lowering the concentration of the pollutant in solution, or by changing the chemical system to favor the release (e.g., a change in pH, or an increase in competing ions).

Flocculation

Suspended solids and colloids carrying adsorbed trace elements can aggregate to form larger particles, which become more susceptible to the processes of sedimentation and filtration. An important component of the suspended solids in most river systems is clay particles. Clay particles aggregate quite slowly in fresh water but readily in seawater [Stumm and Morgan 1970]. Flocculation can lead to segregation and concentration of trace metals. In studies of metal interactions with Chesapeake Bay flocs, most metals were concentrated in the smaller size fractions [Gibbs 1982, 1986].

Sedimentation

The sediments are the ultimate repository of most particles, including large amounts of toxic trace elements. In particular, the sediments of the turbidity maximum zone downstream of a major source are often a major repository. Within the Chesapeake Bay, Officer et al. [1984] and Helz et al. [1985] determined that most of the particulate material and associated toxics entering from the Susquehanna River is deposited at the head of the Bay.

The combined processes of adsorption, flocculation, and sedimentation are no doubt largely responsible for the high concentration of toxics found in the sediments near sources of pollution [Huggett et al. 1971; Pheiffer 1972; Owens et al. 1974; Villa and Johnson 1974; Goldberg et al. 1978; Sinex and Helz 1981; Wong and Moy 1984; Di Guilio and Scanlon 1985; Wright and Foster 1987]. For example, the concentrations of arsenic, copper, manganese, nickel, lead, tin, and zinc in Chesapeake Bay sediments decline seaward from maxima at Baltimore Harbor, Susquehanna River, and Elizabeth River [Helz and Huggett 1987].

Organisms within the water column and on the bottom produce aggregated fecal pellets containing unassimilated organic matter and inadvertently ingested sediment particles. Studies of zinc have indicated that rapid recycling between the sediment and water column takes place in the northern Chesapeake Bay. This cycle has been attributed to uptake by phytoplankton, ingestion by zooplankton, and rapid return to the sediments in fecal pellets [Carpenter et al. 1975].

Remobilization

Incorporation into sediments and subsequent burial does not forever remove trace elements from possible uptake by organisms. Many organisms reside in the sediments, and are thus exposed to the toxic material. Toxic substances may be present in the pore waters of sediment in higher concentration than the water column. For example, arsenic present in water as arsenate can be adsorbed onto surface sediments. When buried in reducing sediments the arsenic is reduced to arsenite, which is present in much higher concentration in the interstitial waters than in the water column. In the Patuxent River, the arsenite concentration in the pore waters is about 50 times higher than arsenate in the surface waters [Riedel et al. 1987]. Other metals coprecipitated with iron and manganese oxyhydroxides, such as copper, zinc, and lead, can be remobilized in pore waters, particularly in the northern Chesapeake Bay, where low sulfate concentrations result in less sulfide formation under reducing conditions [Carpenter et al. 1975; Troup and Bricker 1975]. Cadmium also may be remobilized from Chesapeake Bay sediments [Helz et al. 1975].

Trace elements in sediment pore waters can also be returned to the water column by diffusion, resuspension of the sediments, or the activities of benthic organisms. Benthic infauna alter the transport and availability of pollutants in sediments in several ways. Organisms may change the distributions of contaminants in sediments in a variety of ways, including mechanical disturbance (mixing, pelletization, and sorting), chemical changes (increased oxygen penetration or organic enrichment), increase of microbial activity, ingestion of sediment constituents, increase of surface area, and direct uptake of metals [Rice and Whitlow 1985]. Burrowing activities of *Nereis succinea* in contaminated sediments, for example, increased the flux of arsenic by a factor of 5 [Riedel et al. 1987]. This enhancement corresponded to an approximately equal increase in the surface area of the sediment due to worm burrows (Figure 5).

Redox reactions and equilibria

Several of the toxic trace elements participate in redox reactions that alter their biological availability. Most of the important redox reactions of trace elements are relatively slow compared with inorganic and organic complexation reactions.

In the Chesapeake Bay, organic carbon from the highly productive surface waters fuels a strong oxygen demand in the sediments and bottom waters. This demand produces anoxic, sulfide-rich interstitial water and seasonally anoxic bottom waters in the deep channel of the Bay. In such waters a variety of metals (arsenic, chromium, iron, manganese, and selenium) can be reduced to forms different from the oxidized form commonly present. Arsenic, for example, is most stable in oxidized waters as the arsenate ion. However, in anoxic bottom and interstitial waters, arsenate is largely reduced to arsenite [Peterson and Carpenter 1983; Sanders 1985; Riedel et al. 1987]. Arsenite in the sediments and bottom waters can diffuse or be mixed up into the surface waters, where it slowly oxidizes back to arsenate. Arsenate is primarily available and toxic to phytoplankton, which have a requirement for phosphate, whereas arsenite is more toxic to fauna [Peoples 1975].

Indirect evidence suggests that sulfide is present in oxidized seawater and in the Chesapeake Bay at concentrations of 10^{-10} to 10^{-11} M [G. Cutter, personal communication; Elliot et al. 1985], enough to have significant effects on the speciation of trace metals such as silver and mercury. However, standard analytical techniques are not sensitive enough to measure sulfide at those concentrations. Moreover, in the Chesapeake Bay, where there is occasional anoxia and a constant source of sulfide diffusing from the sediments, concentrations are likely to be more variable.

Another source of reduced metal ions is photoreduction. The interaction of light, or ultraviolet radiation, on DOC or on metal organic complexes can either directly reduce certain metal ions or produce redox active compounds such as peroxide or superoxide that can reduce metals. It has been shown that peroxide concentrations in the Patuxent River undergo a diurnal cycle, with concentrations increasing in the daylight hours and decreasing at night [Kieber and Helz 1986]. Peroxide, in turn, has been shown to reduce chlorine from industrial sources to chloride [Helz and Kieber 1985], and reduce some copper(II) to copper(I) [Moffet and Zika 1983]. Chromate is reduced photochemically in the presence of high DOC levels [Riedel 1985b; G. Helz and R. Kieber, personal communication]. Since photochemical reactions are favored by the presence of high DOC and abundant light, it is likely to be most important in shallow, organic-rich estuarine systems.

Organo-metallic compounds

Several metals (antimony, arsenic, lead, mercury, selenium, and tin) are present in the Chesapeake Bay partly as covalently bound organo-metallic compounds (e.g., methylarsonic and dimethylarsinic acids [Sanders 1985, 1986; Riedel et al. 1987], alkyl lead, methylmercury [Zoller et al. 1983], organic selenium [Takayanagi and Wong 1984], and methyl and butyltin compounds [Hallas and Cooney 1981; Hallas et al. 1982; Brinckman et al. 1983; Gilmour et al. 1985; Hall et al. 1986; Westbrook et al. 1986]). In some cases (methylarsenic compounds, methylmercury, organic selenium, and methyltins), these compounds are formed in the environment from inorganic metal by the local biota. Metals can also be methylated by exchange reactions with other methylated compounds [Manders et al. 1984; Bellama et al. 1985; Brinckman et al. 1985].

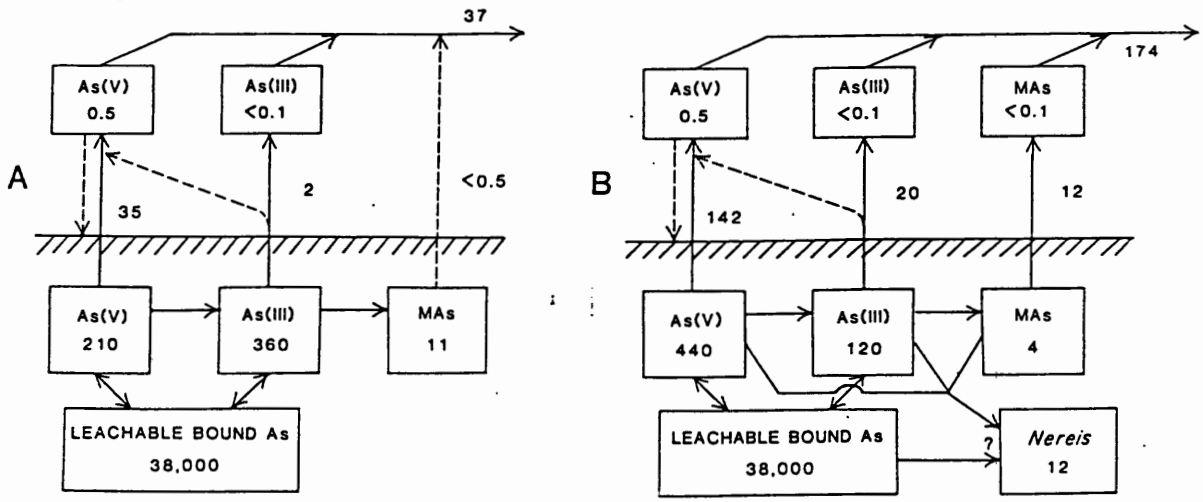

Figure 5. Cycling of arsenic in undisturbed and bioturbated estuarine sediment contaminated with arsenic [from Riedel et al. 1987].

For example, biological uptake of arsenic leads to the production of reduced and methylated arsenic compounds, some of which are more toxic to higher trophic levels than was the original arsenic compound [Peoples 1975; Sanders 1980; Sanders 1985]. Within the Chesapeake Bay, large fractions (up to 80%) of the arsenic may be present in these reduced or methylated forms (Figure 6).

SUMMARY AND RESEARCH DIRECTIONS

The overall conclusion must be that the complexity of the interactions between physical, chemical, and biological factors is extreme. This complexity hampers our ability to fully understand (and more important, predict) pollutant transport, availability, uptake, and impact.

Past efforts have focused largely upon determining the concentration of contaminants within the various compartments of the system. Although this research has value, we must move more toward careful study of the processes themselves.

ACKNOWLEDGEMENTS

The preparation of this article was supported by the Academy of Natural Sciences.

QUESTIONS

Q: Did you measure oxic vs. anoxic events in your bioturbation studies?
A: No, these were strictly laboratory experiments; but the sediments were mostly anoxic. The worms made themselves an oxic area, and it may be that this increase of surface area exposed to oxygen accounts for the increase in flux.

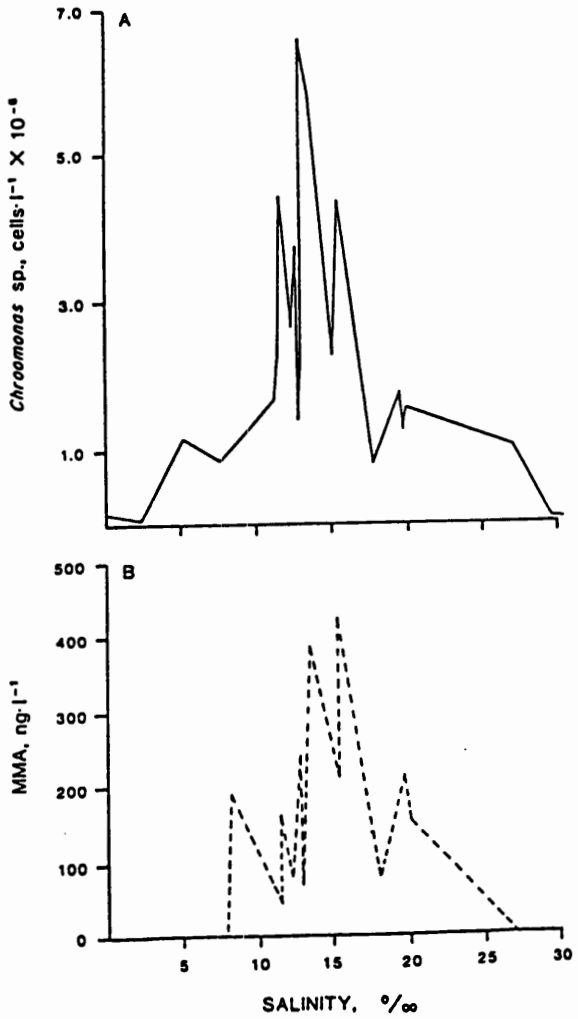

Figure 6. Correlation between species composition of phytoplankton in Chesapeake Bay with methylated arsenic [from Sanders 1985].

RELEVANT LITERATURE

Abbe, G.R. 1982. Growth, mortality and copper-nickel accumulation by oysters, *Crassostrea virginica*, at the Morgantown steam electric station on the Potomac River, Maryland. J. Shellfish Res. 2: 3-13.

Abbe, G.R. and J.G. Sanders. 1986. Condenser replacement in a coastal power plant: copper uptake and incorporation in the American oyster, *Crassostrea virginica*. Mar. Environ. Res. 19: 93-113.

Anderson, D.M. and F.M.M. Morel. 1978. Copper sensitivity of *Gonyaulax tamarensis*. Limnol. Oceanogr. 23:283-295.

Anderson, M.A. and F.M.M. Morel. 1982. The influence of aqueous iron chemistry on the uptake of iron by the coastal diatom, *Thalassiosira weissflogii*. Limnol. Oceanogr. 27:789-813.

Azam, F. and B.E. Volcani. 1981. Germanium-silicon interactions in biological systems, p. 43-67. In: T.L. Simpson and B. E. Volcani (eds.), Silicon and siliceous structures in biological systems. Springer-Verlag.

Baier, R.E., D.W. Goupil, S. Perlmutter and R. King. 1974. Dominant chemical composition in sea-surface films, natural slicks, and foams. J. Rech. Atmos. 8:571-600.

Bellama, J.M., and W.H. Zoller. 1983. An investigation of the character of and constituents of slicks on water surfaces near power plants in the Chesapeake Bay. Report to MD Power Plant Siting Program, PPRP -83.

Bellama, J.M., K.L. Jewett and J.D. Nies. 1985. Rates of methylation of mercury(II) species in water by organotin and organosilicon compounds, p.239-247. In: A. E.

Martell and K. J. Irgolic (eds.), Environmental inorganic chemistry. Verlag Chemie.

Bieri, R., O. Bricker, R. Byrne, R. Diaz, G. Helz, J. Hill, R. Huggett, R. Kerhin, M. Nichols, E. Reinharz, L. Schaffner, D. Wilding and C. Strobel. 1982. Toxic substances, p. 263-375. In: E.G. Macalaster, D.A. Barker and M. Kasper (eds.), Chesapeake Bay Program technical studies: a synthesis. EPA, Washington, D. C.

Blum, J.J. 1966. Phosphate uptake by phosphate starved Euglena. J. Gen. Physiol. 49:1125-1136.

Blust, R., E. Verheyen, C. Doumen, and W. Decleir. 1986. Effects of complexation by organic ligands on the bioavailability of copper to the brine shrimp, *Artemia* sp. Aq. Toxicol. 8:211- 221.

Brinckman, F.E., G.J. Olson and J.S. Thayer. 1985. Biological mediation of marine metal cycles, p.239-258. In: A. Sigleo and A. Hattori (eds.), Marine and estuarine geochemistry, Lewis Publ.

Brinckman, F.E., J.A. Jackson, W.R. Blair, G.J. Olson and W. P. Iverson. 1983. Ultratrace speciation and biogenesis of methyltin transport in estuarine waters, p. 39-72. In: C.S. Wong, E. Boyle, K.W. Bruland, J.D. Burton, and E.D. Goldberg (eds.), Trace metals in sea water. Plenum.

Carpenter, J.H., W.L. Bradford and V. Grant. 1975. Processes affecting the composition of estuarine waters (HCO_3, Fe, Mn, Zn, Cu, Ni, Cr, Co, and Cd), p. 188-214. In: L.E. Cronin (ed.), Estuarine Research, Vol. 1. Academic Press.

Chau, Y.K. and P.T.S. Wong. 1978. Occurrence of biological methylation of elements in the environment, p. 39-53. In: F.E. Brinckman and J.M. Bellama (eds.), Organometals and organometalloids ocurrence and fate in the environment. ACS Symposium series 82.

Church, T.M., J.M. Tramontano, J.R. Scudlark, T.D. Jickells, J.J. Tokos, Jr., A.H. Knap, and J.N. Galloway. 1984. The wet deposition of trace metals to the western Atlantic Ocean at the mid-Atlantic coast and on Bermuda. Atmos. Environ. 18:2657-2664.

Cole, J.J., R.W. Howarth, S.S. Nolan and R. Marino. 1986. Sulfate inhibition of molybdate assimilation by planktonic algae and bacteria: some implications for the aquatic nitrogen cycle. Biogeochemistry 2:179-196.

Di Guilio, R.T. and P.F. Scanlon. 1985. Heavy metals in aquatic plants, clams, and sediments from the Chesapeake Bay, U. S. A. Implications for the waterfowl. Sci. Tot. Environ. 41:259-274.

Dzombak, D.A., W. Fish and F.M.M. Morel. 1986. Metal-humate interactions. 1. Discrete ligand and continuous distribution models. Environ. Sci. Technol. 20:669-675.

Elliot, S., E. Lu, and F.S. Rowland. 1985. Marine tropospheric H_2S and OCS imply significant levels of sulfide anions in seawater. Trans. Amer. Geophys. Union 66: 1309.

Engel, D.W. and W.G. Sunda. 1979. Toxicity of cupric ion to the eggs of the spot *Leiostomus xanthurus* and the Atlantic silverside *Menidia menidia*. Mar. Biol. 50:121-126.

Engel, D.W., W.G. Sunda and B.A. Fowler. 1981. Factors affecting trace metal uptake and toxicity to estuarine organisms. I. Environmental parameters, p. 127- 144. In: W.B. Vernberg, F.P. Thurberg, A. Calabrese, and F.J. Vernberg (eds.), Biological monitoring of marine pollutants. Academic Press.

Fish, W., D.A. Dzombak and F.M.M. Morel. 1986. Metal-humate interactions. 2. Application and comparison of models. Environ. Sci. Technol. 20:676-683.

Fowler, S.W. 1982. Biological transfer and transport processes, p.1-66. In: G. Kullenberg (ed.), Pollutant transfer and transport in the sea, Vol. II. CRC Press.

Fujiki, M. 1980. Methylmercury accumulation in plankton and fish, p. 485-492. In: R.A. Baker (ed.), Contaminants and sediments, Vol. 2. Ann Arbor Science.

Garrett, W.D. and R.A. Duce. 1980. Surface microlayer samplers, p. 471-490. In: F. Dobson, L. Hasse, and R. Davis (eds.), Air-sea interaction instruments and methods. Plenum Press.

Gibbs, R.J. 1982. The differentiation of toxic metals in estuaries. Thalassia Jugosl. 18:181-190.

Gibbs, R.J. 1986. Segregation of metals by coagulation in estuaries. Mar. Chem. 18:149-159.

Gilmour, C.C., J.H. Tuttle, and J.C. Means. 1985. Tin methylation in sulfide bearing sediment, p. 239-258. In: A. Sigleo and A. Hattori (eds.), Marine and estuarine geochemistry. Lewis Publishers.

Goldberg, E.D., V. Hodge, M. Koide, J.J. Griffen, E. Gamble, O.P. Bricker, G. Mattisoff, G.R. Holdren, and R. Braun. 1978. A pollution history of the Chesapeake Bay. Geochim. Cosmochim. Acta 42:1413-1425.

Gucinski, H. 1986. The effect of sea surface microlayer enrichment on TBT transport, p. 1266-1274. In: Oceans 86 Conference proceedings, Vol. 4. Washington D.C.

Hall, L.W., M.J. Lenkevich, W.S. Hall, A.E. Pinkney, and S.J. Bushang. 1986. Monitoring organotin concentrations in Maryland waters of Chesapeake Bay, p. 1275-1279. In: Oceans 86 Conference proceedings, Vol. 4. Washington, D.C.

Hallas, L.E. and J.J. Cooney. 1981. Tin and tin resistant microorganisms in Chesapeake Bay. Appl. Environ. Microbiol. 41:446-471.

Hallas, L.E., J.C. Means, and J.J. Cooney. 1982. Methylation of tin by estuarine microorganisms. Science 215:1505-1507.

Hanson, A.K., Jr. and J.G. Quinn. 1983. The distribution of dissolved and organically complexed copper and nickel in the Middle Atlantic Bight. Can. J. Fish. Aquat. Sci. 40 (Suppl. 2):151-161.

Hardy, J.T. 1982. The sea-surface microlayer: biology, chemistry and anthropogenic enrichment. Prog. Oceanogr. 11:307-328.

Hardy, J.T., E.A. Crecelius, and R. Kocam. 1986. Concentra-

tion and toxicity of sea-surface contaminants in Puget Sound. Final report to NOAA, OAD.

Hardy, J.T., E.A. Crecelius, L.D. Antrim, V.L. Broadhurst, P.D. Boehm, and W.G. Steinhauer. 1987. Aquatic surface contamination in Chesapeake Bay. Draft final report to MD DNR, Power Plant Research Program.

Harris, R.L., G.R. Helz, and R.L. Cory. 1975. Processes affect the vertical distribution of trace components in the Chesapeake Bay, p. 176-185. In: T.M. Church (ed.), Marine chemistry in the coastal environment. ACS Symposium series 18.

Harrison, G.I. and F.M.M. Morel. 1983. Antagonism between cadmium and iron in the marine diatom *Thalassiosira weissflogii*. J. Phycol. 19:495-507.

Helz, G.R., and R.J. Huggett. 1987. Contaminants in Chesapeake Bay; the regional perspective. In: S.K. Majumdar, L.W. Hall, and H. Austin (eds.), Contaminant problems and management of living Chesapeake Bay resources. Penn. Academy Sci., in press.

Helz, G.R. and R.J. Kieber. 1985. Hydrogen peroxide: a minor but significant contribution to chlorine demand, p. 1033-1040. In: R.L. Jolley, R.J. Bull, W.P Davis, S. Katz, M.H. Roberts, and V. Jacobs (eds.), Water chlorination: environmental impact and health effects, Vol. 5. Lewis Publ.

Helz, G.R., R.J. Huggett, and J.M. Hill. 1975. Behavior of Mn, Fe, Cu, Zn, Cd, and Pb discharged from a wastewater treatment plant into an estuarine environment. Water Res. 9:631-636.

Helz, G.R., S.A. Sinex, K.L. Ferri, and M. Nichols. 1985. Processes controlling Fe, Mn, and Zn in sediments of northern Chesapeake Bay. Est. Coast. Shelf Sci. 21:1-16.

Howarth, R.W. and J.J. Cole. 1985. Molybdenum availability, nitrogen limitation and phytoplankton growth in natural waters. Science 229:653-655.

Huggett, R.J., M.E. Bender, and H.D. Slone. 1971. Mercury in sediments from three Virginia estuaries. Ches. Sci. 12:280-282.

Huggett, R.J., F.A. Cross, and M.E. Bender. 1975. Distribution of copper and zinc in oysters and sediments from three coastal plain estuaries, p. 224-238. In: F.G. Howell, J.B. Gentry, and M.H. Smith (eds.), Mineral cycling in southeastern ecosystems. NTIS CONF-740513.

Hung, Y-W. 1982. Effects of temperature and chelating agents on cadmium in the American oyster. Bull. Environ. Contam. Toxicol. 28:546-551.

Jenne, E.A., D.C. Girvin, J.W. Ball, and J.M. Burchard. 1978. Inorganic speciation of silver in natural waters—fresh to marine, p. 41-61. In: D.A. Klein (ed.), Environmental impacts of artificial ice nucleating agents. Dowden, Hutchinson, & Ross.

Kieber, R.J. and G.R. Helz. 1986. Two method verification of hydrogen peroxide determinations in natural waters. Anal. Chem. 58:2312-2315.

Kingston, H.M., R.R. Greenberg, E.S. Beary, B.R. Hardas, J.R. Moody, T.C.Rains, and W.S. Liggett. 1982. The characterization of the Chesapeake Bay: A systematic analysis of toxic trace elements. EPA-79-D-X-0717.

Macek, K.J., S.R. Petrocelli and B.H. Sleight III. 1979. Consideration in assessing the potential for, and significance of biomagnification of chemical residues in aquatic food chains. In: L.L. Marking and R.A. Kimrle (eds.) Aquatic Toxicology, ASTM special publication 667, American Society of Testing and Materials, Philadelphia, pp.251-268.

Manders, W.F., G.J. Olson, F.E. Brinckman, and J.M. Bellama. 1984. A novel synthesis of methyl tin tri-iodide with environmental implications. J. Chem. Soc. Commun. 1984:538-540.

Mills, G.L. and J.G. Quinn. 1984. Dissolved copper and copper-organic complexes in the Narragansett Bay estuary. Mar. Chem. 15:151-172.

Moffet, J.W. and R. G. Zika. 1983. Oxidation kinetics of Cu(I) in seawater: implications for its existence in the marine environment. Mar. Chem. 13:239-251.

Morel, F.M.M. and N.M.L. Morel-Laurens. 1983. Trace metals and plankton in the ocean, p. 841-870. In: C.S. Wong, E. Boyle, K.W. Bruland, J.D. Burton, and E.D. Goldberg (eds.), Trace metals in sea water. Plenum.

Newell, A.D. 1983. An investigation of copper-organic complexes in the Patuxent River, Maryland. Thesis, University of North Carolina, Chapel Hill.

Newell, A.D. and J.G. Sanders. 1986. The relative copper binding capacities of dissolved organic compounds in a coastal-plain estuary. Environ. Sci. Technol. 20:817-821.

Nichols, M., R. Harris, and G. Thompson. 1982. Fate, transport, and transformation of toxic substances: significance of suspended sediment and fluid mud. EPA-600/8-82-089.

Officer, C.B., D.R. Lynch, G.H. Setlock, and G. R. Helz. 1984. Recent sedimentation in Chesapeake Bay, p. 131-158. In: V.S. Kennedy (ed.), The estuary as a filter. Academic Press.

Owens, J.P., K. Stefansson, and L.A. Sirkin. 1974. Chemical, mineralogic, and palynologic character of the Upper Wisconsinan-Lower Holocene fill in parts of Hudson, Delaware, and Chesapeake estuaries. J. Sed. Pet. 44:390-408.

Peoples, S.A. 1975. Review of arsenical pesticides, p. 1-12. In: E.A. Woolson (ed.), Arsenical pesticides. ACS Symposium series 7.

Peterson, M.L. and R. Carpenter. 1983. Biogeochemical processes affecting total arsenic and arsenic species distributions in an intermittently anoxic fjord. Mar. Chem. 12:295-321.

Pheiffer, T.H. 1972. Heavy metal analyses of bottom sediment in the Potomac River estuary. EPA Region III Technical Report 49.

Phelps, H.L., D.A. Wright, and J.A. Mihursky. 1985. Factors affecting trace metal accumulation by estuarine oysters, *Crassostrea virginica*. Mar. Ecol. Prog. Ser. 22:187-197.

Planas, D. and F.P. Healey. 1978. Effects of arsenate on growth and phosphorus metabolism of phytoplankton. J. Phycol. 14:337-341.

Rice, D.L. and S. I. Whitlow. 1985. Early diagenesis of transition metals: a study of metal partitioning between macrofaunal populations and shallow sediments, p. 21-30. In: The fate and effects of pollutants. MD Sea Grant publ. UM-SG-TS-85-02.

Riedel, G.F. 1984. Influence of salinity and sulfate on the toxicity of chromium (VI) to the estuarine diatom, *Thalassiosira pseudonana*. J. Phycol. 20:496-500.

Riedel, G.F. 1985a. The relationship between chromium (VI) uptake, sulfate uptake, and chromium (VI) toxicity in the estuarine diatom *Thalassiosira pseudonana*. Aq. Toxicol. 7: 191-204.

Riedel, G.F. 1985b. Photoreduction of Cr(VI) in natural waters.Trans. Amer. Geo. Union 66:1257.

Riedel, G.F., J.G. Sanders, and R.W. Osman. 1987. The effect of biological and physical disturbances on the transport of arsenic from contaminated estuarine sediments. Est. Coast. Shelf. Sci. 25:693-706.

Rueter, J.G. and F.M.M. Morel. 1981. The interaction between zinc deficiency and copper toxicity as it affects silicic acid uptake mechanisms in the marine diatom *Thalassiosira pseudonana*. Limnol. Oceanogr. 26:67-73.

Sanders, J.G. 1979. Effects of arsenic speciation and phosphate concentration on arsenic inhibition of *Skeletonema costatum* (Bacillariophyceae). J. Phycol. 15:424-428.

Sanders, J.G. 1980. Arsenic cycling in marine systems. Mar. Environ. Res. 3:257-266.

Sanders, J.G. 1982. Chlorination of estuarine water: the occurrence and magnitude of carbon oxidation and its impact on trace metal transport. Environ. Sci. Technol. 16:791-796.

Sanders, J.G. 1985. Arsenic geochemistry in Chesapeake Bay: dependence upon anthropogenic inputs and phytoplankton species composition. Mar. Chem. 17:329-340.

Sanders, J.G. 1986. Alteration of arsenic transport and reactivity in coastal marine systems after biological transformation. Rapp. P.-v. Reun. Cons. int. Explor. Mer 186:185-192.

Sanders, J.G. and G.R. Abbe. 1986. The processes and pathways determining silver uptake and accumulation in American oysters in Chesapeake Bay. Progress report for 1985 prepared for Baltimore Gas & Electric Co.

Sanders, J.G. and G.R. Abbe. 1987a. The role of suspended sediments and natural phytoplankton in the partitioning and transport of silver in estuaries. Cont. Shelf Res. 7:1357-1361.

Sanders, J.G. and G.R. Abbe. 1987b. Silver transport and impact in estuarine and marine systems. In: G. Suter and M. Lewis (eds.), 11 Symposium on aquatic toxicology and hazard assessment, in press.

Sanders, J.G. and G.R. Abbe. 1987c. The processes and pathways determining silver uptake and accumulation in American oysters in Chesapeake Bay. Progress report for 1986 prepared for Baltimore Gas & Electric Co.

Schrift, A. 1954. Sulfur-selenium antagonism. I. Antimetabolite action of selenate on the growth of *Chlorella vulgaris*. Am. J. Bot. 41:223-230.

Seitzinger, S. 1986. The effect of pH on the release of phosphorus from Potomac River sediments. Final report to EPA Chesapeake Bay Program. ANSP report 86-8F.

Seliger, H.H., J.A. Boggs, and W.H. Biggley. 1985. Catastrophic anoxia in the Chesapeake Bay in 1984. Science 228: 70-73.

Shigenaka, G. and J.A. Calder. 1987. Organic compounds and metals in selected fish, bivalve molluscs, and sediments of the Chesapeake Bay: preliminary results from the National Status and Trends Program, p. 69-80. In: M.P. Lynch and E.C. Krome (eds.), State of the Chesapeake Bay—Second annual monitoring report—Compendium. Chesapeake Research Consortium publ. 125.

Sigleo, A.C. and G.R. Helz. 1982. Composition of estuarine colloidal material: major and trace elements. Geochim. Cosmochim. Acta 45:2501-2509.

Sinex, S.A. and G.R. Helz. 1981. Regional geochemistry of trace elements in Chesapeake Bay sediments. Environ. Geol. 3:315-323.

Stumm, W. and J.J. Morgan. 1970. Aquatic chemistry. Wiley Interscience.

Sunda, W.G. 1975. Relationship between cupric ion activity and the toxicity of copper to phytoplankton. Dissertation, Massachusetts Institute of Technology.

Sunda, W.G. and R.R.L. Guillard. 1976. Relationship between cupric ion activity and toxicity of copper to phytoplankton. J. Mar. Res. 34:511-529.

Sunda, W.G. and A.K. Hanson. 1987. Measurement of free cupric ion concentration in seawater by a ligand competition technique involving copper sorption onto C18 SEP-PAK cartridges. Limnol. Oceanogr. 32:537-551.

Sunda, W.G., D.W. Engel, and R.M. Thuotte. 1978. Effect of chemical speciation on toxicity of cadmium to grass shrimp *Palemonetes pugio*: importance of free cadmium ion. Environ. Sci. Technol. 12:409-413.

Takayanagi, K. and G.T. Wong. 1984. Organic and colloidal selenium in Chesapeake Bay. Mar. Chem. 14:141-148.

Troup, B.N. and O.P. Bricker. 1975. Processes affecting the transport of materials from continents to oceans, p. 133-151. In: T.M. Church (ed.), Marine chemistry in the coastal environment. ACS Symposium series 18.

van den Berg, C.M.G., A.G. Merks and E.K. Duursma. 1987. Organic complexation and its control of the dissolved

concentrations of copper and zinc in the Scheldt estuary. Est. Coast. Shelf Sci. 24:785-798.

Villa, D., Jr. and P.G. Johnson. 1974. Distribution of metals in Baltimore Harbor sediments. EPA Technical Report 59.

Westbrook, D.J., E.J. Travelstead, F.A. Espourteille, C.D. Rice, and R.J. Huggett. 1986. Tributyltin in whole water and sediment collected from marinas and the Hampton Roads area in southern Chesapeake Bay. Final Report to Johns Hopkins University, Applied Physics Laboratory.

Wheeler, A.E., R.A. Zingaro, K. Irgolic, and N.R. Bottino. 1982. The effect of selenate, selenite, and sulphate on the growth of six unicellular algae. J. Exp. Mar. Biol. Ecol. 57:181-194.

Wood, J.M. 1974. Biological cycles for toxic elements in the environment. Science 183:1049-1052.

Wong, G.T. and C.S. Moy. 1984. Cesium-137, metals and organic carbon in the sediments of the James River estuary, Virginia. Est. Coast. Shelf Sci. 18:37-49.

Wong, P.T.S., Y.K. Chau, O. Kramer, and G.A. Bengert. 1982. Structure-toxicity relationship of tin compounds on algae. Can. J. Fish. Aquat. Sci. 39:483-488.

Wright, D.A. and G.D. Foster. 1987. Toxicant monitoring in the upper Chesapeake Bay, p. 55-62. In: M.P. Lynch and E.C. Krome (eds.), State of the Chesapeake Bay—second annual monitoring repor—Compendium. Chesapeake Research Consortium publ. 125.

Wright, D.A. and C.D. Zamuda. 1987. Copper accumulation by two bivalve molluscs: salinity effect is independent of cupric ion activity. Mar. Environ. Res. 23:1-14.

Young, D.R. 1984. Methods of evaluating pollutant biomagnification in marine ecosystems, p. 261-278. In: H.H. White (ed.), Concepts in marine pollution measurements. MD Sea Grant publ. UM-SG-TS-84-03.

Young, G.R., J.A. Swader, and S.W. Bingham. 1980. Kepone uptake by algae and effects on growth, photosynthesis, and respiration. Va. J. Sci. 31: 47-54.

Zamuda, C.D. 1984. The bioavailability of copper to the American oyster *Crassostrea virginica.* Dissertation, University of Maryland.

Zamuda, C.D. and W.G. Sunda. 1982. Bioavailability of dissolved copper to the American oyster *Crassostrea virginica.* I. Importance of chemical speciation. Mar. Biol. 66:77-82.

Zamuda, C.D., D.A. Wright, and R.A. Smucker. 1985. The importance of dissolved organic compounds in the accumulation of copper by the American oyster, *Crassostrea virginica.* Mar. Environ. Res. 16:1-12.

Zirino, A. and S. Yamamoto. 1972. A pH-dependent model for chemical speciation of copper, zinc, cadmium, and lead in seawater. Limnol. Oceanogr. 17:661-671.

Zoller, W.H., G.K. Barley, M.S. Germani, and D.S. Ballantine. 1983. Fate of mercury release from power plants. Final report to MD Power Plant Siting Program PPRP-79.

Zuckerman, J.J., R.P. Reisdorf, H.V. Ellis, and R.R. Wilkerson.1978. Organotins in biology and the environment, p. 388-422. In: F.E. Brinckman and J.M. Bellama (eds.), Organometals and organometalloids: occurrence and fate in the environment. ACS Symposium series 82.

Zuelke, R.W. and D.R. Kester. 1983. Copper speciation in marine waters, p. 773-788. In: C.S. Wong, E. Boyle, K.W. Bruland, J.D. Burton, and E.D. Goldberg (eds.), Trace metals in sea water. Plenum.

Bioavailability of Organic Pollutants to Aquatic Organisms

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INTRODUCTION

Settlement of the Chesapeake Bay region began in earnest in the early 17th century. The native American population and early colonists were impressed by the abundance of fish and shellfish and located their population centers to take advantage of these and other natural resources. Introduction of wastes into the bay was coincident with this settlement. As the human population increased, so did the pressure on the ecological system. In the 20th century significant quantities of synthetic chemicals began to be introduced, many of which were toxic and nonbiodegradable [Faust and Hunter 1971]. Today the areas surrounding the bay are experiencing unprecedented development; introduction of toxic organic pollutants has correspondingly increased. Water quality and the abundance of many aquatic organisms have suffered accordingly [O'Connor and Huggett 1988].

The bioavailability of organic pollutants is of recent concern and has not been as thoroughly studied as that of toxic metals. The initial belief was that "insoluble" organic xenobiotics were not available to aquatic organisms. They were assumed to be eliminated by irreversible binding to bottom sediments, which supposedly removed any significant threat to the ecosystem. More recently, we have come to realize that the term insoluble is a misnomer. All organic compounds possess some water solubility. In fact, it has been established that many low-solubility compounds are bioaccumulated or biomagnified to high concentrations in the tissues of organisms. Within several homologous series of organic compounds, toxicity has been negatively correlated with water solubility [Veith et al. 1983; Konemann 1981]. However, since chemically dissimilar compounds may exert their toxic effects via different mechanisms and influence different physiological functions, the solubility/toxicity relationship may not be directly applicable across series; the case of alcohols [Veith et al. 1983] versus organophosphorus pesticides [DeBruin 1976] is a good example. Sediments in many cases have been determined to be a source of toxic compounds, rather than merely a sink [Willford et al. 1987]. As a consequence, we have been forced to re-evaluate our understanding of the bioavailability of organic pollutants and their significance to the health of the Chesapeake Bay ecosystem.

PHYSICAL, CHEMICAL, AND BIOLOGICAL BASIS OF BIOAVAILABILITY

Water

Water is an obvious and important route for the exposure of aquatic organisms to organic pollutants. The water solubility of a compound has a profound influence on its environmental fate and bioavailability. Biota may come into contact with high concentrations of compounds that exhibit significant solubility, e.g., alcohols, phenols, and benzenes. Accidental spills and untreated effluents may result in toxic concentrations of these compounds in the water, causing obvious acute effects such as fish kills.

Some of the most toxic organic compounds possess low water solubilities. Classes of these compounds detected in the Chesapeake Bay include polynuclear aromatic hydrocarbons (PAHs), heterocyclic aromatic compounds (HACs), and halogenated pesticides. Table 1 gives several specific examples. These compounds may be taken up by biota directly from water, although dissolved concentrations will generally be very low.

The mechanism of accumulation is believed to be simple partitioning from water into lipid-rich biological tissues [Esser 1986]. Active biologically mediated transport mechanisms are thought to be less prevalent for organic pollutants than for trace metals; many of the metals have critical functions in enzyme systems. The hydrophobic nature of lipophilic xenobiotics provides

the driving force for the partitioning process. The presence of nonpolar organic solutes is not compatible within the polar water phase.

Since laboratory experiments designed to determine bioavailability directly using fish and invertebrates are expensive and time-consuming, surrogate tests have been proposed. For example, the bioaccumulation tendency of a lipophilic organic generally correlates with the n-octanol/water partition coefficient or K_{ow}. Octanol has been suggested as a substitute for biological lipids in these experiments, although some researchers have suggested that critical differences exist in the thermodynamics of partitioning between water, fish lipids, and octanol [Opperhuizen et al. 1988]. The classical approach for the determination of an octanol/water partition coefficient involves the addition of the test compound to a vessel containing mutually saturated octanol and water phases. The contents of the vessel are then thoroughly mixed and the system is allowed to come to equilibrium. The concentrations of the test compound in each of the two phases are subsequently determined and the coefficient calculated [Karickhoff and Brown 1979]. Difficulties (e.g., emulsions, detection limits, and contamination) are inherent in determinations for compounds possessing K_{ow} values greater than 100,000. Indirect measurement of the K_{ow} has also been suggested. For example, high-performance liquid chromatography (HPLC) has been used [Brooke et al. 1986]. Basically, a series of chemically similar compounds, for which the partition coefficients are known, are co-injected with the compound of interest onto a reverse-phase HPLC column. A correlation of retention time with K_{ow} is then determined and the partition coefficient for the compound of interest is calculated from this relationship. K_{ow} values are often expressed as logarithms, due to the magnitude of the values and the relationship of this parameter to bioconcentration factors (BCFs). Table 2 lists the log BCF and log K_{ow} of several organic compounds which have been detected in the environment. Note that the BCFs are less than the K_{ow} values.

Equations relating K_{ow} directly to bioconcentration factors (BCF) in various types of organisms and to water solubility have been reported [Isnard and Lambert 1988; Esser 1986]. It has been reported that concentrations of lipophilic pollutants will be similar in aquatic organisms in general, provided exposure has been equal, equilibrium has been established, and the relative lipid contents of the biota have been normalized [Adams 1988]. Obviously, differences in the biotransformation capabilities of the organisms may alter this relationship.

K_{ow} determinations provide no information concerning biotransformation or biological effects. These phenomena are often significant. For example, English sole exposed to benzo(a)pyrene and Aroclor 1254 exhibited progressive accumulation of PCBs, but little accumulation of the PAH. This difference was attributed to extensive metabolism of benzo(a)pyrene by the fish [Malins et al. 1987]. Information on metabolism is very important in assessing the fate and effects of chemicals, especially since biotransformation may result in the production of more toxic or mutagenic products [Buhler and Williams 1988]. Bruggeman et al. [1984] observed that guppies bioaccumulated hexachlorobiphenyl, but not hexabromobenzene during aqueous exposures. They attributed this observation to the existence of an upper molecular size threshold, limiting transport across membranes. The importance of molecular volume has also been suggested by other workers [Doucette and Andren 1987]. Thus the use of octanol/water partition coefficients alone may result in

Table 1. Water solubility [May et al. 1978] and LC50 [Trucco et al. 1983] for some common environmental contaminants. Values for LC50 encompass a variety of organisms and conditions and are used for illustrative purposes only.

Compound	Solubility (mg/l)	LC50 (mg/l)
Naphthalene	31.7	1.00
Ronnel	1.08*	0.49†
Phenanthrene	1.00	0.10
Benz(a)anthracene	0.009	0.01
Arochlor 1254	0.012+	0.003+
p,p'-DDT	0.003*	0.0002†

* Data from Chiou et al. (1977).
† Data from Johnson et al. (1980).
\+ Data from National Research Council (1979).

Table 2. The log BCF and log K_{ow} of several common environmental contaminants are given. Values are from Veith et al. [1979], except as noted.

Compound	log BCF	log K_{ow}
Naphthalene	2.63	3.37*
Pentachlorophenol	2.89	5.01
Phenanthrene	3.42	4.46
p,p'-DDT	4.47	5.75
Chlordane	4.58	6.00
Aroclor 1254	5.00	6.47

*Chiou et al. (1977).

an overestimation of the BCF in the case of extremely large molecules.

The major avenue of entry into biota for lipophilic compounds varies depending on the organism's size, morphology, and ecology [Knezovich and Harrison 1987]. For example, sorption to the general cell surface may predominate for single-cell biota, such as algae or heterotrophic bacteria. In higher organisms the body surface may contribute, but specialized respiratory structures (e.g., gills in fish) often are more important. Because of the large amounts of water processed by gills and the high lipid/water partition values (often >1,000,000), significant amounts of xenobiotics may rapidly bioaccumulate. Physiological changes may alter the bioavailability and thus the toxicity of organics to organisms. Conklin and Rao [1978] exposed molting adult grass shrimp to pentachlorophenol during various phases of the ecdysial cycle. They observed greater sensitivity and enhanced uptake immediately after ecdysis and attributed these effects to increased permeability of the cuticle. Newly molted blue crabs were observed to contain higher burdens of radiolabeled benzo(a)pyrene-derived material, compared with nonmolting crabs [Hale 1988] after laboratory exposure of these organisms.

Sediments

As previously mentioned, many organic compounds are rapidly sorbed to particulate matter. These compounds may adsorb to the mineral surface directly or to organic constituents of particles. The latter site is thought to be the most important [Hodson and Williams 1988]. Means et al. [1980], in determinations of batch equilibrium sorption isotherms, reported that adsorption of PAHs and HACs onto suspended soil/sediment was independent of substrate pH, cation exchange capacity, textural composition, or clay mineralogy. This statement is probably an oversimplification for direct application to waters of the Chesapeake Bay, because of the effects of factors such as pH on dissolved organics that may sorb lipophilic pollutants. These parameters are quite important in the behavior of polar organics and heavy metals. Grain size has been mentioned as a significant factor by some researchers [Marcus et al. 1988]. However, sediment grain size and organic content are generally correlated in the aquatic environment. The relationship of pollutant adsorption to organic content of sediments is well documented [Karickhoff and Brown 1979]. Equations relating sediment adsorption have been reported and generally employ coefficients such as K_{ow} [Dzombak and Luthy 1984].

The exact physical/chemical mechanism of sorption of chemicals to particulates is still uncertain. It has been postulated that the reduction of the water-organic interfacial area achieved by sorption is critical [Mackay and Powers 1987]. The mechanisms of uptake into biological lipids and the organic constituents of particulates appear similar. In comparison, parameters such as hydrogen bonding and gross electrostatic attractions contribute little to the sorption of nonpolar organics to sediments [Voice and Weber 1983].

Sorption of organic pollutants reduces but does not eliminate availability to biota. Malins et al. [1987] reported significant correlations of the occurrence of liver disease in bottom-dwelling fish with the level of sediment PAH contamination. Huggett et al. [1987] found a variety of abnormalities in fish of the Elizabeth River, Virginia, compared with specimens from less polluted rivers. The Elizabeth is heavily contaminated with PAHs. Hargis et al. [1984] reported acute toxicity and lesion formation in fish held over Elizabeth River sediments in the laboratory. Fish exposed to water alone that had passed over these contaminated sediments also showed signs of chemical stress. This result indicated that the xenobiotics were quite bioavailable.

Duration of contact between lipophilic compounds and sediments has been observed to affect their rate of desorption and bioavailability [Varanasi et al. 1985; Voice and Weber 1983]. Similar observations in regard to the elimination of lipophilic residues from aquatic biota, i.e., biphasic depuration patterns, have been reported [Spacie and Hamelink 1982]. Haddock et al. [1983] reported that extraction efficiency of PAHs, using organic solvents, was a function of PAH/sediment contact time. Coal is an extreme illustration of the bioavailability question. It contains a large variety and quantity of PAHs. However, these compounds were found to be biologically unavailable to oysters during laboratory exposures to suspended coal dust [Bender et al. 1987]. Indeed, coal and activated charcoal are used in water treatment for the removal of dissolved organics. An analogous example may be the low availability of metals incorporated in the crystalline lattice structure of mineral grains. The transfer of lipophilic xenobiotics from organic reservoirs of sediments to those in biota does not present the favorable thermodynamics observed for partitioning from water to sediments or biota. Indeed, transfer from sediments to biota may entail movement into the polar water phase as an intermediate step. The octanol/water partition coefficient will describe the tendency for this movement to occur, desorption obviously being less favorable than sorption. Wood et al. [1987] observed that lower chlorinated congeners were preferentially desorbed from PCB-contaminated sediments. They also observed that dipteran larvae present in these sediments bioaccumulated PCB congeners containing two to four chlorines in preference to higher chlorinated congeners.

More typically, the bioconcentration of PCBs has been highest for congeners possessing five to seven chlorines, in accordance with their K_{ow}.

Evidence that accumulation and toxicity of organic xenobiotics are related to the positions of the substituents is also available [Kuehl et al. 1987]. Sediments collected from sites in the Hampton Roads area of Virginia were analyzed by capillary gas chromatography and a halogen-selective Hall detector [Hale, in preparation]. Significant concentrations of both PCBs and highly chlorinated polychlorinated terphenyls (PCTs) were identified at one location. Clams collected at this same site exhibited only PCBs, but the accumulation of lower-chlorinated PCT congeners observed in bivalves at another site indicated that PCTs were bioavailable.

Exposure of biota to particulate-associated xenobiotics may occur via direct epidermal contact with sediment particles. Contact with interstitial water, which contains higher concentrations of pollutants than the overlying waters, may be even more important [Knezovich and Harrison 1987]. Suspension feeders such as oysters may filter contaminated particles from the water, and other organisms (e.g., benthic worms) may ingest bulk sediment [Reynoldson 1987]. Again, bioavailability of this adsorbed material is lower than that of xenobiotics dissolved in water.

Resuspension of contaminated sediments via dredging operations has been observed to increase the bioavailability of Kepone in the James River [Lunsford et al. 1987]. Severe hydrographic events, e.g. floods and hurricanes, may resuspend sediments [Wood et al. 1987]. Bioturbation has also been reported to increase bioavailability of organic toxics. Reynoldson [1987] reviewed the results of several studies of bioturbation. He reported that tubificid worms transported 90% of hexachlorobenzene, pentachlorobenzene, and trifluralin to the sediment surface from a uniformly mixed sediment in laboratory experiments over a 50-day period. This resulted in a four- to six-fold increase in contaminants in the overlying water, compared with unperturbed systems. This work was done by Karickhoff and Morris.

Estuarine circulation and chemistry may result in the formation of turbidity maxima. This phenomenon is observed when suspended particulates flocculate upon contact with saline water and settle. The particles are then transported up-estuary by tidal currents to less saline water, where they may dissociate and again be carried downstream. This cycling leads to the development of a turbid zone with a high sedimentation rate. Lipophilic pollutants sorb to particulates and may concentrate in these zones, as was observed for Kepone in the James River [Huggett et al. 1980].

Surface Microlayer

Organisms on the surface of the water may be exposed to elevated concentrations of xenobiotics via the surface microlayer. Lipophilic organics accumulate there because of solubility, surface tension, and specific gravity considerations. This accumulation may also be critical to eggs and larvae of species that frequent this layer [Hardy et al. 1987]. The surface microlayer will be discussed further during this conference.

Suspended and Dissolved Organic Matter

Naturally occurring organic matter, e.g., humic acids, may sorb lipophilic xenobiotics. McCarthy [1983] reported that dissolved and colloidal organic matter reduced the bioavailability of PAHs to daphnia in laboratory experiments. The effect may be physical or chemical. The molecular size of the complex may limit bio-uptake, as mentioned previously. The chemical nature, e.g., the polarity, may also affect the bioaccumulation potential of the bound pollutant.

Carter and Suffet [1982] reported that DDT may sorb to dissolved humic materials and thus remain in the water column for extended periods of time. Parameters such as pH may affect the residence time of naturally occurring organic matter and, in turn, affect that of associated pollutants. Sorption of lipophilic xenobiotics to organics and particulates has been reported to have varying effects on persistence [Leslie et al. 1987]. Important routes of degradation include photochemical and microbial pathways.

Food

Organic pollutants may be available to biota via consumption of contaminated prey items or plant material. Ingestion of contaminated particulates was discussed previously. Biomagnification has been identified as a significant factor for the transmission of some persistent xenobiotics. For example, food has been observed to serve as the major vehicle for uptake of Kepone in blue crabs. Exposure of these crustaceans to Kepone via water, during laboratory experiments, resulted in minimal body burdens. However, when contaminated food (in the form of either James River or laboratory-exposed oysters) was provided, significant concentrations of Kepone were accumulated [Schimmel et al. 1979]. Obviously, differing feeding strategies will put certain organisms at greater risk, both because of the items selected, and because of the efficiency of uptake. The efficiency of transfer of lipophilic xenobiotics from the tissues of prey to consumers will be, as from the organic reservoirs of particulates, much less than that from water. Transfer of pollutants to humans via consumption of contaminated seafood is also of considerable concern.

SUMMARY OF BIOAVAILABILITY AND ASSESSMENT TECHNIQUES

A variety of techniques have been utilized to assess bioavailability. Laboratory procedures are appealing since variables may be more easily controlled. Bioaccumulation studies, using tissue burdens of parent compounds, suffer since biota are often able to metabolize xenobiotics which normally go undetected by conventional procedures. Radiotracers have the advantage that total xenobiotic material present in the organism may be determined; however, they are expensive, have limited availability, and require specialized handling. Bioaccumulation and toxicity studies often use environmentally unrealistic exposure scenarios, e.g., solvent carriers and filtered seawater, to obtain "reasonable results." Application of organic pollutants to sediments and immediate use in experiments ignores the effects of contact time on pollutant/sediment binding. The shortcomings of octanol/water determinations have been discussed. Adsorption isotherm determinations provide valuable information, although the observation that organic components of particulates are the major constituent into which lipophilic pollutants partition must be considered. Nonetheless, these techniques provide information critical to our understanding of the general processes controlling bioavailability.

Organic pollutants are readily accumulated by aquatic organisms. Many compounds exhibit appreciable toxicity. Of particular concern are compounds of low solubility. These compounds may be bioaccumulated or biomagnified to high concentrations. They sorb to particulates, which may be ingested or settle to the bottom. Sediment-associated contaminants exhibit reduced bioavailability, but detrimental effects have been documented. Little information regarding the properties of the sediment/water interface is available, considering its importance with respect to pollutant fate and bioavailability. Differing feeding strategies and habitat selection may result in toxics being more available to some organisms than to others. Because the Bay is shallow, bioturbation, weather events, construction, boat traffic, and dredging may render sediment-adsorbed organics more available via re-working or resuspension.

The Chesapeake Bay represents a particularly complex theater for the investigation of the bioavailability question. Salinity regimes range from fresh to saline. These conditions may have measurable effects on the residence time of organic species and the physiology of aquatic organisms. Content of suspended particulates and content of colloidal and dissolved organics also differ drastically within the Bay. Tidal and riverine flows affect the disposition of toxics in both the water column and sediments. Turbidity, dissolved oxygen content, and temperature all influence the degradation rates of xenobiotics. These complexities will manifest themselves when laboratory data concerning bioavailability are applied to actual field situations.

REFERENCES

Adams, W. J. Bioavailability of neutral lipophilic organic chemicals contained on sediments: a review. In: Dickson, K. L.; Maki, A. W.; Brungs, W., eds. Fate and effects of sediment bound chemicals in aquatic systems. New York: Pergamon Press, 1988: p. 219-244.

Bender, M. E.; Roberts, M. H., Jr.; deFur, P. O. Unavailability of polynuclear aromatic hydrocarbons from coal particles to the Eastern oyster. Environ. Poll. 44:243-260; 1987.

Brooke, D. N.; Dobbs, A. J.; Williams, N. Octanol:water partition coefficients (P): measurement, estimation, and interpretation, particularly for chemicals with $P > 10^5$. Ecotox. Environ. Safety 11:251-260; 1986.

Bruggeman, W. A; Opperhuizen, A.; Wijbenga, A.; Hutzinger, O. Bioaccumulation of super-lipophilic chemicals in fish. Toxicol. Environ. Chem. 7:173-189; 1984.

Buhler, D. R.; Williams, D. E. The role of biotransformation in the toxicity of chemicals. Aquat. Toxicol. 11:19-28; 1988.

Carter, C. C.; Suffet, I. H. Binding of DDT to dissolved humic materials. Environ. Sci. Technol. 16:735-740; 1982.

Chiou, C. T.; Freed, V. H.; Schmedding, D. W.; Kohnert, R. L. Partition coefficient and bioaccumulation of selected organic compounds. Environ. Sci. Technol. 11:475-478; 1977.

Conklin, P. J.; Rao, K. R. Toxicity of sodium pentachlorophenate to the grass shrimp, *Palaemonetes pugio*, in relation to the molt cycle. In: Rao, K. R., ed. Pentachlorophenol: chemistry, pharmacology, and environmental toxicology. New York: Plenum Press, 1978:181-192.

DeBruin, A. Biochemical toxicology of environmental agents. Elsevier, Amsterdam; 1976.

Doucette, W. J.; Andren, A. W. Correlation of octanol/water partition coefficients and total molecular surface area for highly hydrophobic aromatic compounds. Environ. Sci. Technol. 21:821-824; 1987.

Dzombak, D. A.; Luthy, R. G. Estimating adsorption of polycyclic aromatic hydrocarbons on soils. Soil Sci. 137:292-307; 1984.

Esser, H. O. A review of the correlation between physico-chemical properties and bioaccumulation. Pestic. Sci. 17:265-276; 1986.

Faust, S. J.; Hunter, J. V., editors. Organic compounds in the aquatic environment. New York: Marcel Dekker, Inc.; 1971.

Haddock, J. D.; Landrum, P. F.; Glesy, J. P. Extraction efficiency of anthracene from sediments. Anal. Chem. 55:1197-1200; 1983.

Hale, R. C. Dynamics of organic pollutants in blue crabs, *Callinectes sapidus*, collected from the lower Chesapeake Bay region. In Lynch, M.P., and Krome, E.C., eds. Understanding the Estuary: Advances in Chesapeake Bay Research. Proceedings of a Conference, 29-31 March 1988, Baltimore, MD. CRC publication no. 129; Chesapeake Bay Program publication no. 24-88/

Hardy, J. T; Crecelius, B. A.; Antrim, L. D.; Kiesser, S. L.; Broadhurst, V. L.; Boehm, P. D.; Steinhauer, W. G. Aquatic surface microlayer contamination in Chesapeake Bay. 1987; 43 p. Maryland Power Plant Research Program; PPRP-100.

Hargis, W. J.; Roberts, M. H., Jr.; Zwerner, D. E. Effects of contaminated sediments and sediment-exposed effluent water on an estuarine fish: acute toxicity. Mar. Environ. Res. 14:337-354; 1984.

Hodson, J.; Williams, N. A. The estimation of the adsorption coefficient (K_{oc}) for soils by high performance liquid chromatography. Chemosphere 17:67-77; 1988.

Huggett, R. J.; Bender, M. E.; Unger, M. A. Polynuclear aromatic hydrocarbons in the Elizabeth River, Virginia. In: Dickson, K. L.; Maki, A. W.; Brungs, W., eds. Fate and effects of sediment bound chemicals in aquatic systems. New York: Pergamon Press, 1987: p. 327-341.

Huggett, R. J.; Nichols, M. M.; Bender, M. E. Kepone contamination of the James River Estuary. In: Baker, R. A., ed. Contaminants and sediments, Volume 1: Fate and transport, case studies, modeling, toxicity. Ann Arbor: Ann Arbor Science; 1980: p. 33-52.

Isnard, P.; Lambert, S. Estimating bioconcentration factors from octanol-water partition coefficient and aqueous solubility. Chemosphere 17:21-34; 1988.

Johnson, W. W.; Finley, M. T. Handbook of acute toxicity of chemicals to fish and aquatic invertebrates. 1980; 98 p. Resource Publication 137-U.S. Fish and Wildlife Service. Available from U.S. Government Printing Office.

Karickhoff, S. W.; Brown, D. S. Determination of octanol/water distribution coefficients, water solubilities, and sediment/water partition coefficients for hydrophobic organic pollutants. Environmental Research Laboratory-Athens, Ga. 1979; 25 p. Available from NTIS; PB80-103591.

Knezovich, J. P; Harrison, F. L. The bioavailability of sediment-sorbed organic chemicals: a review. Water Air Soil Poll. 32:233-245; 1987.

Konemann, H. Quantitative structure-activity relationships in fish toxicity studies. Part 1: Relationship for 50 industrial pollutants. Toxicol. 19:209-221; 1981.

Kuehl, D. W.; Cook, P. M.; Batterman, A. R.; Butterworth, B. C. Isomer dependent bioavailability of polychlorinated dibenzo-p-dioxins and dibenzofurans from municipal incinerator fly ash to carp. Chemosphere 16:657-666; 1987.

Leslie, T. J.; Dickson, K. L.; Jordan, J. A.; Hopkins, D. W. Effects of suspended solids on the water column biotransformation of anthracene. Arch. Environ. Contam. Toxicol. 16:637-642; 1987.

Lunsford, C. A.; Weinstein, M. P.; Scott, L. Uptake of Kepone by the estuarine bivalve *Rangia cuneata*, during dredging of contaminated sediments in the James River. Water Res. 21:411-416; 1987.

Mackay, D.; Powers, B. Sorption of hydrophobic chemicals from water: a hypothesis for the mechanism of the particle concentration effect. Chemosphere 16:745-758; 1987.

Malins, D. C.; McCain, B. B.; Brown, D. W.; Varanasi, U.; Krahn, M. M.; Myers, M. S.; Chan, S. Sediment-associated contaminants and liver diseases in bottom-dwelling fish. Hydrobiol. 149:67-74; 1987.

Marcus, J. M.; Swearingen, G. R.; Williams, A. D.; Heizer, D. D. Polynuclear aromatic hydrocarbon and heavy metal concentrations in sediments at coastal South Carolina marinas. Environ. Contam. Toxicol. 17:103-113; 1988.

May, W. E.; Wasik, S. P.; Freeman, D. H. Determination of the solubility behavior of some polynuclear aromatic hydrocarbons in water. Anal. Chem. 50:997-1000; 1978.

McCarthy, J. F. Role of particulate organic matter in decreasing accumulation of polynuclear aromatic hydrocarbons by *Daphnia magna*. Arch. Environ. Contam. Toxicol. 12:559-568; 1983.

Means, J. C.; Wood, S. G.; Hassett, J. J.; Banwart, W. L. Sorption of polynuclear aromatic hydrocarbons by sediments and soils. Environ. Sci. Technol. 14:1524-1528; 1980.

National Research Council. Polychlorinated Biphenyls. 1979; 182 p. Available from the National Academy of Sciences.

O'Connor, J. M.; Huggett, R. J. Aquatic pollution problems, North Atlantic coast, including Chesapeake Bay. Aquat. Toxicol. 11:163-190; 1988.

Opperhuizen, A.; Serne, P.; Van der Steen, J. M. D. Thermodynamics of fish/water and octanol/water partitioning of some chlorinated benzenes. Environ. Sci. Technol. 22:286-291; 1988.

Reynoldson, T. B. Interactions between sediment contaminants and benthic organisms. Hydrobiol. 149:53-66; 1987.

Schimmel, S. C.; Patrick, J. M., Jr.; Faas, L. F.; Oglesby, J. L.; Wilson, A. J., Jr. Kepone: toxicity and bioaccumulation in blue crabs. Estuaries 2:9-15; 1979.

Spacie, A.; Hamelink, J. L. Alternative models for describing the bioconcentration of organics in fish. Environ. Toxicol.

Chem. 1:309-320; 1982.

Trucco, R. G.; Engelhardt, F. R.; Stacey, B. Toxicity, accumulation and clearance of aromatic hydrocarbons in *Daphnia pulex*. Environ. Poll. Ser. A, 31:191-203; 1983.

Varanasi, U.; Reichert, W. L.; Stein, J. E.; Brown, D. W.; Sanborn, H. R. Bioavailability and biotransformation of aromatic hydrocarbons in benthic organisms exposed to sediment from an urban estuary. Environ. Sci. Technol. 19:836-841; 1985.

Veith, G. D.; Call, D. J.; Brooke, L. T. Structure-toxicity relationships for the fathead minnow, *Pimephales promelas:* Narcotic industrial chemicals. Can. J. Fish. Aquat. Sci. 40:743-748; 1983.

Veith, G. D.; DeFoe, D. L.; Bergstedt, B. V. Measuring and estimating the bioconcentration factor of chemicals in fish. J. Fish. Res. Board. Can. 36:1040-1048; 1979.

Voice, T. C.; Weber, W. J., Jr. Sorption of hydrophobic compounds by sediments, soils and suspended solids. Water Res. 17:1433-1441; 1983.

Willford, W. A.; Mac, M. J.; Hesselberg, R. J. Assessing the bioaccumulation of contaminants from sediments by fish and other aquatic organisms. Hydrobiol. 149:107-111; 1987.

Wood, L. W.; Rhee, G.; Bush, B.; Bernard, E. Sediment desorption of PCB congeners and their bio-uptake by dipteran larvae. Water Res. 21:875-884; 1987.

Introduction to Plenary Session

In writing the genetics chapter for *Perspectives*, we were not able to cover all the areas of interest, and the speakers for this session were chosen in the hope that they might fill some of the gaps left by the chapter.

One particular point that was mentioned in *Perspectives* was that female striped bass seemed to be homing extensively in the Bay, whereas the males appeared to wander. This apparent divergence of behavior awaited confirmation after analysis of the 1987 data. The 1987 data have been analyzed and they do support the apparent divergence.

—*R.C.*

New Methodology for Immunologic Discrimination of Stocks or Populations in Fish Species

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The immunogenetic section of the National Fish Health Research Laboratory has worked with striped bass immunology for three years. Preliminary work shows significant differences between populations, but to be useful the work must reach the point where an individual fish of unknown origin can be assigned to the population of its origin with reasonable certainty.

A variety of approaches can be categorized as immunological, including the use of some natural agglutinins found in the serum of pigs. These agglutinate red blood cells without the addition of any antibody, to a different extent in different individuals and different populations. In some Pacific salmon species, this approach has indicated a divergence between Asian and North American stocks. Approaches using antiserum to purified red blood cells of Pacific salmon species have shown variable agglutination strengths in large assemblages from widely separated geographic areas. Yet another approach has been the induction of antibody production in rabbits and other animals using the serum of various salmon species, with double diffusion in agarose and an analysis of precipitin bands present and missing.

These three techniques do not lend themselves to quantitation and are extremely insensitive in comparison to more recently developed techniques, primarily radioimmune assay in combination with a competitive binding assay. This approach, in use since 1980, focuses not on the division of breeding groups within a species, but rather on a taxonomic assessment of the identity of the species. Serum proteins are used, as these secreted proteins display many more readily-acceptable amino acid substitutions than do more functional proteins such as enzymes.

The competitive binding assay (Figure 1) has two steps. In the first step, the antibody that has been prepared against a purified protein is incubated with an unknown, unpurified serum that contains that antigenic protein. If the intact serum is identical to the source from which the antibodies were prepared, then all the antibody will be bound; whereas if there is lesser identity, some of the antibody against which the antiserum was prepared will be unbound and available for subsequent reactions.

In the second step a competition mixture of antigen, antibody, and antigen-antibody complex is poured into a receptacle coated with the antigen used to prepare the antibody. If free antibody remains that has not been consumed by the antigen excess, it is available to react with the antigen that has been immobilized on the receptacle. That amount can be quantitated by using a second antibody, labeled with either a radioactive

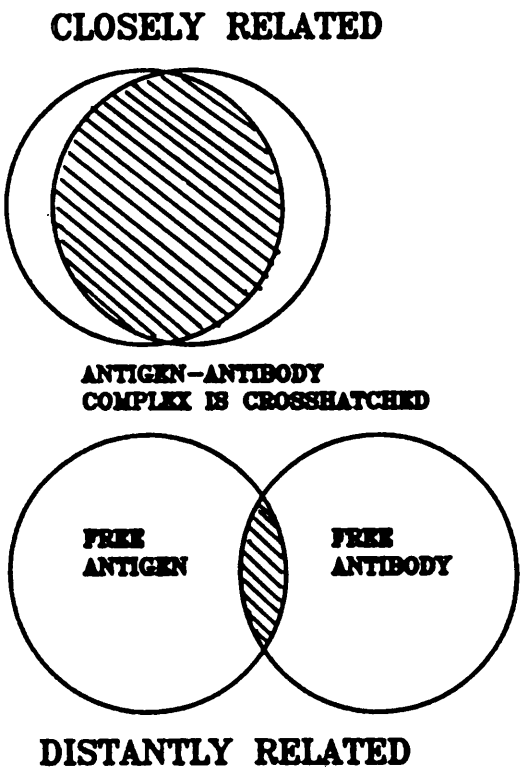

Figure 1. Diagram of amount of antibody bound with different degrees of similarity to reference antigen, where much antibody remains unbound when the relationship is remote and more is consumed with relationships of increasing similarity.

isotope or an enzyme for which specific substrate and cofactors can be introduced, and the resulting color development (corresponding to the amount of protein bound) can be measured spectrophotometrically.

Although this approach works well there are complications. The first level of complication is that two populations may produce individuals with identical absorbance readings. The situation is analogous to examining a polyacrylamide gel from the end rather than from the side — three bands at entirely different locations may still produce the same absorbance. The advantage of this system, however, is that the antigen-antibody complex can be removed with the use of affinity columns (provided that the antigen is not identical to the antigen against which the antibody was prepared), so that the effluent contains only unique antigen recognition sites. That portion of the protein now is unique to the population with regard to another population. In examining larger numbers of populations, a battery of immune reagents is necessary. It is also necessary to be careful about defining reference standards, since a sample of any geographic population is likely also to contain individuals that originated elsewhere. The finding that males are much less likely than females to return to their place of origin should caution striped bass researchers to segregate their data on the basis of gender.

RELEVANT LITERATURE

Chapman, R.W. 1987. Changes in the population structure of male striped bass, *Morone saxatilis*, spawning in the three areas of Chesapeake Bay from 1984 to 1986. U. S. Fish and Wildlife Service, Fish. Bull. 85(1):167-170.

Cushing, J.E. 1956. Observations on serology of tuna. U.S. Fish and Wildlife Services, Special Scientific Report, Fisheries (183). 14 pp.

Fabrizio, M.; Saila, S.B. 1986. Identification of striped bass stocks in coastal waters from New Jersey to Massachusetts: 1983 to 1985. Rhode Island Division of Fish and Wildlife, Annual Progress Report, Project AFC 4-4, Wakefield, RI.

Fujino, K. 1970. Immunological and biochemical genetics of tunas. Trans. Am. Fish. Soc. 99:152-178.

Lowenstein, J.M.; Molleson, T.; Washburn, S.I. 1982. Piltdown jaw confirmed as orang. Nature 299:294.

Lowenstein, J.M.; Ryder, O.A. 1985. Immunological systematics of the extinct quagga (Equidae). Experientia 41:1192-1193.

Rago, P.J.; Richards, R.A. 1987. Emergency striped bass research study report for 1986. U.S. Dept. of the Interior & U.S. Dept. of Commerce (unnumbered document). 79 pp.

Ridgway, G.J.; Klontz, G.W. 1961. Blood types in Pacific salmon. Int. North Pacific Fish. Comm., Bull. (5):49-55.

Ridgway, G.J., Cushing, J.E., Durall, G.L. 1961. Serological differentiation of populations of sockeye salmon, *Oncorhynchus nerka*. Int. North Pacific Fish. Comm., Bull. (3):5-10.

Ridgway, G.J., Klontz, G.W., Matsumoto, C. 1962. Intraspecific differences in serum antigens of red salmon demonstrated by immunological methods. Int. North Pacific Fish. Comm., Bull (8):1-13.

Mitochondrial DNA Analysis of Chesapeake and Delaware Bay Populations of *Fundulus heteroclitus*

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The molecular approach to genetics has opened doors to questions in population and evolutionary biology that were previously unapproachable by conventional methods. The analysis of proteins has revealed extensive genetic variation in natural populations. The molecular approach is allowing us to address questions concerning population structure, gene flow, and evolution. Dr. Powers' laboratory at Johns Hopkins makes use of the mummichog *(Fundulus heteroclitus)* as a model marine organism to address these issues. *F. heteroclitus* is found all along the Atlantic coast from Nova Scotia to Georgia, and of course in the Chesapeake Bay. In sampling along the coast and determining the gene frequencies for lactate dehydrogenase B (Ldh-B), it has been shown that one allele is fixed in Florida and another in Maine. Similar clines have been observed for many other loci.

Two models have been proposed to explain these clines. One hypothesis is that the populations have always been in contact with each other, and the clines are the product of chance and adaptive forces, selection, and nonrandom migration. Alternatively, the clines could result from historical isolation of the two populations with subsequent removal of the isolating barrier and resumption of contact. (This is known as secondary intergradation.) Recent work with mitochondrial DNA by Gonzalez and Powers has allowed us to eliminate the first possibility. This research has shown that the Maine type fish and the Georgia type fish are very different, having been separated sometime during the last three to six million years.

PROPERTIES OF MITOCHONDRIAL DNA

Mitochondrial DNA (mtDNA) is a small circular DNA found in the mitochondria of all eukaryotes. In *F. heteroclitus* it is about 17 kB in size. It is maternally inherited, allowing the tracing of maternal lineages. The molecule evolves rapidly and variation is sensitive to small population sizes.

The mechanics of the technique involve isolating the mitochondria by cell fractionation and subsequent extraction of the nucleic acids. Then a restriction enzyme that recognizes specific nucleotide sequences is used; it cleaves the mtDNA where those sequences appear, generating fragments that can be sized on an agarose gel. The *Fundulus* mtDNA shows two major patterns. For analysis a battery of restriction enzymes is used to generate a "fingerprint" of the mtDNA. The five restriction enzymes usually used to distinguish between northern and southern types of *Fundulus* in the Bay detect a minimum difference of six nucleotide changes. Gonzalez and Powers have shown that there are two main *F. heteroclitus* mtDNA patterns along the Atlantic coast. The current zone of secondary intergradation is found near northern New Jersey, with some mtDNA variation within the groups and substantial differences between the two groups.

REGIONAL HISTORY

Fifteen thousand years ago the Chesapeake Bay had not yet been formed. The coastline was near the continental shelf, and the glaciers extended to northern New Jersey. A relevant question is whether the zone of secondary intergradation ever existed further southward. If so, it would be reasonable to expect some zone of secondary integradation in the Delaware and/or Chesapeake Bays. To answer this question we examined 540 fish from 20 locations, typing them for

mtDNA and nuclear-encoded enzymes. Isolated refugia of *F. heteroclitus* with northern type mtDNA were found in five areas of the Bay. Eleven of the populations had only southern or northern mtDNA types, and the remaining nine populations contained both mtDNA types. Populations within a river are very different. In fact, populations in the upper James and upper Potomac resemble each other more closely than they do the populations in the lower portions of their respective rivers. The presence of the northern type mtDNA fish in these rivers and in the upper Bay imply that these fish once occupied the entire Bay and were probably present at the mouth of the Bay when it was initially forming.

Similar clines are also observed in the gene frequencies of northern and southern alleles of allozyme loci. For example, Ldh-B allele frequencies have similar clines but none of the populations are fixed for either allele.

The population in the upper Potomac, near Mount Vernon, is fixed for the northern mtDNA type but contains the southern Ldh-B alleles. This pattern could indicate that "southern" males are dispersing and breeding with more sedentary females. Conversely, populations near the mouth of the Bay are fixed for southern mtDNA and contain the northern Ldh-B alleles, presumably contributed by northern males dispersing and breeding. These patterns suggest that the males are heavily involved in gene flow in *F. heteroclitus*. Alternatively, selection or genetic drift could explain these patterns.

CONCLUSIONS

Both northern and southern races of *Fundulus* inhabit the Chesapeake and Delaware Bays. The northern race was found in refugia in the upper bays and in the upper James, Potomac, Big Choptank and Patuxent rivers; the southern race was found elsewhere in the bays. If more rivers had been sampled, it is likely that more refugia would have been found, as every river sampled in the upper tidal region had a refugia. These isolated refugia result in multiple zones of secondary intergradation. We conclude that the northern race fish on the Atlantic coast once lived as far south as the present mouth of the Chesapeake Bay.

These studies emphasize two points. One, the history of these populations is connected to previous glaciations. The northern race either was present or came into the Bay as the glacier retreated, and the southern race then followed. Thus the history of glaciation has had a strong influence on the biota present in the Bay and their genetics. Two, this work with *F. heteroclitus* and Chapman's data on striped bass have shown that mtDNA research can provide valuable insights into the living resources in the Bay and elsewhere.

QUESTIONS

Q: Could mtDNA techniques be used to determine how much interchange there is between, for instance, the populations in the lower James and the population in the lower Potomac?
A: Yes, with direct sequencing or enzymes that recognize more sites.
Q: Have you considered that the difference in mtDNA frequencies could be related to salinity differences?
A: It could be that the northern-type mtDNA populations are adapted to lower salinity, as their refugia are generally of lower salinity; but the northern type also lives in high-salinity water in Maine.
Q: *Fundulus* is usually thought of as a territorial species. Do you agree that these data suggest that the females are more territorial?
A: I think the data suggest that males could be dispersing more, but to say that females are more territorial is a further extrapolation from the data.
Q: Could a freshwater species of *Fundulus* be responsible for the northern type of mtDNA?
A: The mtDNA of *Fundulus diaphanus*, a sympatric species in upper tidal areas, has a very different pattern from that of *F. heteroclitus*. Granted, there are many species of *Fundulus*, and the mtDNA could have come from one of them, but I think that's unlikely since we find the same northern mtDNA pattern in Maine.
Q: For your scenario you must postulate that there is no larval transport downriver.
A: These fish are generally in very still water, in marshy areas, and for them to be transported downriver they would have to venture into water that for them would be very rough.
Q (followup): Has any data been collected on the salinity tolerances of eggs and larval stages?
A: There was some work done on salinity tolerances. Unfortunately the authors were not aware that they were dealing with two different races, so comparisons are difficult.
A: One of the things we don't know is how far south one can find the northern type of *F. heteroclitus* in a river. It is possible, for example, that the Cape Hatteras area, where there are a lot of currents, could be where the original isolating barrier was located.

Genetic Analysis of Oysters that Grow at Different Rates

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Two years ago I met a man who claimed he could grow oysters in less than a year. He grows them cultchless in a floating raft culture. They look different from native oysters, with a pointed umbo and a very deep valve on one side and a very flat valve on the other. These inbred oysters also have a much higher percentage meat than native oysters, because native animals have a much thicker shell. As you would expect, the inbred animals show little variation in allele frequencies. Differences in allele frequencies are evident in native vs. inbred oysters.

Because the animals are so different, we wanted to study them. The inbred oysters are grown in a cultchless manner that the grower maintains as a secret. We were able to use epinephrine to induce the cultchless metamorphosis of eyed larvae. We tested grow-out of these oysters in floating rafts. Tens of thousands of oysters can be grown in this way in a small space. We generated larvae from native animals and from some broodstock of the inbred strain and raised them in floating rafts in a tidal creek and compared the growth rates.

At 3.5 months the animals were all just short of 2 inches long. The inbred animals grew faster than the native ones. The growth rate for the inbred strain was at least 50% higher during the juvenile period and about 20% higher at the end of a four-month period. Both sets of oysters grew very well in the cultchless floating raft system; most of the animals would have reached market size in less than a year. It appears this approach would be well-suited to aquaculture.

Why are these animals different in their growth rates? We approached the question genetically and compared large and small representatives (1-2 SD below and above mean size) of both strains. Several alleles present in the native population were absent in the inbred population, presumably due to loss during inbreeding. Within groups, one large difference is found at Lap-2C. None of the very large animals contained a C allele. Otherwise the size groups within the strains were not very different.

Analysis of variance with respect to weight and length vs. strain and genotype showed that two genes appear to affect growth in the inbred strain, and in the native animals, four or five loci have an effect. So far only Lap-2 can be associated with a specific effect: animals with the C allele do not do as well as animals without it. We cannot say that growth rate is inhibited by the enzyme.

Genetic analysis of oysters with respect to growth rate and other measures of fitness may be a productive strategy in learning more about the basic biology of oysters; this kind of analysis may suggest useful management strategies for shellfish and perhaps other species in the Bay as well.

QUESTIONS

Q: Work has been done at VIMS on developing cultchless oysters by removing them from cultch once they have set.
A: Yes, this work is different in that the epinephrine induces what is thought to be a hormonal response, a willingness to metamorphose without substrate.
Q: Your work is very similar to work on Long Island with *Mytilis edulis*, which has shown that *Mytilis edulis* with a certain genotype is less efficient at using or conserving protein.
A: By chance or not, the same locus is involved there as here.
Q: Osmoregulation may also be associated with this allele; what effects have been seen with alteration of the salinity?
A: We have not yet tried altering the environment, but one of the beauties of this sytem is that the environment can be manipulated to a great degree.

Introduction to Plenary Session

The idea that waters and sediments of aquatic systems are linked by exchanges of energy and nutrients is not new. References to the concept can be found in the limnological literature in the late 1930s, and the foundations for the ideas were set down even earlier in the century. What has developed in the last 10 years is an appreciation of the importance of these exchanges in shallow marine ecosystems. Nixon, for example, has hypothesized that nitrogen availability and ultimately the productivity of these systems is regulated by the processing of planktonic organic matter in benthic communities. Kemp and Boynton have emphasized the importance of nutrient cycling at estuarine interfaces, and one of the important interfaces is between the sediments and the waters. Sanders in his presentation yesterday said that in addition to the exchanges of organic matter and nutrients, exchanges across the sediment-water interface can influence the bioavailability of toxics and other xenobiotic compounds.

The impetus, however, for the *Perspectives* article and the presentation today, came from those charged with modeling water quality in the Bay. They discovered that without feedback loops between the waters and the sediments, the Bay just didn't work right. Our purpose today is to continue trying to understand how these processes fit together and how the waters and sediment are coupled.

The talks that follow will be organized around a general conceptual model (Figure 1); they will focus on the mid-mesohaline portion of the Bay; and they will aim toward assembling quantitative budgets. The model is divided into three parts: the formation of organic matter, the processing of that matter in the water column and its sinking; and its fate on the bottom.

—*J.G.*

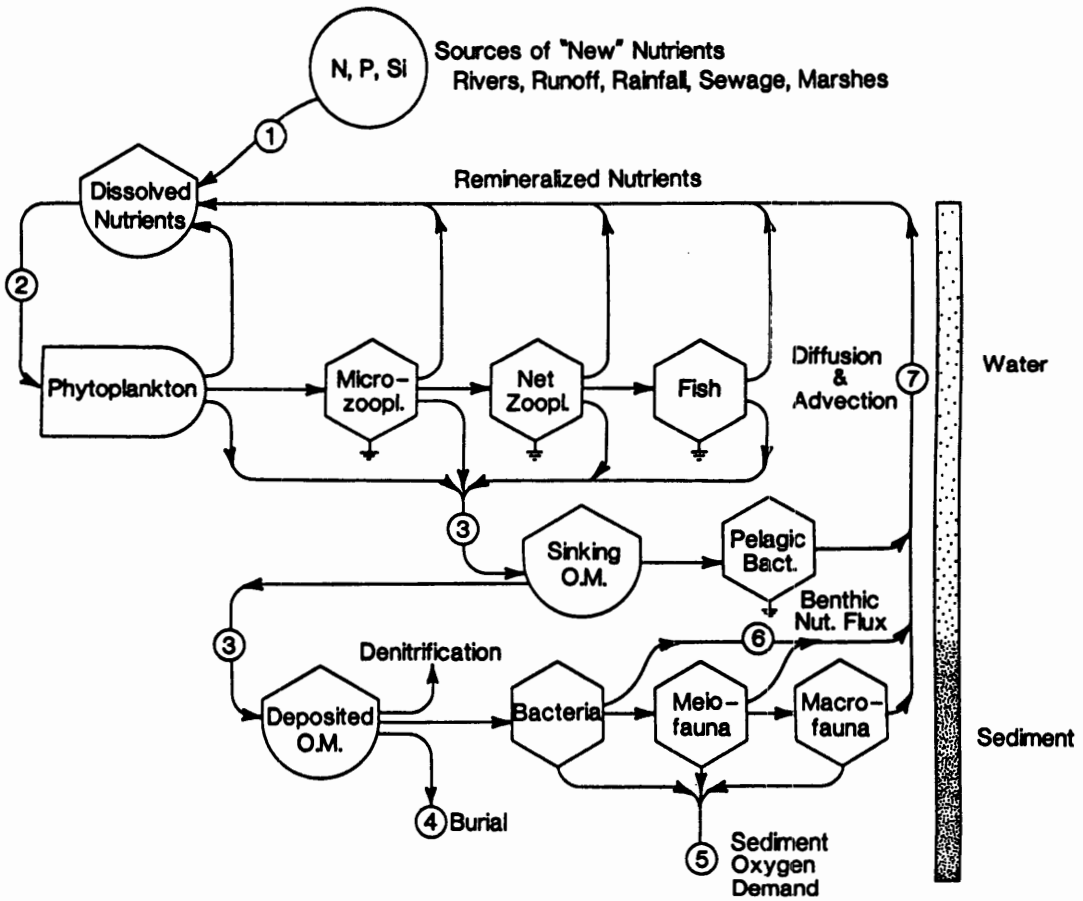

Figure 1. Conceptual model of nutrient element flows in plankton-based systems.

Production of Particulate Organic Matter in the Mesohaline Reach of the Chesapeake Bay

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One of the most important conclusions that emerged from early work on this problem (a conclusion that is now debatable) was that most of the input of organic material into the Bay downstream of the turbidity maximum is accounted for by phytoplankton. More recently, work by Ducklow, Peele, Tuttle, and others has shown that a large fraction of this production appears to be cycled through bacterioplankton. Thus we can view the production and fate of particulate organic material in the water column as a function of two interacting networks of flow: (1) a phytoplankton-based network, which accounts for most biomass yield and particulate organic matter flux to metazoan consumers; and (2) a bacterioplankton-based network (the "microbial loop"), which accounts for most internal flows (to the water column) and most of the flux of particulate organic matter to microbial consumers (within the water column). In the context of benthic-pelagic coupling, it is likely that the phytoplankton network dominates the flux of organic matter to the benthos while bacterioplankton are more directly involved in the rapid turnover of carbon and associated nutrients in the water column.

This presentation addresses the problem of the response of phytoplankton to nutrient input. Hugh Ducklow will speak later today in a concurrent session on the bacterial component of the system. Four points important to the topic are:

- Distributions of phytoplankton productivity and biomass in both time and space;
- Environmental factors that are responsible for this variation;
- The fate of phytoplankton production as deduced from patterns in the distribution of phytoplankton;
- Nitrogen vs. phosphorus limitation of the system as a whole.

The area we have concentrated on is the mesohaline reach of the Bay between the Bay Bridge and the Patuxent River. Many properties vary longitudinally down the axis of the Bay from the freshwater areas down to the sea, and there are good reasons for focusing on the mesohaline reach. This is where most of the nutrient input delivered by the Susquehanna River is assimilated and where accumulations of phytoplankton biomass are highest.

Harding et al. [1986] recently found that integral phytoplankton production over the water column could be described as a function of chlorophyll concentration and the amount of light absorbed by phytoplankton in the euphotic zone. Thus the variation in integral production down the axis of the Bay can be parameterized in terms of biomass and light as a first approximation.

Lateral variability has been studied by a number of people, with the earliest probably being Flemer in the late 1960s. The findings then were similar to the 1984 findings: biomass (as indexed by chlorophyll α) was higher on the western shore. More recent studies [Malone et al. 1986] also show that the production per unit of chlorophyll is generally higher on the eastern shore. It is possible that on the western shore biomass is accumulating while on the eastern shore it is turning over rapidly, perhaps because of removal by benthic and pelagic consumers.

SEASONAL VARIABILITY

Annual cycles of phytoplankton biomass and productivity

Annual cycles of phytoplankton biomass and productivity in the mesohaline reach of the Bay are seasonally out of phase, with biomass peaking during spring and productivity peaking during summer (Figure 1). Large interannual variations in phytoplankton productivity also occur (Figure 1). Although there is evidence that phytoplankton productivity is phosphorus-limited during spring and nitrogen-limited during summer [D'Elia et al. 1986], annual phytoplankton production appears to be more sensitive to N than to P loading (Figure 2), and the Bay as a system removes the inorganic N input more efficiently than the P input [Fisher et al. in press] (Tables 1 and 2). The major external source of N to the mesohaline reach of the Bay is the

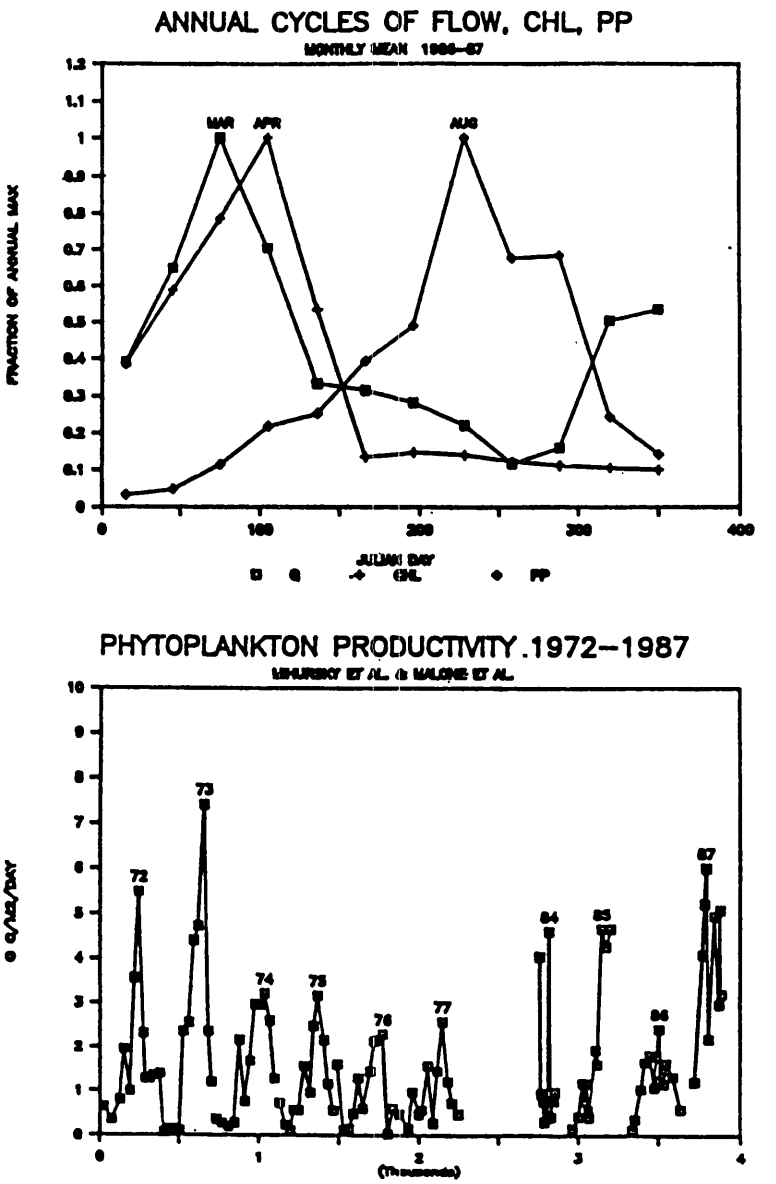

Figure 1. Upper panel: annual cycles of Susquehanna River discharge (□), chlorophyll a content of the water column (+), and phytoplankton productivity (◇) of the euphotic zone (monthly averages over the mesohaline reach of the Bay for 1986-1987 as a fraction of the maximum monthly means). Lower panel: interannual time series of phytoplankton productivity in the mesohaline reach of the Bay.

Susquehanna River, which supplies >90% of the inorganic N input and 70-80% of the total N input [Schubel and Pritchard 1986; Fisher et al. in press]. As is typical of mid-latitude rivers, the annual cycle of freshwater discharge exhibits a spring maximum and a summer minimum. Consequently, 50-60% of the annual N input occurs during the spring freshet which precedes the spring peak in biomass by about one month (Figure 1).

Comparison of variations in nitrate input from the Susquehanna and total phytoplankton biomass in the mesohaline reach of the Bay suggests that seasonal and interannual variations in phytoplankton biomass are related to the riverine input of N (Figure 2). Water column and euphotic zone levels of biomass (as nitrogen, g N) are significantly correlated with the supply of nitrate-N (Q-N) by the least square regressions:

(1) water column phyto-N = 4.6 + 8 (Q-N) (r = 0.93)

(2) euphotic zone phyto-N = 3.5 + 4 (Q-N) (r = 0.88)

As indicated by the slopes of these regressions, a unit increase in nitrate-N input results, on average, in a fourfold increase in phytoplankton biomass in the euphotic zone and an eightfold increase in the water column as a whole. While these calculations must be considered rough approximations, they are indicative of the effects of two-layered estuarine circulation and N recycling on the accumulation of phytoplankton biomass within the mesohaline reach of the Bay. The development of the spring biomass maximum appears to be due to sedimentation and accumulation of biomass and to the advection of biomass with bottom water from downstream.

The annual cycle of phytoplankton productivity is characterized by a winter minimum and a summer maximum (Figure 1). Consequently, nitrogen assimilation by phytoplankton is inversely related to N supply (Figure 2). This suggests the importance of light and temperature as parameters of phytoplankton productivity, as Harding et al. [1986] have shown. However, as reported by Boynton et al. [1982], the magnitude of the summer productivity maximum shows large interannual variations, which are related to variation in phytoplankton growth rate. These variations were correlated (r = 0.96) with vertical salinity stratification. This correlation and the results of nutrient enrichment studies indicating that phytoplankton growth is likely to be N-limited during summer [D'Elia et al. 1986] support the hypothesis that the magnitude of the summer productivity maximum is a function of physical processes that regulate the recycling of ammonium from below the pycnocline into the euphotic zone [Malone et al. 1986].

Thus the annual cycle of phytoplankton productivity in the mesohaline reach of the Bay is governed by light, temperatures, and nitrogen recycling. Seasonal and interannual variations in phytoplankton biomass occur in response to variations in freshwater flow and associated variations in nutrient flux. The occurrence of a summer productivity maximum and interannual variations in the magnitude of this maximum reflect the effects of vertical stratification on the return flux of ammonium from the benthos to the euphotic zone.

FATE OF PHYTOPLANKTON PRODUCTION

The observation that biomass accumulates during spring while growth rates are relatively low and that most of the accumulation occurs below the euphotic zone implies that the removal rate of phytoplankton is also low. Measurements of chlorophyll degradation products of primary consumers are consistent with this conclusion and suggest that the rapid decline of the spring bloom during late May and early June is a consequence of increased grazing pressure. Vertical distributions of chlorophyll α, low grazing pressure, and the dominance of diatoms also suggest that the vertical flux of phytoplankton biomass to the benthos should exhibit a seasonal maximum in the spring. The rapid turnover of phytoplankton biomass during summer probably reflects the combination of high phytoplankton growth rate and grazing pressure [White and Roman 1988], which suggests that vertical flux during this period is dominated by fecal material of phytoplankton.

Several things can be learned from looking at the variations in chlorophyll and the percentage of nitrogen in the dissolved organic nitrogen (DON) pool vs. the nitrate pool, down the axis of the Bay along the salinity

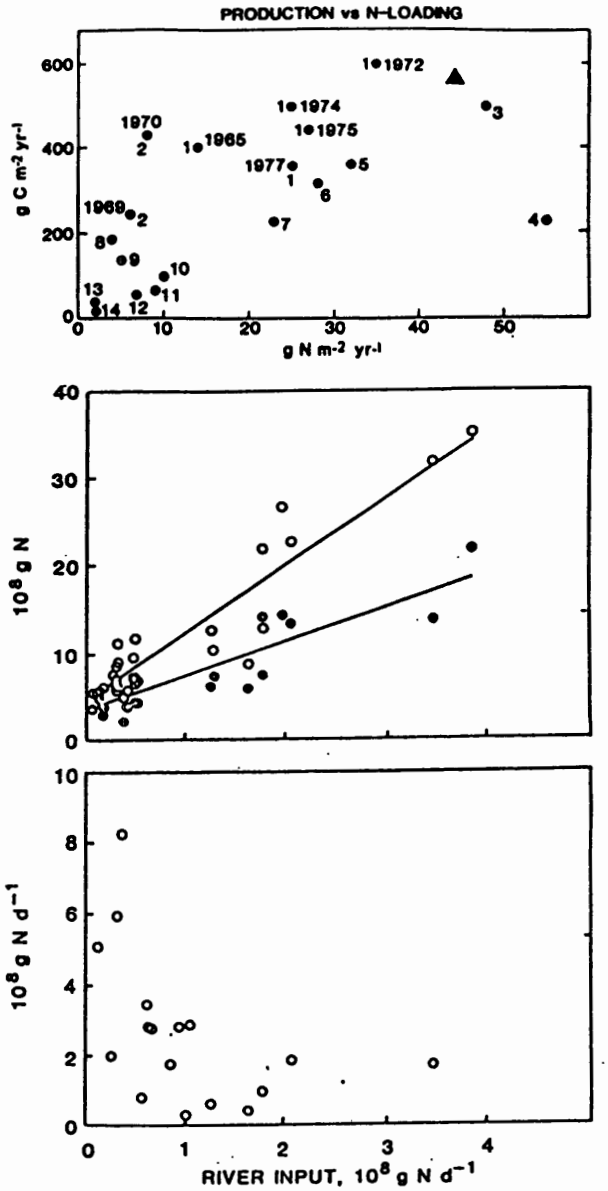

Figure 2. Relationships between (upper panel) annual phytoplankton production and nitrogen loading and (lower panels) monthly mean phytoplankton biomass (10^8 g N, euphotic zone = ●, water column = ○), nitrogen assimilation (10^8 g N d^{-1}), and riverine nitrate input (from Malone et al., in press).

gradient [MCarthy et al. 1977, Fisher et al., in press]. It appears that most of the nitrogen in the water column ultimately ends up in the DON pool. Total nitrogen decreases, but in the water column there is a shift from mainly nitrate to mainly DON. This would imply that nitrogen is being exported from the Bay in the form of DON or is lost via denitrification and burial. The ratios of total nitrogen to phosphorus decrease along the axis of the Bay, from very high ratios (>60) to ratios approaching Redfield. This change implies a net loss of

Table 1. Nutrients in Chesapeake Bay and its coastal plume: changes in distribution and stoichiometry of N and P from freshwater to the sea [Fisher et al., in press].

	TN (μM)	TP (μM)	NO_3	TON (%)	PO_4	TOP (%)	Inorganic	Organic
Fresh	129	1.4	67	28	27	73	235	35
Sea	18	1.4	3	83	31	69	6	17

approaching Redfield. This change implies a net loss of nitrogen relative to phosphorus, and phosphorus is actually exported from the Bay. The concentration of phosphate in the freshwater endmember and that in the seawater endmember are similar as are the percentages of inorganic and organic P. Phosphorus recycles rapidly as it goes down the axis, but its distribution between pools does not change significantly. This suggests that the community metabolism of the Bay as a whole is nitrogen-limited.

QUESTIONS

Q: We have little data on seaweed and microbenthic algae, but some of these species might also be a major source of organic material in the system, especially the detritus feeders. The microbenthic bloom in the spring does not occur at the same time as the phytoplankton bloom, and there are other discrepancies. There could be another whole system here that we know very little about.

A: I agree that the conclusion of Flemer and Biggs (that phytoplankton are the major source of organic material downstream of the turbidity maximum) needs to be reevaluated, especially in the light of your comment. But in terms of the accumulation of phytoplankton biomass, there is so little light reaching the bottom at the time of the phytoplankton bloom that you would not expect microbenthic production to be a major contributor during that time.

Table 2. Nutrient levels (μM) and ratios in the coastal plume of Chesapeake Bay [Garside, personal comm.]

Month	DIN	PO_4	$Si(OH)_4$	N/P	N/Si
February	0.22	0.29	0.14	0.8	1.6
April	1.35	0.23	0.73	5.9	1.8
June	0.22	0.63	2.98	0.3	0.1
August	0.29	0.78	1.82	2.7	0.2

REFERENCES

Boynton, W.R.; Kemp, W.M.; Keefe, C.W. A comparative analysis of nutrients and other factors influencing estuarine phytoplankton production. In: Kennedy, V.S. (ed.) Estuarine comparisons. Academic Press, New York, p. 69-90; 1982.

D'Elia, C.F.; Sanders, J.G.; Boynton, W.R. Nutrient enrichment studies in a coastal plain estuary: phytoplankton growth in large-scale, continuous cultures. Can. J. Fish. Aquat. Sci. 43:397-406; 1986.

Fisher, T.R., Harding, L.W., Stanley, D.W., Ward, L.G. Phytoplankton, nutrients, and turbidity in the Chesapeake, Delaware, and Hudson estuaries. Estuar. Coast. Shelf Sci. (in press).

Harding, L.W.; Meeson, B.W.; Fisher, T.R. Phytoplankton production in two east coast estuaries: photosynthesis-light functions and patterns of carbon assimilation in Chesapeake and Delaware Bays. Estuar. Coast. Shelf Sci. 23:773-806; 1986.

McCarthy, J.J.; Taylor, W.R.; Taft, J.L. 1977. Nitrogenous nutrition of the plankton in the Chesapeake Bay. Limnol. Oceanogr. 22:996-1011.

Malone, T.C.; Kemp, W.M.; Ducklow, H.W.; Boynton, W.R.; Tuttle, J.H.; Jonas, R.B.; Lateral variation in the production and fate of phytoplankton in a partially stratified estuary, Mar. Ecol. Prog. Ser., 32:149-160; 1986.

Schubel, J.R., Pritchard, D.W. Responses of upper Chesapeake Bay to variations in discharge of the Susquehanna River. Estuaries 9:236-249; 1986.

White, J.R., Roman, M.R. Grazing and egg production by Chesapeake Bay zooplankton in spring and summer. In Lynch, M.P., and Krome, E.C., eds. Understanding the Estuary: Advances in Chesapeake Bay Research. Proceedings of a conference, 29-31 March 1988, Baltimore, Md. Chesapeake Research Consortium, Gloucester Point, VA. CRC publication no. 129.

Deposition of Organic Matter to the Sediment Surface

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My topic is the connection between particulate organic matter in the overlying water and its deposition to the sediments. In addition, I want to discuss carbon balancing.

The technique we are using to measure deposition is fixed moorings of sediment trap arrays collecting at three depths with multiple cups. The upper cup is in the upper mixed layer, or the euphotic zone, the middle cup is designed to be at the pycnocline, and the lower cup (which largely measures resuspension) is in the lower layer.

TEMPORAL DISTRIBUTION

For 1985 and 1986 we measured deposition rates, chlorophyll standing stocks in the water column, and the carbon production rates in the overlying waters. We have over three years of data on deposition rates, and a distinct annual pattern emerges: a spring pulse of deposition of chlorophyll, a hiatus, then relatively high deposition rates in the summer, and often high deposition in the fall (which we know little about). The question that arises is what is the source of the chlorophyll that is deposited. We are measuring only fluorometric chlorophyll and lack data on pigment degradation. What we can address is the relationship of the chlorophyll standing stocks, the production rates, and the deposition patterns, particularly the pulses.

The carbon:chlorophyll ratios are relatively low in the spring and high in the summer; they give us some idea of the nature of the material that is being deposited. When the ratio is high, we suspect that the material being deposited has already undergone some degradation, since chlorophyll degrades more rapidly than carbon. The nitrogen:phosphorus ratios relative to carbon can give some idea about nutrient limitation, although interpretation based on these ratios should be cautious. Throughout the spring and summer of 1984 and 1985, both the carbon:phosphorus and nitrogen:phosphorus ratios suggested a deficiency of phosphorus. This may suggest that particulate production, the phytoplankton themselves, may be phosphorus-limited at this season.

VERTICAL DISTRIBUTION

In both 1985 and 1986, samples from the surface waters indicated phosphorus deficiency. The relative proportion of phosphorus also increases as particles fall from the surface toward the bottom. It may be that these particles strip the available phosphorus from the water as they settle, probably through chemical sorption processes. It is also worth noting (although time precludes a discussion here today) that there are definite differences between seston and the material in the sediment traps.

Year to year variations are apparent in the phosphorus:carbon and phosphorus:nitrogen ratios. A comparison of 1984, a relatively wet year, with 1986, a relatively dry year, shows definite separation of these ratios, with river flow appearing as a dominant forcing function. Associated with river flow are the changes in stratification and in delivery of nutrients.

Euphotic zone chlorophyll and deposition of chlorophyll are correlated, but differently in the spring than in the summer. The relationship is still consistent with the hypothesis to be presented by Malone and Roman and White later in this meeting, as far as the mechanism for delivery of phytoplankton to the bottom. In the summer the chlorophyll stocks in the water turn over much more rapidly than in the spring, perhaps largely because of grazing. In general the relationship between primary production and deposition is weak. Long-term well-integrated data sets (covering years) might reveal these relationships, but at the scale we are studying (month to month) they are not readily found. Such a relationship was found, however, in 1984, which was an exceptional year. A plot of primary production on the x-axis vs. sedimentation of carbon on the y-axis, with data partitioned for the flanks and channel, shows

a relationship in the flanks but not in the water directly above the traps. If we interpret this mechanistically, we must invoke deposition, resuspension, and ultimately some lateral transport.

A plotting of production vs. water column respiration for the spring-summer period shows the positive relationship that one would expect to see, but with a great deal of scatter.

If net daytime primary production is calculated, and then the respiration of the upper layers and the lower layers is subtracted, the residual should be the material that is deposited on the sediment surface. That residual correlates very closely with actual trap collections. Either there are lots of compensating errors, or these data are beginning to be believable. We can begin to try to develop carbon budgets for the water column, to understand relationships between production and deposition, and to examine the community metabolism in the benthos.

Uptake and Release of Nutrients from Sediments

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The Chesapeake Bay is a very appropriate place to study sediments. Tom Horton in his recent book *Bay Country* put it as well as it can be put: "When you think Chesapeake Bay, think skinny." We have lots of watershed and lots of sediments, but almost no water, even though the Chesapeake is one of the largest estuaries, in terms of area, in the U.S. They Bay is shallow, and hence we could expect a substantial relationship between the waters and the sediments. Under these circumstances it makes sense to study the benthos, which we can think of as the memory, or the modulator of many processes in the Bay.

The uptake and release of nutrients from sediments has been measured at a number of stations, most of them in Maryland. An opportunity has arisen recently to collaborate with Virginia workers in the collection of these data for the lower portion of the Bay. Data from these additional stations should give us a better perspective on these processes along a substantial estuarine gradient.

Our instrumentation is simply a box core that is lowered to the surface of Bay sediments. The microcosm that is obtained clearly shows the small but important structures of the bottom. Water in the microcosm is sampled every half hour for five to six hours, and the change in concentration over time of such variables as oxygen and nutrients is noted. Then it is possible to calculate the flux from the water to the sediments or vice versa.

SPATIAL PATTERNS

In some of the tributaries, the sediment oxygen demand (SOD) is quite high — in the range of 1.5-1.7 g O_2 m^{-2} d^{-1}. In the mainstem the rates are lower, for reasons we are not sure of. Extremely high ammonium fluxes are seen in two places: the lower Potomac River and mid-Bay regions. Both of these locations experience frequent summer anoxia. The rates of ammonium flux and SOD at these locations are comparable with rates for very productive estuaries. Large fluxes of phosphorus are also seen at two stations that become anoxic. The spatial patterns on the Bay bottom are emerging, but slowly. Variability is high so that patterns can be difficult to detect, but we believe that we are now beginning to see some real patterns,such as those mentioned above.

TEMPORAL PATTERNS

A large peak in SOD is seen in the spring, probably largely in response to the deposition of large amounts of material from the water column to the sediments. SOD is lower but still substantial through the summer and fall. This pattern is found at many of the stations. Another common pattern, found in the upper tributaries, is peak ammonium fluxes not in the spring but in the summer. So there is a kind of disjointedness about SOD being high in the spring but ammonium release from the sediments tending to be higher in the summer, particularly at the deep hypoxic and anoxic stations.

REGULATING FACTORS

A number of factors may be regulating sediment fluxes: bottom water concentrations of oxygen or some other nutrient; the rate at which organic matter is deposited to the sediment surface; the characteristics of the sediment itself; and temperature.

A plot of bottom water oxygen concentration vs. SOD is so scattered that it is hard to draw any conclusions at all. One point is clear, however; when oxygen concentration in the bottom water is below about 2 mg/l, high SOD is never found. Perhaps at this range oxygen concentration is limiting the demand. More important, this kind of plot tells us that we must stop thinking only of simple, single-factor analysis.

Another paradigm is that as deposition increases, so does SOD. Our data indicate that this could be the case. The hint of relationship between deposition and

SOD response is heartening. Malone has talked about forcing from the land in plankton response, and Kemp has seen a response in deposition from plankton characteristics. We are seeing parts of this model coming together: some response of SOD to the magnitude of deposition.

There is a range of about a factor of three in the amount of particulate nitrogen in the surficial sediments (approximately the top 1 cm). Does that particulate nitrogen in the very top sediments — recently deposited material — send us any message regarding what enters or leaves the sediments? The answer seems to be yes. The flux of nitrogen from the sediments to the water increases as the amount of particulate nitrogen in the sediments increases. This may imply that the sediment memory is short, on the order of months to a year, rather than years to decades.

We have also found that temperature and ammonium flux are closely related, in an exponential function. Comparison of these data in the Chesapeake with other data from Narragansett Bay reveals the same shape, but a different curve. The differences may stem from differences in depositional rates. Phosphorus flux is also exponentially related to temperature, and the curves in the Chesapeake and Narragansett bear the same relationship with each other.

FUTURE STUDIES

We know a little about total sediment metabolism; we measure oxygen consumption; but there is a lot of carbon and nitrogen and phosphorus processing in areas of the Bay where there is no oxygen. We need to know more about the metabolism in anoxic areas as well.

Another area for future study is nitrification and denitrification. Denitrification is important because it is a terminal sink for nitrogen. Since nutrient control is a management issue, understanding the magnitude and factors regulating these processes is of prime importance.

Finally, appropriate time scales for measurements are all-important. Not only is it important to think about what we are measuring, but it is also essential to think about how often variables are measured. Appropriate time scale can mean the difference between meaningless data and data that answer some questions.

The Trophic Structure of Pelagic Communities: Hypotheses and Problems

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INTRODUCTION

A number of factors influence the trophic structure of pelagic communities, and their interrelationships can be very complex. The forcing functions range from natural environmental phenomena to anthropogenic inputs to inter- and intraspecies interactions. The pertinent time scales can vary from hours to decades. The effects may be direct or indirect, they may operate on an organism's physiology, tolerances, and resource supply, or they may influence its predators. There are also random fluctuations to consider, as well as interannual variability and long-term periodicities.

Traditionally, natural populations were thought to be limited primarily by the supply of food, habitat, or other resources. Beginning in the 1960's, theoretical ecologists proposed that predation was as important as resource limitation in structuring plant and animal communities. The field of aquatic ecology was not immune to these developments. For example, enhanced primary production was long considered to be due almost exclusively to eutrophication. However, limnologists quickly tested the predation theory, and their results demonstrated that the presence or absence of primary and secondary carnivory was a dominant force in determining the magnitude of primary production. According to this theory, changes at the top of the food web cascade down to alter the structure at lower trophic levels.

Both of these scenarios (top-down and bottom-up regulation) emphasize changes in abundance. Perhaps more important to the structure of food webs are changes in community composition. This point is often overlooked in the current emphasis on rate measurements and flux calculations, but a growing body of evidence, both anecdotal and quantitative, suggests its importance. To illustrate this, I have selected three trophic groups that appear to be important in the Chesapeake Bay (dinoflagellates, copepods, and gelatinous predators). I have listed some of the potential effects of their dominance, effects that may cascade both up and down pelagic food webs. Here I should emphasize that the proposed relationships and mechanisms are those thought to be operating in other estuaries, and therefore are best viewed as food for thought in the context of Chesapeake Bay. After illustrating these, I will underscore some of the difficulties in ascribing cause and effect in trophic responses, which include interannual variability, long-term climatic changes, and natural fluctuations.

DINOFLAGELLATES

The dinoflagellate class of phytoplankton is abundant in the Bay, particularly in the summer. They have certain physiological characteristics with significant ramifications for both pelagic and benthic food webs:

- Attenuated irradiance and subsurface nutricline favor motile taxa [Eppley and Harrison 1975].
- Layered, reverse-flow structure favors retention of migrating cells [Tyler and Seliger 1978, 1981] and accumulation of benthic cysts which can seed subsequent blooms [Tyler et al. 1982].
- Some dinoflagellates excrete substances that inhibit growth rates of diatoms and their synthesis of antibacterial compounds [Aubert and Pesando 1971; Uchida 1977].
- Several species are poor food for copepods, altering their behavioral feeding patterns and inducing reverse peristalsis and regurgitation [Huntley et al. 1986; Gill and Harris 1987].
- The summer-dominant copepod (*Acartia tonsa*) does not feed actively on co-occurring Chesapeake Bay dinoflagellates, and avoids vertically migrating populations [Sellner and Olson 1985].
- Blooms of inedible dinoflagellates have been implicated in regulating copepod dynamics elsewhere [Lindahl and Hernroth 1983].
- Dinoflagellates are poor food for oyster larvae

[Davis and Chanley 1956], perhaps due to interference by ejection of trichocysts [Ukeles and Sweeney 1969].
- At least one species of dinoflagellate passes undigested through guts of adult oysters [Galtsoff 1964].
- Bloom concentrations of non-toxic species have been implicated in mortality of larval fish [Potts and Edwards 1987], perhaps due to exudates that clog their gills [Jenkinson 1986].
- Ciliate grazing can regulate the timing and magnitude of dinoflagellate blooms [Watras et al. 1985].
- Dinoflagellates are good food for larval and juvenile menhaden [June and Carlson 1971; Friedland et al. 1984; Stoecker and Govoni 1984].

From such data, one might conclude that copepods and oysters would have difficulty in a dinoflagellate-dominated system. In contrast, the simultaneous occurrence of large populations of dinoflagellates, ciliates, and menhaden in Chesapeake Bay may not be entirely coincidental.

SPRING BLOOM HERBIVORY

The spring diatom bloom in the Bay seems to occur at a time and place in which an abundant crustacean zooplankton community is seldom present. Observations relevant to this trophic mismatch include:
- Phytoplankton biomass peaks in the mesohaline mainstem in late spring [Malone 1987].
- Total copepod abundance is low and declining [Allan et al. 1976; Burton et al. 1986; Brownlee and Jacobs 1987].
- The winter-dominant copepod *Eurytemora affinis* is apparently caught in a temperature-salinity squeeze [Roddie et al. 1984].
- Salinities are generally too low for *Acartia hudsonica* [Jeffries 1962] and, where salinities are acceptable, temperatures are too warm [Sullivan and McManus 1986].
- Spring temperatures are too low for the copepod *Acartia tonsa*.
- Rotifers and protozoans are seasonally abundant but spatially heterogeneous [Allan et al. 1976; Brownlee and Jacobs 1987]; their distribution with respect to the spring bloom has not been described.

The interesting point is that in estuaries to the south of Chesapeake Bay, temperatures are warm enough for *Acartia tonsa* to dominate year-round. Thus, one might speculate that the minimal grazing of the spring bloom by copepods may be partially determined by the geographic location of the Bay. If it were located further north, either *Eurytemora* or *Acartia hudsonica* would do fine, depending on salinity; and if it were further south, the euryhaline *Acartia tonsa* would dominate. This apparent absence of significant pelagic grazing by crustaceans is a point I will return to later.

GELATINOUS PREDATORS

Evidence that ctenophores and medusae can drastically reduce crustacean zooplankton stocks is extensive [Huntley and Hobson 1978; Moller 1979; Deason and Smayda 1982]. They appear to be abundant in Chesapeake Bay, especially in summer, where their impact may be considerable.
- Associated cascading effects on other trophic levels may include diminished survival of fish larvae and reduced recruitment [Greve and Reiners 1980; Moller 1980; Parsons and Kessler 1987], and elevated phytoplankton production [Deason and Smayda 1982].
- Interannual variability in gelatinous predators in Narragansett Bay was strongly correlated with crustacean zooplankton biomass immediately prior to their development [Deason and Smayda 1982].
- Survival and growth of juvenile ctenophores and medusae is enhanced by an abundant protozoan community [Stoecker et al. 1987].
- Most oyster larvae in Chesapeake Bay could potentially be removed by ctenophore predation. In Barnegat Bay [Nelson 1925] ctenophores contained an average of 14 larvae = daily ingestion of 168-336 larvae/ctenophore, using appropriate gut residence times [Reeve 1980; Sullivan and Reeve 1982]. Oyster larvae concentrations in Choptank River were 12 x $10^3/m^3$ [Seliger et al. 1982]. Ctenophore concentrations are 10-100/m^3 [Miller 1974; Kremer and Nixon 1976]. Combining these data suggests a potential daily ingestion of 14-28% to 140-280%.

Two important conclusions can be derived from these considerations. First, the magnitude of gelatinous predator stocks is a function of the food available at the time of their appearance. This correlation has potential implications for the relationship between phytoplankton species composition and zooplankton success. Second, young ctenophores and medusae show better survival and growth on diets of microzooplankton than on diets of crustacean zooplankton. Since dinoflagellates are a good food for microzooplankton, a direct trophic link may exist between dinoflagellate blooms and large stocks of gelatinous predators.

VARIABILITY

I emphasize that these proposed trophic relationships are hypothetical, at least in the context of the Chesa-

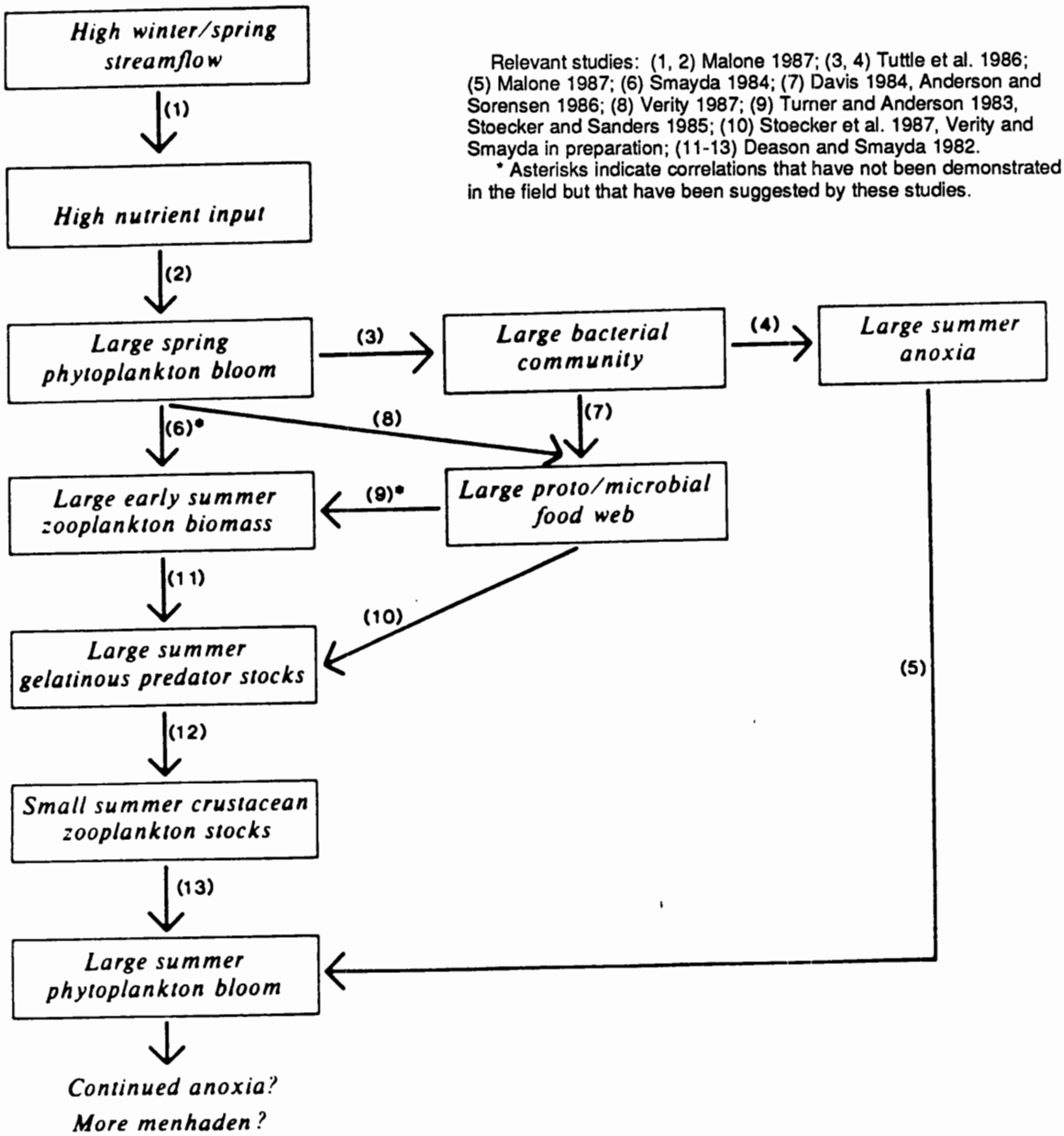

Figure 1. Hypothetical trophic relationships among major plankton components in the Chesapeake Bay during spring and summer in years with high freshwater runoff.

peake Bay. The difficulty in ascribing cause and effect reflects the difficulty in identifying the number of dependent and independent variables operating in natural systems at any given moment, and in regulating them during experimental studies. As an example, consider the effects of interannual variability on these processes (Figure 1).

Nutrient input to the Bay is thought to be a function of streamflow: years with high flow introduce more nutrients than low-flow years. The magnitude of the spring bloom is related to this input. Bacterial production is coupled to phytoplankton production, and their degradation of the settled bloom is thought to be a primary driver of summer anoxia. A by-product of that metabolic activity is regenerated nutrients, such that the summer bloom may be determined by the magnitude of the spring bloom, which is ultimately a function of streamflow.

What is especially provocative about the Bay (if it is true) is the lack of a direct trophic connection between spring pelagic production and summer dynamics. As discussed earlier, this may partially reflect the geography of the Bay. Elsewhere, there appears to be a more direct trophic connection, although the phenomenon is

poorly documented. Alternatives may exist in the Chesapeake either via grazing of bacteria by protozoans or their direct ingestion of phytoplankton. In either case, an active microbial/protozoan food web may support both crustacean omnivores and larval and juvenile gelatinous predators. In any event, the factors regulating the magnitude of the early summer zooplankton biomass may be critical, as elsewhere the magnitude of this biomass determines the success of gelatinous organisms. In those systems, years with large stocks of gelatinous predators are years with extensive summer phytoplankton blooms. If this relationship is applicable to the Chesapeake Bay, there are several possible effects of such interannual variability on food web structure.

If interannual variability were not enough of an obstacle to interpreting changes in trophic structure, there are also long-term climatic effects and periodicities to resolve. Documented environmental phenomena include:

- Declines in incident irradiance over the northeast Atlantic, 1948 to 1965 [Cushing and Dickson 1976]
- The dark decade in New England during the 1970's [Smayda 1984]
- Eleven-year and 180-year sunspot cycles [Southward et al. 1975]
- Temperature cycles [Taylor et al. 1957; Jeffries and Terceiro 1985; Brady 1976; Cushing and Dickson 1976; Sutcliffe et al. 1977]
- Long-term changes in wind strength [Taylor and Stephens 1980] and circulation patterns of the Gulf Stream, the northeast Atlantic, and the Mediterranean [Taylor and Stephens 1980; Vucetic 1983]
- Periodicity in precipitation and runoff [Biggs and Smullen 1987]

Proposed effects of these phenomena on pelagic food webs include changes in:

- Timing, magnitude, duration, and community composition of phytoplankton blooms and of zooplankton responses [Cushing and Dickson 1976; Southward 1983; Smayda 1984]
- Overwintering survival of ctenophores [Frank 1986]
- Fecundity of planktivorous fish [Tanasichuk and Ware 1987]
- Recruitment and year-class strength of planktivorous and piscivorous fish [Cushing and Dickson 1976]
- Yield of commercial fisheries [Sutcliffe et al 1977]
- Shifts in dominance from fish to gelatinous predators [Lindahl and Hernroth 1983]

Finally, natural fluctuations can also alter the structure of pelagic food webs. Perhaps the best example is from the English Channel, where phytoplankton, crustacean and gelatinous zooplankton, and demersal and pelagic fish showed dramatic changes in community structure associated with climatic variability (Figure 2). As changes in current patterns caused warmer water to invade the English Channel for 30-40 years, the cold-water flora and fauna of the 1920's were replaced by warm-water species in the 1950's. Nutrient concentrations declined, large calanoid copepods became rare, the chaetognath *Sagitta elegans* was replaced by *S. setosa*, and herring disappeared and were replaced by pilchards. In the 1970's, cold water returned and the plankton reverted to that characteristic of the 1920's. What is provocative is that the herring did not return, but instead the pilchards were replaced

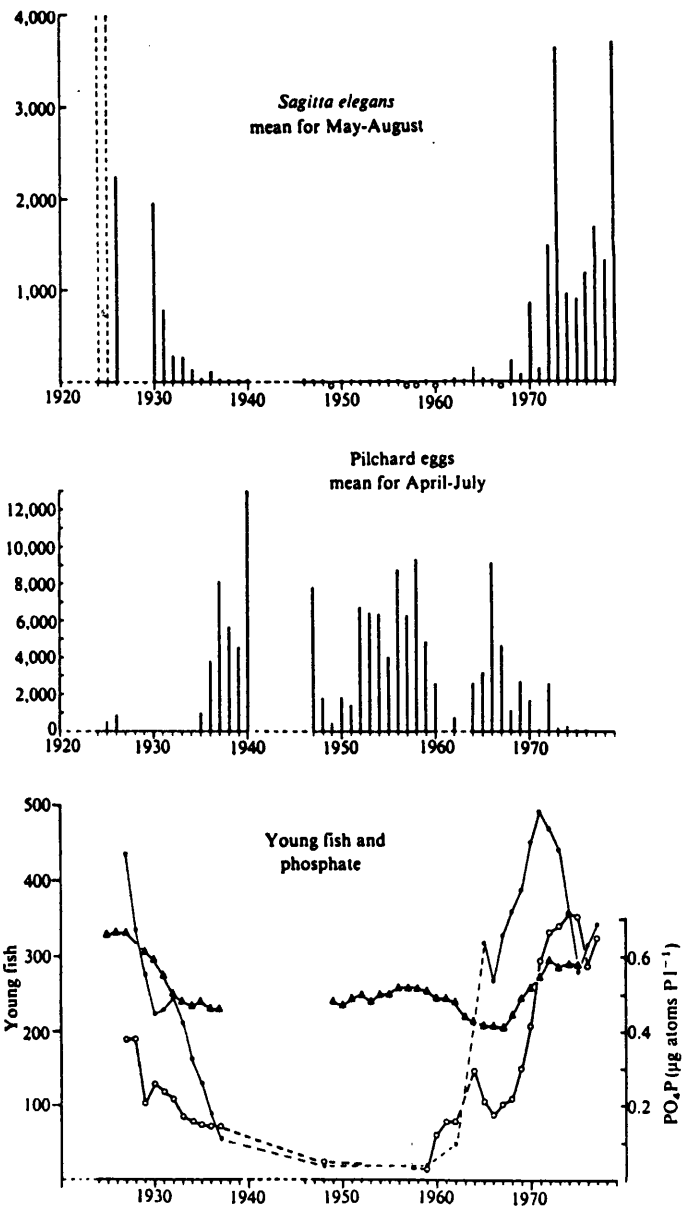

Figure 2. Long-term (1920-1979) changes in plankton community structure in the English Channel [Southward 1980].

by mackerel. This change is thought to be due to coincidence between the timing of the return of favorable conditions and natural fluctuations in herring and mackerel recruitment. Such a hysteresis underscores an assumption that is often inherent in ecosystem analysis and in management strategies, that if the addition of a species or a change in environmental forcing has a certain effect, then its removal will reverse that effect. These kinds of data emphasize, however, that different constraints may be operating at that time, and new species interactions may be more persistent than previous ones. Thus, the structure of pelagic food webs is perhaps best viewed as the product of processes with strong historical components, such as eutrophication, and those processes requiring more continual expression, such as predation. Our challenge for the future is to quantify and predict their relative roles in regulating the trophic structure of pelagic communities.

RELEVANT LITERATURE

Allan, J.D.; Kinsey, T.G.; James, M.C. Abundances and production of copepods in the Rhode River subestuary of Chesapeake Bay. Ches. Sci. 17:86-92; 1976.

Andersen, P.; Sorensen, H.M. Population dynamics and trophic coupling in pelagic microorganisms in eutrophic coastal waters. Mar. Ecol. Prog. Ser. 33:99-109; 1986.

Aubert, M.; Pesando, D. Telemediateurs chemiques et equilibre biologique oceanique. Deuxieme partie. Nature chemique de l'inhibiteur de la synthese d'un antibiotique produit par un diatomee. Revue int. Oceanogr. med. 21:17-22; 1971.

Biggs, R.B.; Smullen, J.T. Freshwater inflow to the upper Chesapeake Bay and Potomac River: seasonal to century-long variations. Eos 68:1733; 1987.

Brady, D.K. Are the Chesapeake Bay waters warming up? Chesapeake Sci. 17:225-227; 1976.

Brownlee, D.C.; Jacobs, F. Mesozooplankton and microzooplankton in the Chesapeake Bay. In: Majumdar, S.K.; Hall, L.V. Jr; Austin, H.M., eds. Contaminant problems and management of living Chesapeake Bay resources. Philadelphia: Pennsylvania Acad. Sci., in press; 1987.

Burton, W.; Moss. I.; Jacobs, F. Chesapeake Bay water quality monitoring progam mesozooplankton component: August 1984 - December 1985. Maryland Office of Environmental Programs, Baltimore, Maryland; 1986.

Cushing, D.H.; Dickson, R.R. The biological response in the sea to climatic changes. In: Russell, F.S.; Yonge, M., eds. Adv. Mar. Biol. London: Academic Press, Inc.: 14:2-122; 1976.

Davis, H.C.; Chanley, P.E. Effects of some dissolved substances on bivalve larvae. Proc. Natl. Shellfisheries Assoc. 96:59-74; 1956.

Davis, P.G. Geographic, seasonal and diel distribution of phototrophic and heterotrophic components of the picoplankton and nanoplankton in the North Atlantic. PhD Thesis Univ. Rhode Island; 1984.

Deason, E.E.; Smayda, T.J. Ctenophore-zooplankton-phytoplankton interactions in Narragansett Bay, Rhode Island, USA, during 1972-1977. J. Plank. Res. 4:203-217; 1982.

Eppley, R.W.; Harrison, W.G. Physiological ecology of *Gonyaulax polyhedra*, a red water dinoflagellate of southern California. In: LoCicero, V.R., ed. Proc. First Int. Conf. on Toxic Dinoflagellate Blooms. Wakefield, MA: Mass. Sci. Technol. Foundation; 11-22; 1975.

Frank, K.T. Ecological significance of the ctenophore *Pleurobrachia pileus* off southwestern Nova Scotia. Can. J. Fish. Aquat. Sci. 43:211-222; 1986.

Friedland, K.D.; Haas, L.W.; Merriner, J.V. Filtering rates of the juvenile Atlantic menhaden *Brevoortia tyrannus* (Pisces: Clupeidae), with consideration of the effects of detritus and swimming speed. Mar. Biol. 84:109-117; 1984.

Galtsoff, P.S. The American oyster *Crassostrea virginica* Gmelin. Fish. Bull. 64:219-238; 1964.

Gill, C.W.; Harris, R.P. Behavioral responses of the copepods *Calanus helgolandicus* and *Temora longicornis* to dinoflagellate diets. J. Mar. Biol. Assoc. U.K. 67:785-801; 1987.

Greve, W.; Reiners, F. The impact of predator-prey waves from estuaries on the planktonic marine ecosystem. In: Kennedy, V.S., ed. Estuarine Perspectives. New York: Academic Press, Inc.: 405-435; 1980.

Huntley, M.E.; Hobson, L. Medusa predation and plankton dynamics in a temperate fjord. J. Fish. Res. Bd. Can. 35:257-261; 1978.

Huntley, M.; Sykes, P.; Rohan, S.; Martin, V. Chemically-mediated rejection of dinoflagellate prey by the copepods *Calanus pacificus* and *Paracalanus parvus*: mechanism, occurrence and significance. Mar. Ecol. Prog. Ser. 28:105-120; 1986.

Jeffries, H.P. Succession of two *Acartia* species in estuaries. Limnol. Oceanogr. 7:354-364; 1962.

Jeffries, H.P.; Terceiro, M. Cycle of changing abundances in the fishes of the Narragansett Bay area. Mar. Ecol. Prog. Ser. 25:239-244; 1985.

Jenkinson, I.R. Oceanographic implications of non-newtonian properties found in phytoplankton cultures. Nature 323:435-437; 1986.

June, F.C.; Carlson, F.T. Food of young Atlantic menhaden, *Brevoortia tyrannus*, in relation to metamorphosis. Fish. Bull. 68:493-512; 1971.

Kremer, P.; Nixon, S.W. Distribution and abundance of the ctenophore, *Mnemiopsis leidyi*, in Narragansett Bay. Est. Coast. Shelf Sci. 4:627-639; 1976.

Lindahl, O.; Hernroth, L. Phyto-zooplankton community in coastal waters of western Sweden - an ecosystem off balance? Mar. Ecol. Prog. Ser. 10:119-126; 1983.

Malone, T.C. Nutrient limited phytoplankton production in the mesohaline reach of Chesapeake Bay. EOS 68:1688; 1987.

Miller, R.J. Distribution and biomass of an estuarine ctenophore population, *Mnemiopsis leidyi* (A. Agassiz).

Chesapeake Sci. 15:1-8; 1974.
Moller, H. Significance of coelenterates in relation to other plankton organisms. Meeresforsch. 27:1-18; 1979.
Moller, H. Scyphomedusae as predators and food competitors of larval fish. Meeresforsch. 28:90-100; 1980.
Nelson, T.C.. On the occurrence and food habits of ctenophores in New Jersey inland coastal waters. Biol. Bull. 48:92-111; 1925.
Parsons, T.R.; Kessler, T.A. An ecosystem model for the assessment of plankton production in relation to the survival of young fish. J. Plank. Res. 9:125-137; 1987.
Potts, G.W.; Edwards, J.M. The impact of a *Gyrodinium aureolum* bloom on inshore young fish populations. J. Mar. Biol. Assoc. U.K. 67:293-297; 1987.
Reeve, M.R. Comparative experimental studies on the feeding of chaetognaths and ctenophores. J. Plank. Res. 2:381-393; 1980.
Roddie, B.D.; Leakey, R.J.G.; Berry, A.J. Salinity-temperature tolerance and osmoregulation in *Eurytemora affinis* (Poppe) (Copepoda: Calanoida) in relation to its distribution in the zooplankton of the upper reaches of the Forth estuary. J. Exp. Mar. Biol. Ecol. 79:191-211; 1984.
Seliger, H.H.; Boggs, J.A.; Rivkin, R.B.; Biggley, W.H.; Aspden, K.R.H. The transport of oyster larvae in an estuary. Mar. Biol. 71:57-72; 1982.
Sellner, K.G.; Olson, M.M. Copepod grazing in red tides of Chesapeake Bay. In: Anderson, D.M.; White, A.W.; Baden, D.G., eds. Toxic dinoflagellates. New York: Elsevier, Inc., 245-250; 1985.
Smayda, T.J. Variations and long-term changes in Narragansett Bay, a phytoplankton-based coastal marine ecosystem: relevance to field monitoring for pollution assessment. In: White, H.H.; ed. Concepts in marine pollution measurements. College Park, MD: Maryland Sea Grant College, Univ. Maryland: 663-679; 1984.
Southward, A.J. The Western English Channel - an inconstant ecosystem? Nature 285:361-366; 1980.
Southward, A.J. Fluctuations in the ecosystem of the Western Channel: a summary of studies in progress. Proceedings 17th European Marine Biology Symposium, Brest, France, 27 September - 1 October. Oceanol. Acta No. SP: 187-189; 1983.
Southward, A.J. Butler, E.I.; Pennycuick, L. Recent cyclic changes in climate and in abundance of marine life. Nature 253:714-717; 1975.
Stoecker, D.K.; Govoni, J.J. Food selection by young larval gulf menhaden (*Brevoortia patronus*). Mar. Biol. 80:299-306; 1984.
Stoecker, D.K.; Sanders, N.K. Differential grazing by *Acartia tonsa* on a dinoflagellate and a tintinnid. J. Plank. Res. 7:85-100;1985.
Stoecker, D.K.; Verity, P.G.; Michaels, A.E.; Davis, L.H. Feeding by larval and post-larval ctenophores on microzooplankton. J. Plank. Res. 9:667-683; 1987.
Sullivan, B.K.; McManus, L.T. Factors controlling seasonal succession of the copepods *Acartia hudsonica* and *Acartia tonsa* in Narragansett Bay, Rhode Island: temperature and resting egg production. Mar. Ecol. Prog. Ser. 28:121-128; 1986.
Sullivan, B.K., Reeve, M.R. Comparison of estimates of the predatory impact of ctenophores by two independent techniques. Mr. Biol. 68:61-65; 1982.
Sutcliffe, W.H. Jr.; Drinkwater, K.; Huir, B.S. Correlations of fish catch and environmental factors in the Gulf of Maine. J. Fish. Res. Bd. Can. 34:19-30; 1977.
Tanasichuk, R.W.; Ware, D.M. Influence of interannual variations in winter sea temperature of fecundity and egg size in Pacific herring (*Clupea harengus pallasi*). Can. J. Fish. Aquat. Sci. 44:1485-1495; 1987.
Taylor, A.H.; Stephens, J.A. Latitudinal displacements of the the Gulf Stream (1966 to 1977) and their relation to changes in temperature and zooplankton abundance in the NE Atlantic. Oceanol. Acta 3:145-149; 1980.
Taylor, C.C.; Bigelow, H.B.; Graham, H.W. Climatic trends and the distribution of marine animals in New England. Fish. Bull. 57:293-345; 1957.
Turner, J.T.; Anderson, D.M. Zooplankton grazing during dinoflagellate blooms in a Cape Cod embayment, with observations of predation upon tintinnids by copecods. Publicazioni della Stazione Zoologica di Napoli I: Marine Ecology 4:359-374; 1983.
Tuttle, J.H.; Malone, T.C.; Jonas, R.B.; Ducklow, H.W.; Cargo, D.G. Nutrient-dissolved oxygen dynamics in Chesapeake Bay: the roles of phytoplankton and microheterotrophs under summer conditions, 1985. Final Report to EPA, Chesapeake Bay Program: UMCEES Ref. No. 86-125a-CBL; 1986.
Tyler, M.A.; Coats, D.W.; Anderson, D.M. Encystment in a dynamic environment: deposition of dinoflagellate cysts by a frontal convergence. Mar. Ecol. Prog. Ser. 7:163-178; 1982.
Tyler, M.A.; Seliger, H.H. Annual subsurface transport of a red tide dinoflagellate to its bloom area: water circulation patterns and organism distributions in the Chesapeake Bay. Limnol. Oceanogr. 23:227-246; 1978.
Tyler, M.A.; Seliger, H.H. Selection for a red tide organism: physiological responses to the physical environmental. Limnol. Oceanogr. 26:310-324; 1981.
Ukeles, R.; Sweeney, B.M. Influence of dinoflagellate trichocysts and other factors on the feeding of *Crassostrea virginica* larvae on *Monochrysis lutheri*. Limnol. Oceanogr. 14:403-410; 1969.
Uchida, T. Excretion of a diatom-inhibitory substance by *Prorocentrum micans* Ehrenberg. Jap. J. Ecol. 27:1-4; 1977.
Verity, P.G. Abundance, community composition, size distribution, and production rates of tintinnids in Narragansett Bay, Rhode Island. Est. Coast. Shelf. Sci. 24:671-690; 1987.
Vucetic, T. Fluctuation in the distribution of the scyphomedusa *Pelagia noctiluca* (Forskal) in the Adriatic. Proceedings 17th European Marine Biology Symposium, Brest, France, 27 Sept. - 1 Oct., Oceanol. Acta. No. SP:207-211; 1983.
Watras, C.J.; Garcon, V.C.; Olson, R.J.; Chisholm, S.W.; Anderson, D.M. The effect of zooplankton grazing on estuarine blooms of the toxic dinoflagellate *Gonyaulax tamarensis*. J. Plank. Res. 7:891-908; 1985.

Trophic Structure of the Chesapeake Mesohaline Ecosystem

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My subject today will be some details of the carbon flow network of the Chesapeake mesohaline ecosystem. Some of you may have seen some of these results, but they bear repeating, partly because they may seem less ridiculous on second hearing, and partly because they fit well into the context set by the two talks preceding mine.

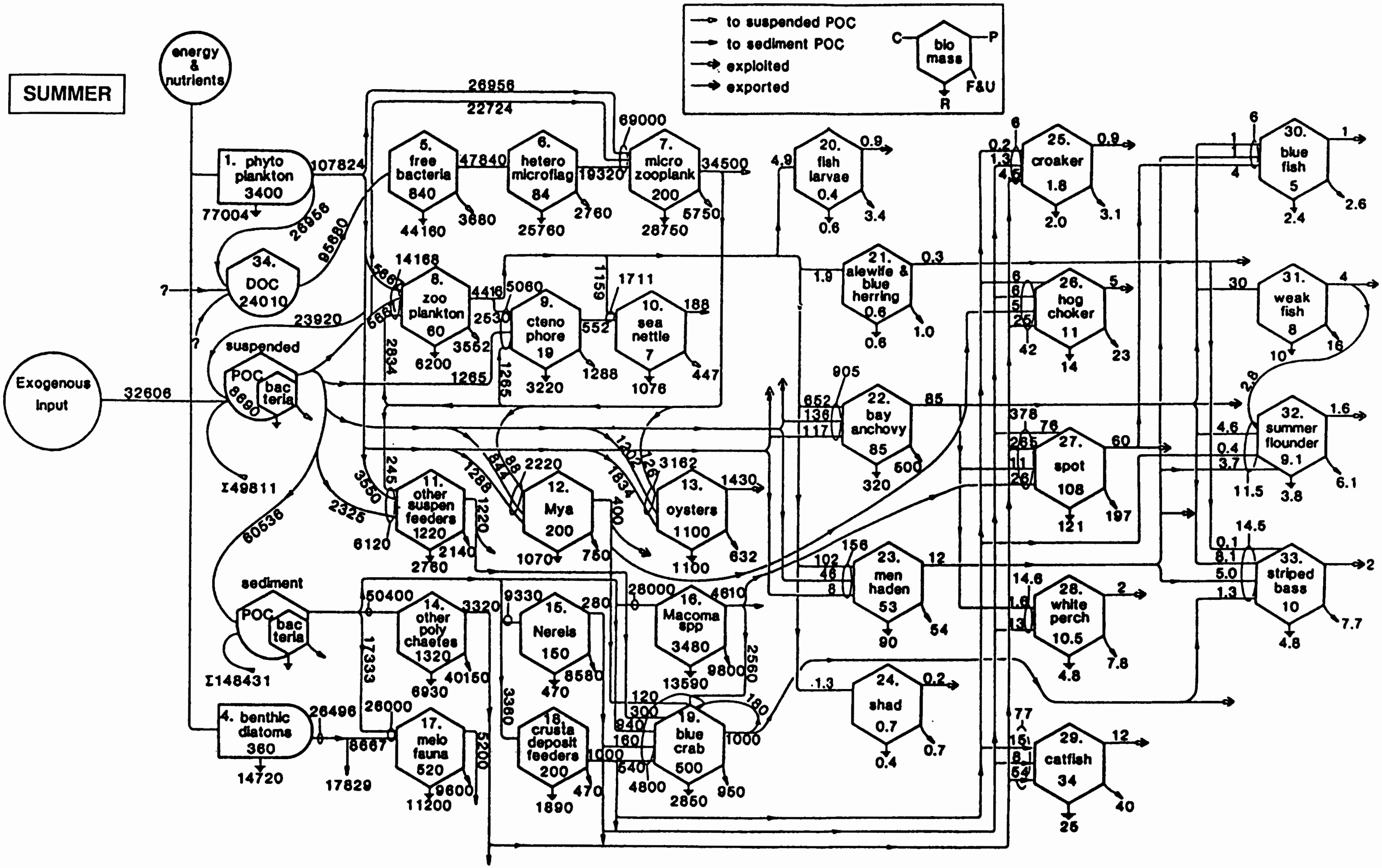
The topic is represented, oddly enough, by a tiny item appearing in the *Baltimore Sun* on Monday, without attribution of any kind, saying: "The most productive fisheries are found in colder areas of the world's oceans, and researchers now believe the reason is that here fish do not have to compete with bacteria for food." Many readers of the *Sun* no doubt scratched their heads in puzzlement; most of us, however, with backgrounds in ecology, understand that the question is one of indirect trophic effects.

The hypothesis that these two elements of the ecosystem do compete in some way would have been approached 10-15 years ago by the creation of a large mathematical model of the Chesapeake Bay system. This approach has fallen out of favor, because these models' predictive ability is problematical and many biologists feel that mathematics do a poor job of representing living, growing, evolving systems. So what approach will work? We must again become ecologists in the true sense of the word, studying the interactions between organisms and their environment. One of the most palpable ways of doing this is to measure the transfers of material and energy through the ecosystem.

This kind of study is just what we proposed to the Maryland Department of Natural Resources several years ago. Dr. Dan Baird collected data from the literature and developed an estimated food web, and I did some of the analysis I'll present now. The area covered was the mesohaline area, about 6-18 ppt. We represented the Bay in 36 compartments (Figure 1). Subdividing it, you will find a planktonic realm, consisting of primary producers, mesozooplankton, and gelatinous predators, as well as the microbial loop or web. In the benthic area are the diatoms, filter feeders, and deposit feeders, and the blue crab. The third large domain is the nekton, consisting primarily of the filter feeders, benthic carnivores, and pelagic piscivores. This kind of network can be shown for each of the seasons, and it is interesting to see that the topology of the system does not change greatly from season to season. Some species migrate out and back in, and the tempo undergoes wide swings, but the only qualitative major change that affects the whole food web is the summertime cropping of the mesozooplankton by the huge stocks of ctenophores.

One of the things that can be done with a network like this, with over 160 exchanges among 36 compartments, is to analyze the exchanges in a matrix. It is possible through mathematical manipulation to estimate the amount transferred between two species and to examine all the indirect effects between them. The matrix of total dependency coefficients that can be calculated yields information about indirect diets, that is, the percentage of a species' total consumption that at some previous time was part of another species (Table 1). In the case of bluefish (species 30), for instance, 7.2% of its intake spent time at some point in compartment 5, bacteria. There are some interesting effects here. For example, 65.8% of striped bass intake was at one time mesozooplankton; the percentage for bluefish was considerably less at 28.7%. Polychaetes show another striking difference: 48.0% of the bluefish diet was at one time polychaetes, whereas the figure for striped bass is only 1.8%. Bacteria eventually make a 7.6% contribution to striped bass and a 7.2% contribution to bluefish.

Figure 1 (overleaf). Estimated food web of the Chesapeake Bay during summer, represented in 36 compartments in the planktonic, benthic, and nektonic realms. Flows are depicted as mg C m^{-2} per 90 days.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Exogenous Input
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4. benthic diatoms 360
5. free bacteria 840
6. hetero microflag 84
7. micro zooplank 200
8. zoo plankton 60
9. cteno phore 19
10. sea nettle 7
11. other suspen feeders 1220
12. Mya 200
13. oysters 1100
14. other poly chaetes 1320
15. Nereis 150
16. Macoma spp 3480
17. meio fauna 520
18. crusta deposit feeders 200
19. blue crab 500
20. fish larvae 0.4
21. alewife & blue herring 0.6
22. bay anchovy 85
23. men haden 53
24. shad 0.7
25. croaker 1.8
26. hog choker 11
27. spot 108
28. white perch 10.5
29. catfish 34
30. blue fish 5
31. weak fish 8
32. summer flounder 9.1
33. striped bass 10
34. DOC 24010
suspended POC bac teria
sediment POC bac teria
Σ49811
Σ148431

Table 1. Total dependency coefficients of 36 compartments of species in the Chesapeake Bay (for identification of compartments, see Table 2).

	25	26	27	28	29	30	31	32	33	34	35	36
1	31.2	35.4	32.3	34.8	31.2	43.7	68.7	65.4	63.9	41.8	42.7	31.2
2	3.3	3.2	3.3	3.2	3.3	3.1	2.3	2.6	2.7	.0	1.4	3.3
3	100.0	90.6	97.0	90.2	100.0	63.8	.0	3.5	6.3	.0	.0	44.1
4	10.0	9.1	9.7	9.0	10.0	6.4	.0	.4	.7	.0	.0	10.0
5	6.8	6.7	6.9	6.9	6.8	7.2	7.3	7.6	7.6	.0	9.7	6.8
6	5.5	5.4	5.6	5.6	5.5	5.9	6.3	6.5	6.5	.0	7.8	5.5
7	8.2	8.2	8.3	8.5	8.2	9.6	11.9	12.0	11.9	.0	11.4	8.2
8	5.1	4.9	7.2	11.8	5.1	28.7	73.0	68.8	65.8	.0	6.8	5.1
9	1.4	1.3	1.4	1.3	1.4	1.2	.9	1.0	1.0	.0	1.7	1.4
10	.1	.1	.1	.1	.1	.1	.1	.1	.1	.0	.1	.1
11	.9	.8	.8	.8	.9	.5	.0	.0	.6	.0	.0	.9
12	.3	9.7	.3	.3	.3	.2	.0	.0	.2	.0	.0	.3
13	.4	.3	.4	.3	.4	.2	.0	.0	.0	.0	.0	.4
14	81.2	68.7	73.0	60.5	77.4	48.0	.0	.9	1.8	.0	.0	24.7
15	26.0	18.6	24.1	42.1	15.0	15.9	.0	.2	.7	.0	.0	5.2
16	5.6	5.0	16.2	5.0	5.6	10.6	.0	.2	4.5	.0	.0	5.6
17	6.9	6.2	6.7	6.2	6.9	4.4	.0	.2	.5	.0	.0	6.9
18	4.1	15.3	1.4	.9	20.5	.9	.0	3.5	1.0	.0	.0	1.0
19	.4	.4	.4	.4	.4	.3	.0	.0	7.9	.0	.0	.4
20	.0	.0	.0	.0	.0	.0	.0	.0	.0	.0	.0	.0
21	.0	.0	.0	.0	.0	.0	.0	.0	.7	.0	.0	.0
22	.3	.3	3.3	10.0	.3	19.6	100.0	65.1	56.6	.0	.0	.3
23	.0	.0	.0	.0	.0	16.8	.0	31.4	34.9	.0	.0	.0
24	.0	.0	.0	.0	.0	.0	.0	.0	.0	.0	.0	.0
25	.0	.0	.0	.0	.0	.0	.0	.0	.0	.0	.0	.0
26	.0	.0	.0	.0	.0	.0	.0	.0	.0	.0	.0	.0
27	.1	.1	.1	.1	.1	65.8	.0	.0	.0	.0	.0	.1
28	.0	.0	.0	.0	.0	.0	.0	.0	.0	.0	.0	.0
29	.0	.0	.0	.0	.0	.0	.0	.0	.0	.0	.0	.0
30	.0	.0	.0	.0	.0	.0	.0	.0	.0	.0	.0	.0
31	.0	.0	.0	.0	.0	.0	.0	16.9	.0	.0	.0	.0
32	.0	.0	.0	.0	.0	.0	.0	.0	.0	.0	.0	.0
33	.0	.0	.0	.0	.0	.0	.0	.0	.0	.0	.0	.0
34	6.8	6.7	6.9	6.9	6.8	7.2	7.3	7.6	7.6	.0	9.7	6.8
35	70.3	67.3	69.7	68.5	70.3	65.6	52.4	56.4	57.3	.0	7.6	70.3
36	100.0	90.6	97.0	90.2	100.0	63.8	.0	3.5	7.2	.0	.0	44.1

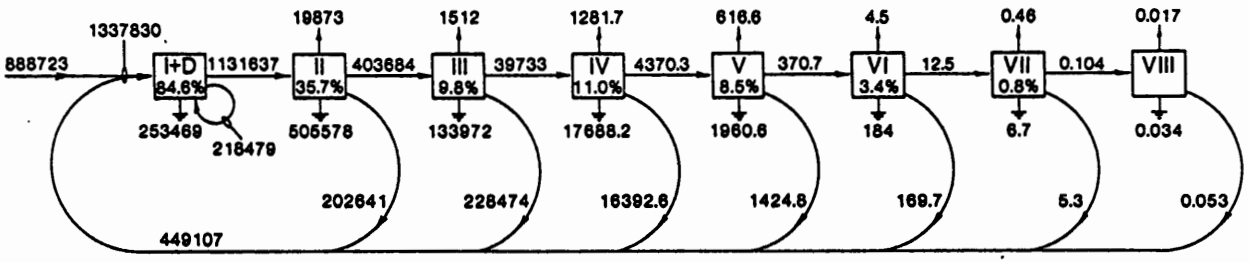

Figure 2. Lindemann-type diagram of the trophic chain. Flows are depicted as mg C m^{-2} per 90 days.

The same matrix algebra can be used to assess the trophic distance of each species from the sources of primary production, and the "average trophic position" at which each species feeds can be calculated (Table 2). An interesting feature of this ranking is that the sea nettle (position 26) has an average trophic level of 3.44, higher than many of the commercially important species in the Bay. The highest trophic position is held by bluefish, but the trophic level is only 4.59, lower than the 5.0 that was deemed significant by Hutchinson [1948] and Pimm [1982].

If we know how much of the diet is coming over pathways of various lengths, we can construct a Lindemann-type diagram of the trophic chain (Figure 2). Each of the boxes now represents parts of populations. One thing to notice is that not much gets past trophic level V, although some pathways reach length VIII. The ratio of detritivory to herbivory in the Bay is 10.5:1, which is really more like a terrestrial system than the open ocean. If we merge the detritus with the plants, we see the classical trophic pyramid that uniformly decreases going up the Lindemann spine. One exception is at trophic level IV, where the ciliates are very efficient at passing on what they have taken in.

We can take the whole system and subtract from it the cycling pathways. Several conclusions can be drawn from the resulting picture. First, it is a bipartite structure, with two non-overlapping domains of carbon recycling in the system, one planktonic and one benthic-nektonic. Among the planktonic, most of the members of the microbial food web are missing. At least for carbon the microbiota do not form a loop, but

Table 2. Trophic rankings and average annual trophic levels of the major components of the Chesapeake Bay mesohaline ecosystem.

Rank	Component	Trophic level	Rank	Component	Trophic level
1	Phytoplankton	1.00	19	Nereis	3.00
2	Dissolved organic carbon	1.00	20	Macoma spp.	3.00
3	Suspended POC	1.00	21	Crustacean deposit feeders	3.00
4	Sediment POC	1.00	22	Ctenophores	3.03
5	Benthic diatoms	1.00	23	Fish larvae	3.16
6	Suspended POC bacteria	2.00	24	Alewife & blue herring	3.16
7	Sediment POC bacteria	2.00	25	Shad	3.16
8	Free bacteria	2.00	26	Sea nettle	3.44
9	Oysters	2.08	27	Blue crab	3.51
10	Mya arenaria	2.09	28	Weakfish	3.84
11	Misc. suspension feeders	2.09	29	Striped bass	3.87
12	Zooplankton	2.16	30	Hogchoker	3.91
13	Meiofauna	2.67	31	White perch	3.98
14	Ciliates	2.75	32	Flounder	3.99
15	Menhaden	2.77	33	Spot	4.00
16	Bay anchovy	2.84	34	Croaker	4.00
17	Heterotrophic microflagellates	3.00	35	Catfish	4.00
18	Misc. polychaetes	3.00	36	Bluefish	4.59

rather serve more as a sink. The filter feeders are not engaged in this recycle; they serve as a shunt taking carbon and energy from the planktonic water column into the benthos and the nekton. Critical arc analysis of the planktonic systems shows that in the summer the sea nettles are probably controlling this system.

The overall picture shows high summer productivity, which along with predator control on the zooplankton fuels "excess" productivity. There are two apparent routes for this excess productivity. One is dissipation in the microbial loop, which now looks like a shunt out of the system; the other is a heavy subsidy to the deposit feeders on the bottom.

In our analysis we were surprised by how much more influential the deposit feeders were than the filter feeders. The question we must now pose is, was this always so? Or was there a change sometime in the last few decades? If so, what was responsible? The ctenophores' grazing control? The quality of the phytoplankton?

REFERENCES

Pimm, S. L. Food webs. Chapman and Hall, London, 1982. 219 pp.

Hutchinson, G. E. The circular causal systems of ecology. Ann. N.Y. Acad. Sci. 50:221-246, 1948.

The Importance of the Microbial Loop in the Chesapeake Bay and its Tributaries

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In recent years many conceptual models of the microbial food web have been developed in terms of carbon transformations, passage to higher trophic levels, and remineralization [Pomeroy 1974; Parsons 1979; Williams 1981]. One such model is shown here (Figure 1) [Azam et al. 1983], and in it you should focus on the second suite of arrows representing the cyanobacteria. The idea is that the microbial food web could be a source of food and energy for higher trophic levels, or it could be a sink.

It should be remembered that although the microbial food web is drawing a lot of attention currently, it has only been in the last 10 to 15 years, with the development of higher-resolution techniques, that we have been able to establish the importance of the very small heterotrophs and autotrophs within all aquatic systems.

The microbial food web is very well established in the Chesapeake Bay, and one question of great interest is whether it can be related to water quality. The forcing functions have been reviewed for almost every level in the trophic system by Verity [1987]. Without repeating all of what he said, we should briefly summarize a few points. First, when Smayda [1983] reviewed phytoplankton in estuaries, he documented that high nutrient loading rates in a stratified estuarine system resulted in the predominance of picoplanktonic and flagellated forms (autotrophs) in contrast to the normal dominance of large flagellates and diatoms. Second, Oviatt et al. [1986] working in mesocosms in Rhode Island showed that high nutrient loading rates induced the most dramatic responses in the bacterial community. The highest nutrient loading rates resulted in a very large increase in bacterial numbers and activity, and water column respiration rather than benthic respiration dominated oxygen demand. Sediment input from agricultural and development practices in the Chesapeake Bay could result in similar plankton responses [Fogg 1986; Glover et al. 1986; Parsons et al.; 1986]. For example, several investigators have suggested that picoplanktonic forms such as the cyanobacteria may be better adapted to low-light conditions than other planktonic forms. Thus high turbidities of the upper Chesapeake and the light limitation documented there by Flemer [1970] and Harding et al. [1986] may be selecting for picoplanktonic forms. Similarly, as Sanders and others mentioned yesterday in summarizing the input of toxics, much work suggests that the input of metals, hydrocarbons, and other materials at sublethal concentrations results in a shift of phytoplankton species composition and sizes. The shift in size is important; sublethal concentrations of metals and toxics may cause the predominance of small cells [Greve and Parsons 1977]. Thus three factors that we know encourage the development of a microbial food web exist in the Chesapeake Bay: high nutrient loadings, high turbidity, and numerous toxic point sources.

What evidence supports the actual existence of a microbial food web in the Bay? Surveying the literature for such evidence confirms the recent advent of interest in the topic, as all of the work has been done in the last few years (Table 1). Ducklow and Tuttle have estimated bacterial densities in the Bay as 10^9-10^{10}

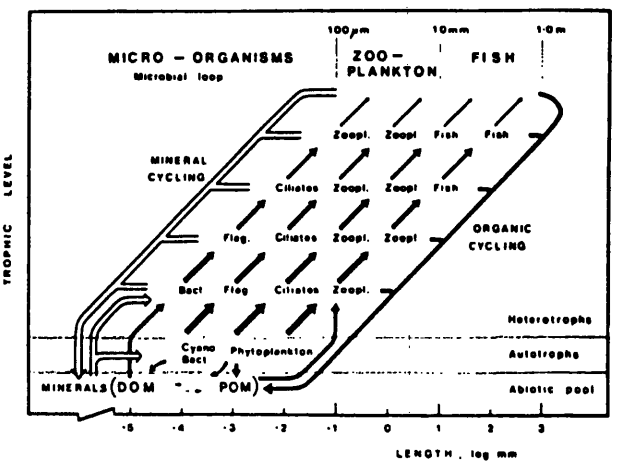

Figure 1. Semi-quantitative model of planktonic food chains. Solid arrows represent flow of energy and materials; open arrows, flow of materials alone [Azam et al. 1983].

Table 1. Evidence for a well developed microbial loop in Chesapeake Bay.

Trophic level	Density (no./l)	Area	Reference
Bacteria	10^{9-10}	Mesohaline region York River	Ducklow 1982 Malone et al. 1986 Tuttle et al. 1987a,b
Heterotrophic nanoflagellates	$1.5\text{-}6.5 \times 10^6$ $0.8\text{-}6.1 \times 10^6$	Mesohaline region York River	Dolan and Coats 1987 Ray 1986
Picoplankton	$<1\text{-}270 \times 10^6$	Turbidity max. & mesohaline reg.	Sellner & Brownlee 1985-1988
	$<1\text{-}14.6 \times 10^6$	Lower Bay	Marshall & Lacouture 1986
	$1\text{-}16.7 \times 10^6$	Hampton Roads	Marshall & Lacouture 1986
	186×10^6	York River	Ray 1986
Microzooplankton (whole-water)	$15.2\text{-}24.3 \times 10^6$	Chester & Choptank Rivers	Brownlee & Jacobs 1987
Microzooplankton (>44 µm)	$1\text{-}15 \times 10^2$	Tidal-fresh Bay	Sellner & Brownlee 1985-1988
	$<1\text{-}4 \times 10^2$	Mesohaline region	Sellner & Brownlee 1985-1988

organisms/liter [Ducklow, 1982; Malone et al., 1986; Tuttle et al., 1987a,b] probably the highest bacterial densities recorded in estuarine environments, and as high in fact as the early stages of a batch culture. For heterotrophic nanoflagellates, a limited but growing data set studied by Ray [1986] and Dolan and Coats [1987] suggested densities from 0.8 to 6.5 x 106 organisms/liter. Picoplankton densities, which probably include the ubiquitous coccoid cyanobacteria along with eukaryotic picoplanktonic forms, have been estimated to range from 1 to 270 x 10^6 cells/liter [Sellner and Brownlee, 1985-1988; Marshall and Lacouture, 1986; Figures 2 and 3). Similar concentrations were observed by Ray [1986] in the York River. These are very high densities of very small cells. High densities of microzooplankton have also been estimated

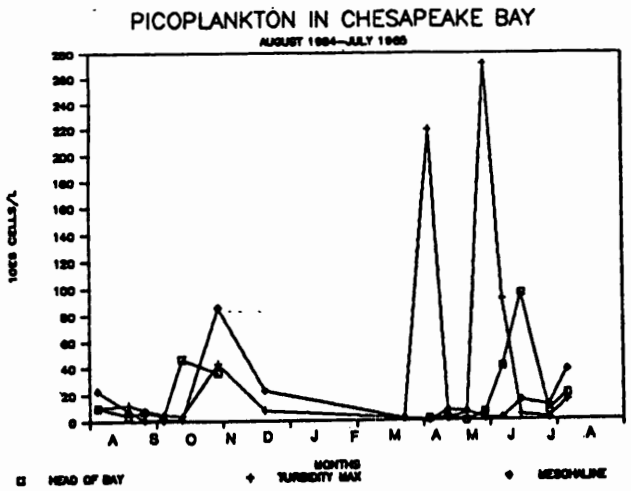

Figure 2. Picoplankton densities in the Chesapeake Bay, August 1984 — July 1985 [Sellner and Brownlee 1985-1988].

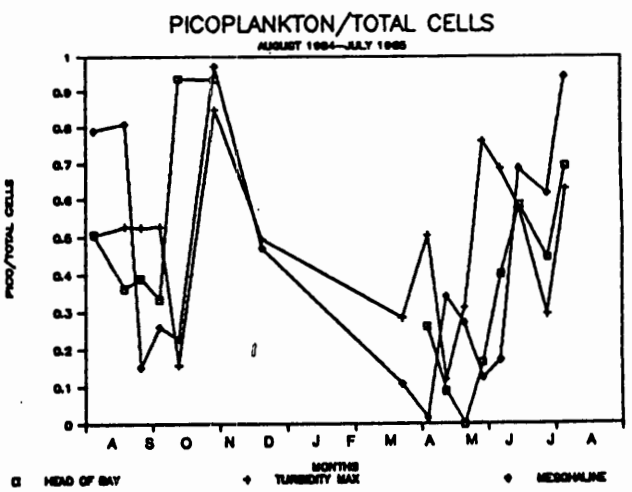

Figure 3. Picoplankton as a proportion of total cells in the Chesapeake Bay, August 1984 — July 1985 [Sellner and Brownlee 1985-1988].

Table 2. Metabolism of the microbial loop.

Trophic level	Metabolism	Area	Reference
Bacteria	30->50% (bact prod/prim prod)	Mesohaline region	Malone et al. 1986
	OX cons.=0.0045(Bact) +0.007; r=0.95	Mesohaline region	Tuttle et al. 1987a, b
Microheterotrophs	P/R < 1	Mesohaline region	Kemp & Boynton 1980 Sellner 1987
	Euphotic zone BOD via oxidation of particulates	Mesohaline region	Tuttle et al. 1987b
Microzooplankton grazing	micro/macro > 1	Mesohaline region	Brownlee et al. 1986 Sellner et al. in prep.

[Sellner and Brownlee 1985-88]. Whole-water counts show total densities of 15,000 to 24,000/liter. [Brownlee and Jacobs, 1987]. The larger microzooplankton (rotifers, tintinnids, etc.) are found at densities from <100 to 1500/liter (Figure 3).

Other indications that we may have a well-developed microbial food web come from metabolism studies (Table 2). Ducklow, working with Malone and Tuttle on tilting [Malone et al., 1986] estimated that between 30% and >50% of the primary productivity passes through the bacteria. Ducklow et al. have summarized these data for the last four years and will present more long-term data in a session this afternoon. Tuttle and co-workers [1987b] have also estimated summer oxygen consumption in the mesohaline Bay, and found that consumption in the water column is directly related to bacterial densities. This relationship suggests that measuring bacterial abundance alone could serve as an indicator of oxygen consumption rates; however, this relationship needs to be confirmed in other salinity regimes and in all seasons before this measure can be applied everywhere.

An active microbial web in the Bay is also suggested by several other data sets. For example, water column respiration off Calvert Cliffs in mesohaline Chesapeake Bay exceeded production in 18 of 23 summer months, 1975-1980 [Sellner 1987]. In addition, euphotic zone oxygen demand in mesohaline summer conditions is primarily due to the oxidation of particulate matter [Tuttle et al., 1987b] reflecting total microbial community metabolism, rather than solely the bacteria.

A comparison of microzooplankton grazing with macrozooplankton grazing indicates that the former is generally much greater than the latter [Brownlee et al., 1986]. Furthermore, recently collected data on the very small ciliates suggest that the grazing pressure during May, June, and July is primarily due to small ciliates [J. Dolan, unpublished data]. In other words, herbivory takes place primarily through these small cells.

These data indicate the presence of a strongly established microbial food web in Chesapeake Bay. One of the questions we should address is: if we can reduce loading, will we see reduced eutrophication, improved water quality, a decrease in anoxia and an increase in fish yields? A theoretical diagram from Parsons [1979] (Figure 4) suggests that we can conceive of current conditions in the Chesapeake Bay as a microautotroph community (the picoplankton and flagellates) being grazed on primarily by a protozoan food web, eventually reaching the coelenterates (discussed by Verity [1987]) as gelatinous predators. If we could reduce the loading rates, could we expect to see selection of large diatoms, large flagellates, larger copepod grazers, and ultimately pelagic fish? Can we eventually, through regulation of inputs of stresses — nutrients, suspended sediments, and toxics — shift the community in this way?

RECOMMENDATIONS

Based on presentations by Verity, me, and later, Ducklow et al., future Bay conditions might be predicted with additional support in several areas. We should continue to estimate the importance of the microbial food web. We need more information on the distribution and the activities of these very small plankton.

We need to do more experiments along the line that Sanders et al. [1987] have pursued, i.e., looking at the effects of nutrient loading and light limitation on

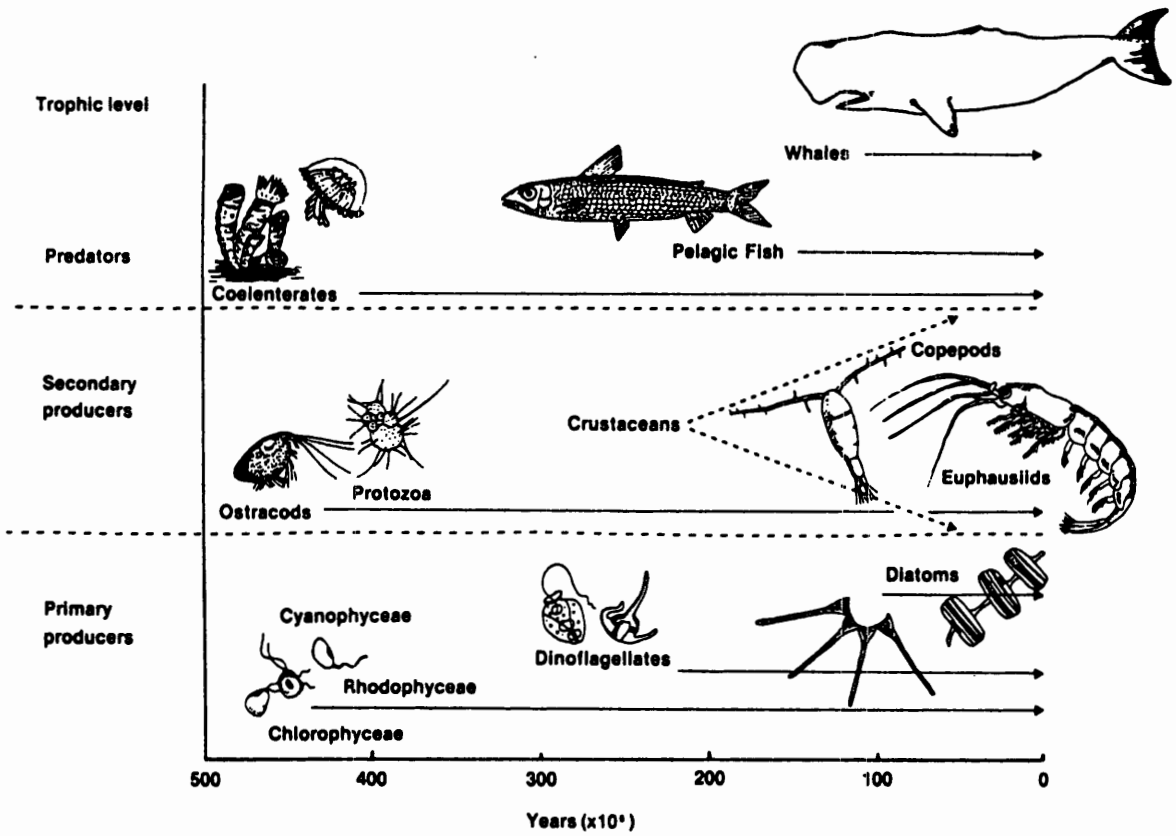

Figure 4. Theoretical diagram of evolutionary relationships between primary producers, secondary producers, and predators.

inducing changes in the microbial webs. Specifically, we should find out whether nutrient increases and light limitation cause an increase in the smaller components in a stratified system. Once the stress is removed, does the system return to diatoms and larger flagellates and copepods? Or does it shift, as Verity suggested earlier, to selection of another metazoan food web, slightly different but still passing carbon to a higher but different trophic level?

Much more work needs to be done toward establishing the relationship between bacterial densities and activities and the oxygen demand in the water column. We have very little historical data on the importance of the microbial food web. It may have been well-established all along, only becoming apparent with the development of high-resolution techniques. A possible approach to this problem would be to examine degradation products of the pigments associated with the picoplanktonic fraction under anoxic conditions. Then using cores and substrata analysis in the same manner as Brush (University of Maryland), coupled with HPLC analysis, we might be able to determine whether the dominant members of the picoplankton have been present over the last several centuries.

RELEVANT LITERATURE

Azam. F., T. Fenchel, J.G. Field, J.S. Gray, L.A. Meyer-Reil and F. Thingstad. 1983. The ecological role of water-column microbes in the sea. Mar. Ecol. Prog. Ser. 10: 257-263.

Brownlee, D.C. and F. Jacobs. 1987. Mesozooplankton and microzooplankton in the Chesapeake Bay. In: S.K. Majumdar, L.W. Hall, Jr. and H.M. Austin (eds.). Contaminant problems and management of living Chesapeake Bay resources. PA Acad. Sci., Easton, PA: p. 217-269.

Brownlee, D.C.; F. Jacobs; S.G. Brownlee, and K. G. Sellner. 1986. Relationship between nutrients and plankton in Chesapeake Bay. III. Potential roles of micro- and macrozooplankton. Abstract and presentation, ASLO, URI, 23-26 June 1986.

Dolan J.R. and D.W. Coats. 1987. Heterotrophic microflagel-

lates in the Chesapeake Bay seasonal abundances in different parts of the water column. EOS 68: 1728.

Ducklow, H.W. 1982. Chesapeake Bay nutrient and plankton dynamics. I. Bacterial biomass and production during spring tidal destratification in the York River, Virginia, estuary. Limnol. Oceanogr. 27: 651-659.

Flemer, D.A. 1970. Primary production in Chesapeake Bay. Ches. Sci. vol II. pp. 117-129.

Fogg, G.E. 1986. Light and ultraphytoplankton. Nature 319: 96-99.

Glover, H.E., M.D. Keller and R.R.L. Guillard. 1986. Light quality and oceanic ultraphytoplankters. Nature 319: 142-143.

Greve, W. and T.R. Parsons. 1977. Photosynthesis and fish production. Hypothetical effects of climate change and pollution. Helgol. wiss. Meeresunters. 30: 666-672.

Harding, L.W., B.W. Meeson and T.R. Fisher Jr. Phytoplankton production in two east coast estuaries: photosynthesis-light functions and patterns of carbon assimilation in Chesapeake and Delaware Bays. Est. Coast. Shelf. Sci. 23:773-806; 1986.

Kemp, W.M. and W. R. Boynton. 1980. Influence of biological and physical processes on dissolved oxygen dynamics in an estuarine system: implications for measurement of community metabolism. Estuarine Coastal Marine Science, vol. II, p. 401-431.

Malone, T.C., W.M. Kemp, H.W. Ducklow, W.R. Boynton, J.H. Tuttle and R.B. Jonas. 1986. Lateral variation in the production and fate of phytoplankton in a partially stratified estuary. Mar. Ecol. Prog. Ser. 32: 149-160.

Marshall, H.G. and R.V. Lacouture. 1986. Seasonal patterns of growth and composition of phytoplankton in the lower Chesapeake Bay and vicinity. Est. Coastal Shelf Sci. 23: 115-130.

Oviatt, C.A., A.A. Keller, P.A. Sampou and L.L. Beatty. 1986. Patterns of productivity during eutrophication: A mesocosm experiment. Mar. Ecol. Prog. Ser. 28: 69-80.

Parsons, T.R. 1979. Some ecological, experimental and evolutionary aspects of the upwelling system. S. Afr. J. Sci. 75: 536-540.

Parsons, T.R., P. Thompson, W. Yong, C.M. Lalli, H. Shumin and X. Huaishu. 1986. The effect of mine tailings on the production of plankton. Acta Oceanolog. Sinica 5: 417-423.

Pomeroy, L.R. 1974. The ocean's food web, a changing paradigm. Bioscience 24: 499-504.

Ray, R.T. 1986. The role of picoplankton in phytoplankton dynamics of a temperate coastal plain estuary. M.S. Thesis, VIMS, Gloucester Pt., VA. 85 pp.

Sanders, J. G., S. J. Cobik, C. F. D'Elia, and W. R. Boynton. 1987. Nutrient enrichment studies in a coastal plain estuary: changes in phytoplankton species composition. Can. J. Fish. Aquat. Sci. 44:83-90.

Sellner, K.G. 1987. Phytoplankton in the Chesapeake Bay: Role in carbon, oxygen and nutrient dynamics. In: S.K. Majumdar, L.W. Hall, Jr. and H.M. Austin (eds.). Contaminant problems and management of living Chesapeake Bay resources. PA Acad. Sci., Easton, PA: p. 134-157.

Sellner, K.G. and D.C. Brownlee. 1985-1988. Maryland Department of the Environment. Chesapeake Bay water quality monitoring program. Phytoplankton and microzooplankton component, data summary. The Acad. Natl. Sci., Benedict Est. Res. Lab., Benedict, MD.

Smayda, T.J. 1983. The phytoplankton of estuaries. In: B.H. Ketchum (ed.). Estuaries and enclosed seas. Elsevier, Amsterdam: p. 65-102.

Tuttle, J.H., R.B. Jonas and T.C. Malone. 1987a. Origin, development and significance of Chesapeake Bay anoxia. In: S.K. Majumdar, L.W. Hall, Jr. and H.M. Austin (eds.). Contaminant problems and management of living Chesapeake Bay resources. PA Acad. Sci., Easton, PA: p. 442-472.

Tuttle, J.H., T.C. Malone, R.B. Jonas, H.W. Ducklow and D.C. Cargo. 1987b. Nutrient-dissolved oxygen dynamics in Chesapeake Bay: The roles of phytoplankton and microheterotrophs under summer conditions, 1985. U.S. EPA, CBP/TRS 3/87, 158 pp.

Verity, P.G. 1987. Factors driving changes in the pelagic trophic structure of estuaries, with implications for the Chesapeake Bay. In: Perspectives on the Chesapeake Bay: Recent advances in estuarine sciences. U.S. EPA, CBP/TRS 16/87, CRC publication no. 127: p. 35-56.

Williams, P.J. leB. 1981. Incorporation of microheterotrophic processes into the classical paradigm of the planktonic food web. Kieler Meeresforsch. Sondh. 5: 1-28.

CONCLUDING SUMMARIES AND PANEL DISCUSSION

"Where do we go from here?"

— Summaries of concurrent sessions, identifying major research questions and associated costs

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The Johns Hopkins University

Toxics
Nicholas Fendinger
Agricultural Research Service
U. S. Department of Agriculture

Nutrients
Carl Cerco
U. S. Army Corps of Engineers

"How do we get there? Who pays the fare?"

— Panel discussion with members from federal and state funding agencies, scientific institutions, and private organizations

Panel Members:

Monica Healy
Office of Governor
State of Maryland

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"Where do we go from here?"

Joseph A. Mihursky, moderator

LIVING RESOURCES: Ed Houde

The perspective of this conference on living resources has been a broad one, including all of the living organisms in the Bay, as well as factors that influence their well-being, abundance, and changes in abundance. The Chesapeake Bay is a life support system, and we have looked at living resources from an ecosystem perspective. We may not have given enough attention to non-harvestable living resources in the past. We are making a good start at understanding the Bay ecosystem, and are beginning strategic planning to continue expanding our knowledge. This conference has been a little disappointing in its lack of coverage of the top levels, that is, the fishes; but why the people doing this research did not join us here is hard to say.

In speaking about benthic-pelagic coupling, Walter Boynton mentioned how thin a layer of water overlies the sediments of the Chesapeake and suggested that this interface of sediments with the water column is an important area of biological activity. A lot of the dynamics of the Bay is associated with this interface, and progress has been made in understanding its relation to nutrient dynamics and associated production processes. We have tracked the seasonality of many of the factors associated with this interface that influence nutrient flux and primary production. This research is not specifically organism-oriented; there is more interest in the total system and its ability to produce. Clearly the anoxia problem, with its association with sediment oxygen demand, has an important impact on the living resources.

The presentations on pelagic trophic structure, in contrast to the data-oriented presentations on benthic-pelagic structure, were related to ideas and hypotheses. Our understanding of the Bay in this area is still limited. Roman, White, and Brownlee have reported some results on grazing potential and abundance of organisms in the water column, but these results are only a start. The question of top-down vs bottom-up control is important, since we know that nutrients have increased at the bottom, and many of the top predators have been removed as well. It was suggested that the removal of a very large biomass of oysters, relatively in the middle of the trophic structure, has had a significant effect on that structure and consequent flow pathways of energy and material.

The basic production mechanisms in the Bay may have changed. The bacterial food web may have become more important than the traditional metazooan food webs, and may now dominate; in any case it certainly appears to have gained in importance. A question unanswered thus far is how important it was in the past; it may have been important but unstudied.

It is possible that the food pathways may have shifted toward jellyfish, away from the metazooan web, and that this may have contributed to the decline in harvestable living resources.

The recruitment process has not been much discussed here, although these episodic events do influence populations. The blue crab work at VIMS, which is highly resolved in time and space, is a good example of the kind of work we need to do to understand the recruitment process and how it affects the abundance of harvestable resources. We need to take an ecosystem approach as well as investigating these specific questions of recruitment. The recruitment process is highly variable, and is influenced by two major factors: adult stocks and environmental conditions in a specific year. It seems now that the environment may be a more important factor than adult stocks. We also need traditional kinds of fisheries research, as well as single-species and multi-species modeling efforts. It is worth repeating here a recommendation that was made at a Sea Grant conference on fisheries in February: that fisheries research should concentrate on the first 100 days of life. It is here that the physics of the system, the available food, the productivity of the Bay, prey relationships, the potential for disease and contaminants, and the role of habitat all are most important.

As far a research needs, there is a critical need for strategic planning. This process is beginning, through STAC, the Monitoring Committee, and the Living Resources Committee. We need time-series analyses, with monitoring; the appropriate time and space scales for monitoring are still open to question. We need to integrate Chesapeake Bay system research better with fishery research.

Restoration strategies are critical, not only for SAVs, but for other resources such as striped bass and other anadromous fishes. We need to learn how to bring these resources back to their former levels.

It is hard to estimate the costs of the research that is needed, except to say that they will be large. Restoring or stabilizing the resources of the Chesapeake Bay is worth doing, but it won't come cheap. One way to carry out research cost-effectively, an intermediate option between a small tank in the laboratory and the whole Bay, is the mesocosm approach. This approach should be explored and used more fully.

DISSOLVED OXYGEN: Eugene Cronin

This conference has shown much evidence of the progress being made in this field. Interest is broad, and some efforts have been interdisciplinary. Sea Grant programs have selected dissolved oxygen as an area of exceptional importance and have supported it over a number of years. The ten papers presented here do not adequately represent what we know or what is being worked on, so it is difficult to summarize on the basis of these papers alone. Some of the needs in anoxia-related research were suggested by papers given in other concurrent sessions: on the causes and degree of vertical stratification in the Bay; the effect of anoxia on the availability of toxics in sediments; the chemical state of inorganic toxicants; the absence or presence of oxygen related to sediment flux. Where and with what intensity is deoxygenation happening? What processes are causing it? What are the ultimate chemical and biological effects? What management is possible? What can and what cannot be controlled?

As sources of oxygen consumption, the microbiota appear to be growing in importance, accounting for perhaps as much as 30-40% of the oxygen demand. In responses to nutrients some obvious possible linkages are seen, but much is yet unclear. Modeling is still useful, as it breaks the world down into understandable pieces. The possibility was raised here that the 99% removal of oysters has contributed significantly to anoxia, as the filtration capacity of this species has been lost without this control on phytoplankton crops, they have increased in response to added nutrients, and the decay of the large phytoplankton biomass may be contributing to the oxygen demand processes.

The biological effects of anoxia have been studied in the goby; this species is not very mobile and can give some idea of the biological effects in a spatially limited area.

The significance of anoxia has been debated. One presenter contends that anoxia has not increased over time, but that the changes observed have been largely a result of spring flows — the most important element, especially in the upper part of the system. One problem may be a mismatch between the predators and the location of the food supply.

Modeling continues to be a vital tool, serving several functions. The importance of verification must not be underestimated.

This conference has been important for its interdisciplinary discussions. We must become predictive, we must develop hypotheses, we must go to the field and test them, then discuss our results and fight it out. People must be aware that we are in a hydrodynamically fluctuating system.

In discerning what research needs to be done, the most effective approach may be to ask experts what research should be funded outside of their field.

SUBMERGED AQUATIC VEGETATION: Robert Orth

Submerged aquatic vegetation gives us a Bay-wide perspective, a unique blend of the scientist, citizen, and manager in action. In 1978 little was known about SAVs. Since then there has been a tremendous change in the attitudes of scientists and managers, and citizens have joined the effort by monitoring. The importance of this citizen involvement should not be underestimated. For instance, the citizens' perceptions of hydrilla have changed, and with the change has come a different approach to control. The chemical approach that led to application of 2,4D to milfoil 30 years ago has given way to mechanical control.

Distribution and abundance: a decade of change. The first task ten years ago was to find out the current distribution. Mapping surveys of SAVs have been conducted annually since 1978. No major shifts have been found; most of the SAV is still in the lower Bay. We still do not know what the distribution was in the 1960s. It is possible that areas now denuded were choked with vegeation then. For the last 10 years snapshot pictures have been used to study local changes in abundance. Now it is possible to overlay the distribution data with water quality data. A resurgence is evident in the York, the Rappahannock, and the mid-Bay area, probably due to climatic changes. The declines came from the upper Bay to the lower, and from upriver to downriver. The increases now being observed are primarily in the vicinity of existing beds.

Water quality criteria. Major monitoring efforts in the York and Choptank Rivers will contribute to the

development of water quality criteria for SAV. As the SAV are immobile, relatively small shifts in water quality can make big differences in SAV growth. This sensitivity has implications for water quality management efforts.

Natural resource value. To clarify the natural resource value of SAVs, physical oceanographers and blue crab specialists need to pool their efforts. SAVs have a role in the early life stages of the blue crab, but the relationship has not been fully described.

Restoration. Restoration of SAVs is important because many appropriate areas are too far from existing areas for the natural spread to revegetate them. Most of the test programs are using whole-plant material; these programs need information on plant spacing, use of fertilizer, and ideal plant size. The latest effort is to use seed processes, as these are less labor-intensive. Transplanting is an important tool for understanding relationships between plants and the environment. Generally transplanting is not feasible in the upriver historical sites far removed from existing beds, because the cost of moving the plant material is too great. The difficulty in guaranteeing the success of SAV makes mitigation a risky proposition. Emphasis should be on protection, not mitigation. Studies should continue with propagules such as seeds and with early successional species and mixed plantings.

Monitoring. The monitoring program should continue, with annual photography so we continue to acquire data even if it is not all mapped. This information has been valuable to managers in assessing the impact of development. The lack of historical data has been a big obstacle to us; knowing this we should not fail to acquire data now for the future. We need a better link between investigations in the upper Bay and those in the lower Bay. Microcosm experiments are needed to help clarify the effect of suspended sediments. We must take a broader view to understand the relationships between blue crabs and their habitat. We need to continue testing restoration methods, considering what effect population pressure may have on the future of SAV in the 21st century.

TOXICS: Nicholas Fendinger

The area of toxics is extremely complex, involving the physical properties of the Bay, physicochemical properties of toxic compounds, and complex interactions with the biota of the Bay. Toxicants entering the water from agricultural and industrial uses, from spills, and from fugitive emissions will be distributed throughout the Bay, and their distribution will be determined in part by the physicochemical characteristics of the compounds. Once distributed, toxic materials will both affect and be affected by the biota. Three areas have been addressed here: source assessment, dynamics of pollutants in the environment, and the interaction of pollutants with biota.

Source assessment. Papers here reported that about a quarter of the 65 Department of Defense installations in the Bay watershed had significant effects on water quality, but most of the impacts were fairly local. Data were also presented on levels of toxic materials in effluent from commercial shipyards, and on their levels of toxicity.

Dynamics. Toxicants accumulate in the microlayer, possibly affecting blue crab larvae and other living resources. Microlayer concentrations can reach 10-1000 times the concentration in the bulk water underneath.

Photochemical oxidants, such as hydrogen peroxide and ozone, in Bay water can also influence the behavior and toxicity of chlorine and chromium in the Bay. The question was raised whether dechlorination does any good, or, conversely, any harm. Evidence on its benefits is mixed. As far as harm, the conversion of sulfite to sulfate has no toxic intermediates, but some copper is released in the process.

Relation to biota. Body burdens of PAHs and PCBs in blue crabs appear to be unrelated to the areas where the crabs are sampled. This lack of relationship may refect the blue crab's mobility through its life history; or it may be due to the shedding of the exoskeleton.

Nutrients are widely considered to be the principal culprit in the decline of the SAV. However, toxics may also be important contributors to their decline. Data were presented here on a bioassay for the phototoxicity of contaminants on SAV. This toxicity test could be used to establish water quality criteria for SAV, and effluents could then be tested for SAV toxicity.

Induceable adaptations have been used in bioassays, specifically in a stress protein immunodetection assay. This assay gives a uniform response regardless of whether stress is caused by temperature, oxidants, or chemicals, and it is inexpensive and easy to perform. It may have broad applicability for testing in the Bay.

Toxic effects on fish include four types of deformations of gill tissue: blood clots, enlargement of cells, abnormal cell numbers, and broken filaments with altered growth patterns. Although the deformities obviously impair the health of the organisms, it is hard to judge their significance at population levels.

A rapid toxicity test for sediments presented here may prove to be a reliable test for sediment toxicity; another useful test is behaviorally based and utilizes the burial rates of clams.

Important areas of toxics not covered here include the important research on the physical dynamics of pollutant transport, understanding of the basic chemical properties of compounds, the importance of sediments and determination of whether they serve as a source or as a sink, and impacts of toxic materials on growth, reproduction, immunology, and genetics of Bay organisms. In addition, the ability to address toxic effects on populations and communities is in a primitive state.

PHYSICAL PROCESSES: Eric Itsweire and Owen Phillips

This summary has two parts: what we know about circulation, and what we need to know.

Models. Local models of circulation seem to be successful. For instance, the circulation model for Hampton Roads developed by the Virginia Institute of Marine Science is a good blend of field observations, simulations, and simple theoretical and numerical analysis. Models of Baltimore Harbor and the Patuxent River are also useful for prediction of general features, but only for average characteristics with an arbitrary choice of parameters. The model of an estuarine plume developed by Horn Point Environmental Laboratory can reproduce basic behavior with simple geometry. The global model developed by NOAA has not been very successful, and it is unclear whether it can be tuned for greater usefulness.

Remote sensing. NOAA research shows Bay-wide patterns of near-surface sediments in response to high runoff, but includes no quantitative estimates of sediment transport. Acoustic and thermistor chain measurements made by Johns Hopkins scientists show transient mixing events and internal waves at the pycnocline, but a possible bias in the sampling must be considered. Measurements of dissolved oxygen and salinity by HPEL and the Chesapeake Biological Laboratory show a negative correlation between DO and salinity at the Choptank-Patuxent transect. Efforts are being made to use physical data to formulate managerial decisions.

Future prospects. The work of Pritchard has provided a base to build on, but we now realize that the impact of transient or sporadic events is still poorly understood. There is a need to integrate remote sensing from acoustic, airborne, and satellite sources with high-resolution in-situ instrumentation in prioritized, well-planned, well-directed studies. In particular, studies should address frontogenesis and frontal evolution, turbulent mixing, and the spatial-temporal response to transient weather events. The corollary to this effort should be an intensive investment in modern equipment for simultaneous observations.

Models need to be developed for description of transient and local events and the reestablishment of water characteristics thereafter; these models should be closely tied to the observational program.

The multidisciplinary approach should be combined with a careful scrutiny of what long-term studies would provide useful scientific and managerial information.

Critical but rapid peer evaluation of the results of research is necessary, with dissemination of the results in clear, nontechnical language to both the biological community and managers and legislators. Reviewed, technical literature is an inefficient means of communication beyond the physical oceanography community. It is not enough for communication to the larger audience to be reliable and complete; it must first of all be comprehensible.

NUTRIENTS: Carl Cerco

Rather than summarizing all the work that has been presented here, I will mention several papers that struck me with the greatest impact. First, the groundwater flow of nutrients into Chincoteague Bay is being measured, and the flow is very substantial. Yet very little is being done in this field. Considering how important nutrients are and how little is being done, groundwater flow is a research area that we need to expand in the Chesapeake system.

Second, the importance of microecology is growing. Nutrient cycling and respiration need more study in bacteria, picoplankton, and microzooplankton. We need more than applied ecology and in situ measurements; we need fundamental research. The picoplankton, and microzooplankton have not been well enough studied generally.

Third, sediments are a hot topic. The whole question of nitrification/denitrification is of great importance, as denitrification is the only exit from the sediments for nitrate nitrogen. We need far greater coverage of the Bay with sediment traps; the trap about to be placed is only the second one in the entire Bay. There is no substitute for actual measurements.

Finally, there is a great need for more spatial resolution in data from the lower Bay and the lower tributaries.

"How do we get there? Who pays the fare?"

Joseph A. Mihursky, moderator

REMARKS BY MONICA HEALY

The great strides EPA, the other federal agencies involved in the Bay clean-up program, and the other three Bay states have made in recent years cannot be exaggerated. Last December they took a giant step forward by signing a new Chesapeake Bay Agreement to guide us into the next century. I won't go into the details of the document because I know other speakers have already done a thorough job of that. However, I will say that the agreement may prove to be the most significant step in the history of the Bay clean-up program. It sets tough new pollution control standards for the entire Bay.

This new document gives us reason for celebration, but it must be tempered with a measure of sobriety. Government officials will have to bite the bullet and infuse billions of dollars into the clean-up effort. They will be required to make the difficult choices between competing interests to guarantee the Bay's vitality.

Assuring the flow of federal funds will not be easy, since the administration and Congress are trying to grapple with the stark realities of our fiscal crisis. There have been major cutbacks in federal programs in an effort to reduce the deficit. Because many sources of federal funds for environmental programs are simply drying up, states and localities have to carry more of the burden. The situation will get worse before it gets better.

The shrinking federal pie will surely reduce the amount of overall federal funding flowing into estuaries over the next several years. However, we have a great deal going for us in our fight to get a pretty good slice of that smaller pie.

We have, first of all, a highly popular and highly visible campaign to clean up the Bay. The Bay's close proximity to Washington makes members of Congress much more sensitive to its problems, and much more aware of what the states are doing to address them.

We also have a good case on the merits. There is a demonstrated need — a $27 million EPA study has documented that our nation's largest and most productive estuary is dying.

Moreover, we have tremendous political clout in Congress. The House members and Senators who represent the Bay states have formed the Chesapeake Bay Congressional Caucus. They are a potent political force in Congress and have been very successful in the past in capturing federal funds and passing legislation to further our efforts. Furthermore, it certainly doesn't hurt that many members of Congress have direct, personal connections to the Bay. When President Reagan singled out the Bay during his 1984 State of the Union Message to a joint session of Congress, a reporter asked a friend of mine why there was such a big round of applause for such a parochial concern. "Simple," my friend told the reporter, "they all have boats on the Bay."

Finally, the lessons of the Bay clean-up program will serve as a useful model for similar campaigns for other estuaries around the country. Thus any investment in the Chesapeake will reap benefits far larger than a healthier Bay.

In sum, it's a good news/bad news scenario for the Bay in terms of prospects for future federal funding.

The bad news is there is limited funding available for programs to aid the clean-up effort. The good news is that we have all the right ingredients to give us a leg up on other states who will be competing for the same money. If we use those ingredients to our advantage, if we work even harder at our lobbying efforts — and that means grassroots support from all of you — and if we continue to work as a team, I'm optimistic that we can be successful in obtaining the federal monies and programs needed to return the Bay to a productive, healthy state.

You're probably wondering how can you help further our cause on Capitol Hill. You can and should be an integral part of our grassroots lobbying efforts. You have done a tremendous job in producing the scientific evidence that has far-reaching ramifications for the future of the Chesapeake Bay, and that research can make a real difference during the decision- making processes on Capitol Hill.

But the research is of little use in Congress unless policymakers can understand its value in practical terms. To be blunt, you have to spoon- feed them. First, politicians and their staffs often have little or no scientific background, so it's important for you to translate what you're doing into the simplest possible terms. For instance, many don't even understand basic terms like "non-point source pollution." It helps to

explain that in language they can understand. Second, you have to show them how your research translates into tangible action for the Bay. Often, scientists come to the Hill and talk about the wonderful research they're doing, but what's going on in politicians' minds when they listen to that is, what good is it for the Bay? For example, you have to explain how scientific research has led to specific standards for discharge permits. Or you could tell them how research led to the ban on TBT.

If you explain science in plain language and show politicians how what you do translates into tangible action, I can guarantee that federal research funds will continue to flow — as they should.

REMARKS BY CHRIS D'ELIA

I am the Program Director of the Biological Oceanography Program at the National Science Foundation, and I have been very appreciative of the efforts Ed Houde made while in the same position several years ago. His work on strategic planning has helped make NSF more active in the integration of future research. Within the Ocean Sciences Division there are four research programs: biological oceanography, chemical oceanography, marine geology and geophysics, and physical oceanography. Out of a total budget of roughly $140 million this fiscal year, the strict research programs garner about half the funds. The other half goes to the Ocean Centers and Facilities Support, which pays for the ships and other infrastructure necessary for ocean research.

The budget in Biological Oceanography this year is about $15 million, of which about $3 million goes into research in the estuarine and coastal areas. Our budget this year did not do as well as we had hoped. Before the stock market aberration in October we had about a 19% increase, but what we actually received in the omnibus appropriation act was about a 3% increase, which means we are not quite keeping up with inflation.

To make the best use of these limited funds we are trying to plan more actively than in the past. The NSF works on a peer-reviewed system in which an individual investigator with an idea puts together a proposal that then goes out for peer review. This system is good at turning the best ideas into meaningful long-term output. The problem is that it is very bad for strategic planning and for directing research in a particular area. Thus we are trying to develop budget initiatives, specific areas we feel need to be stressed more, and we are asking the scientific community to tell us what needs to be done in these areas.

Our primary efforts currently are involved with the Global Ocean Flux Study (GOFS), which deals with the large issues of global climate, the CO_2 effect and the associated problems. Specifically there is interest in transport of carbon to the deepsea sediment reserves of carbon, the overall impact of this process, and the various trophic pathways involved. Another initiative is a study of recruitment: where do organisms come from and how do they interact? The biotechnology initiative in Biological Oceanography has focused particularly on molecular biology and genetic approaches to understanding oceanic processes and ecology. Ridge-crest processes, which involves the study of deep-sea vents and rifts, has a small share for us.

The most important initiative to us here is the land-sea initiative. We need to have the scientific community be very active in developing a land-sea initiative that will be a specific budget line item in the NSF budget. A small amount is available in the next request, and we have a small program jointly with another NSF division, the Ecosystems Dynamics Program — the land margin ecosystem research initiative. This initiative has about $700,000 available this year, which obviously will not go far in the long run. We are trying to develop a significant enhancement to the NSF budget that will involve a lot of community effort in putting together planning documents and stating what the needs are. I hope to see substantial results from this effort within five years, because we understand from top management in NSF that we should not expect enhancements to our core program; we need to talk about more integrated programs. Congress needs to see specific items when we talk about budget enhancements.

REMARKS BY JOHN W. DANIEL

Earlier this month we funded $55 million for Chesapeake Bay programs. A large percentage of this money is direct contribution to the scientific community. An observer from the political arena such as myself who may not understand exactly what you scientists do can either ignore the intricacies of your research, or develop a healthy respect for it. I hope I have developed a healthy respect for it.

We have come to a point in our Chesapeake Bay programs that the level of scientific contribution to policy and programs has become more important, with a prospect of increasing importance in the future. Some

years ago, for instance, the relationships between open privies near streams and tributaries, degraded water quality, and the negative impact on shellfish were evident — to scientists and to non-scientists as well. But contrast this to the absolute essential scientific base necessary to draft and approve and implement a prohibition on the use of tributyltin paints. There is a substantial difference. Without that scientific basis and the capacity to perform the research, that particular action by states bordering on the Bay to improve water quality simply could not have occurred.

Thus in 1988 there is a much greater acceptance in the non-scientific community than there would have been in the past for a scientific conference with very technical topics, and there is a willingness to address the economics necessary to accomplish that research. Policy decisions and regulations must be well founded on scientific fact. To be successful, regulatory programs must be legally defensible, and that can only be achieved with a sound scientific basis.

All of this is expensive, and who will pay the fare? It is incumbent on those of us responsible for policy decisions, especially at the state levels, to coordinate scientific research with our regulatory programs. It seems to me that, as with TBT, the link between the science and the utilitarian value of that research to programmatic functions is a tremendous catalyst to helping you pay the cost. By virtue of the Bay Agreement, for instance, the states are obliged to move forward with toxic control programs in our respective water control agencies. The solutions to those concerns will be individualized to the jurisdictions, and they will not be identical. They will be alike, however, in requiring a sound scientific basis to be legally defensible. If we don't do this, we move backward rather than forward.

So who pays? The easy answer is that we all pay. Federal, state, and local governments pay, the regulated community pays through capital improvements and technological investment, and individuals pay with changed habits and investments of time and energy. The private sector and foundations pay with grant funding and charitable contributions.

We need, however, to maximize what is produced with these payments. This necessitates cooperation rather than competition, and the beneficiary of these efforts must be not individuals but the Chesapeake Bay. If we can show the regional and Bay-wide value of a dollar spent in any individual jurisdiction and demonstrate to those who pay how that science relates to useful programs, we benefit the understanding of the estuary and we demonstrate the necessity of continued funding for advances in Chesapeake Bay research.

REMARKS BY FRANK PERKINS

We are working in an exciting field, one that is appealing to the general public. There are individuals who have the resources and ability to fund our work at a very high level. We are able to make a presentation to a potential donor that is fascinating, exciting, fun, and shows the practical value of our work. We have a lot going for us; and we can look to the Cousteau model for effective presentation of exciting material. The creatures of the Bay are just as exciting as those in the open ocean highlighted by Cousteau.

One of the things that corporations respond to readily is the need for superior instrumentation. An effective selling point is that an institution can compete much better for federal funding when the expense of instrumentation does not have to be added to proposal budgets. As an aside, I might note that a number of federal agencies have excellent shops and fabrication plants (NASA comes to mind right away), and we as marine scientists need to establish relationships with those agencies because a lot of the instrumentation we need can be constructed de novo. When we approach private donors, especially individuals, we must make them aware of what we are doing and try to involve them to some extent in day-by-day operations. Obviously this kind of thing if overdone could be a real burden, but done properly, the hands-on experience in the living world can be a powerful aid in persuading a potential donor to support our work.

Private foundations seem to be orienting themselves increasingly toward public policy and resource management, leaving it to the federal agencies to take care of the basic research in biological and physical sciences; but the potential of private foundations should not be dismissed.

User fees, which have already been mentioned, may be an overlooked source of funds.

As we develop programs for approaching the private sector we must accept the necessity of diverting some funds into the use of video to capture the imagination of the public. This is hard when the money is diverted from research, but video is a tremendous tool and we should be making better use of it.

More specifically, we must be concerned not only about the health of the Bay overall, but also about the health of organisms per se. We need a strong branch analogous to medicine or veterinary science, to deal with this. And this kind of work should be very easy to find support for among private donors.

State funding is being covered by other speakers, but one obvious point that can use emphasis is that when you go to the legislature or the executive branch, you should be prepared to propose something that has a short-term payoff. We know enough about the system that in some areas we can offer a proposal with a high probability of success — oyster culture comes to mind. This kind of proposal is appealing to those who have to justify their appropriation of funds to constituents expecting results.

REMARKS BY IAN MORRIS

The remarks to follow reflect a personal bias and prejudice. They have not been subjected to the usual rigor of literature comparisons for any originality or to peer review for any intellectual validity. They should be read with these apologia clearly in mind.

At the end of this conference, I want to make three brief comments. Firstly, I would like to congratulate the organizers on creating a highly successful gathering, and to highlight one of the most prominent features of the past two and a half days. I do this, not because this prominent feature is hidden to all others here (it has been a feature of much of the coffee discussion), but more because it will lead me into the two other points I wish to discuss. Rarely, if ever, does one see at a technical conference such as this a mix of scientists, government officials, and interested citizens such as we have seen here. It is one of the most intriguing features of recent and current activities in the Chesapeake Bay region. Scientists want their work to matter to those charged with making decisions on the future of the Bay. These decision-makers want to hear the scientists and the citizens. Citizens want to learn of the latest technical information and thinking, and to influence further studies and future decisions. All of this is a major feature of the Chesapeake Bay today. There is no better example of it than in the attendance at this conference. It might be argued, too, that any continued success in addressing the "problems of the Bay" will depend on this closeness of interaction between scientists, citizens, and government. Yet the nature of that interaction must be right, with each segment playing its *appropriate* role.

If the above comments are true, I would like to suggest that they have some profound implications for all of us. Here, I want to address only two. The first concerns what can and should be expected from the scientific and technical community. The second concerns the need for truly transdisciplinary input of information into any decision-making process about an ecosystem and its natural resources. I shall try to make my remarks relate to this conference and so be slightly relevant to the title of this final session.

Much (but not all!) of the research presented at this conference is supported by regulatory agencies; that is, by agencies whose primary responsibility is to recommend the passage of certain laws and to enforce them, so as to lead to improved management of an important natural resource such as the Chesapeake Bay. Inevitably, such agencies need research which guides their decision-making process and which supports them once decisions are made. Under conditions of limited resources and of rigorous accountability in the expenditures of public funds this need becomes even greater. Thus, such agencies are severely constrained so that support for any *specific* piece of research must be justified by pointing to a direct link between it and a *specific* decision or action. Similarly, a scientist who lives in the real world knows that support for a piece of research will depend on arguing for such a direct link. Yet there seem to me to be real dangers in arguing such immediacy and directness between specific research and specific management actions.

On other occasions, I have emphasized the dangers to the scientist, particularly the younger one, where such constraints can severely affect the creativity of a particular study. Here, however, I wish to make a different point. It may be that we in the environmental scientific community cannot deliver this directness and immediacy of connectedness between specific research and specific management needs. It may be, too, that the sponsoring management agencies are misled in asking for such direct linkages. Rather, we might argue, the purpose of research on a particular ecosystem or environmental problem is to weave a tapestry of knowledge which would provide a framework of understanding, within which specific management actions are made. Perhaps, too, we should recognize any particular decision or action improved because of such a framework and not because of any specific research. It would be revealing to analyze the historical record of management decisions on the Bay within this context. It may be that past decisions have benefited more from our increased understanding gained over years of study and not from a specific piece of research. If this is true, it seems to me that it has profound implications for the way in which regulatory agencies can benefit from, and therefore should sponsor, research.

My second point concerns the crucial interdisciplinary nature of research needed to provide the framework of understanding mentioned above. We often

stress the need for interdisciplinary studies by pointing to the desirability of physicists, chemists and biologists working together. But I think this is missing the point. The system of the Chesapeake Bay is much bigger than the water, the sediments, and the aquatic living resources. In the Chesapeake Bay the ratio of watershed area to water volume is very high. Thus it is crucial that we expand our horizons beyond the edge of the water and go into the vast watershed. When we do, we shall realize that the Chesapeake Bay system includes the land use, the demography of population, the culture, the history, the economics, and the politics. Decision-making must be done against a background of awareness of this system "as a whole." We should be speaking of an "information-management" interface and not the narrow "science-management" interface. And the information needs to cut across disciplines.

REMARKS BY DAVID CHALLINOR

As manager of a large research budget one of the things I have to do is find funds to support it, and this can require a lot of imagination as well as hustle. For example, the Smithsonian was given a part of a group of islands on the Eastern Shore that about 200 years ago formed one large island of probably 1000 acres. For the last 200 years this island has been disintegrating rapidly, and now there are half a dozen small islands that erode at least 10 feet a year. The Smithsonian was under a great deal of pressure to "do something" about this erosion, particularly because these islands harbored one of the largest great blue heron rookeries on the Chesapeake Bay. We spent more than $100,000 trying to put in bulkheads, which lasted about two years. Finally in a very wise move the Smithsonian sold those islands. In 20 years they may well be gone.

While we had the islands, however, it was important to try to figure out what we should do. One suggestion was to build them up, reversing nature's process. A large company proposed to use the islands as a repository for plastic-baled rubble from the redevelopment of Baltimore and Philadelphia. Biologists were opposed, fearing among other things the release of toxic metals. The developer painted a rosy picture of how the rubble would be brought in by barge and used to create a lovely hilly island covered with blue heron rookeries.

We were able to get a private foundation to give us some money to find out just how practical these ideas were. It was found that the likelihood of punctured bales leaking toxic materials was fairly high, and it became clear that although the project might solve someone's waste disposal problem, it was unsound from an environmental point of view. In any case, we were able to get funding from the company that wanted to dispose of the waste and also from a private foundation that was simply intrigued by a rather interesting idea. We at the Smithsonian put in a little of our own money as well; companies and foundations are more likely to feel comfortable spending their money on your project if you show them that you are willing to spend your own on it.

So I see two things as useful: first, put some of your own money into the project to make it more "believable;" and second, somehow lure the key characters into the lab or onto the vessel where their excitement and interest gives you a high likelihood of success.

REMARKS BY SHELDON SAMUELS

Science requires public support, but to gain it does not require that we respond to the immediate or even the long-range needs of society. The practical demands for understanding our urgent needs in the environment or other issues are driving forces in every field. Science is also driven by technology. We are, as Bergson perceived, *Homo faber*, man the tool-maker. But we also remain *Homo sapiens*, man the seeker of knowledge. Sadly, *Homo sapiens* is more often forgotten than remembered. The changes in federal disbursements traditionally earmarked for science are an example. Funds are being diverted from science to the regulatory agenda of our federal government at an unconscionable rate. It is not only unwise, it is dangerous to place the future of science in the hands of the regulator.

An example from personal experience began in 1979, when the federal government decided to regulate benzene as a carcinogen in the workplace. The petroleum industry challenged this and brought it to the Supreme Court. My organization challenged the petroleum industry. The Supreme Court's decision was one I did not quite understand. I understood risk assessment, and in fact have written several peer-reviewed papers on quantitative risk assessment, but I did not understand what the Supreme Court meant when it said that priorities for regulation ought to be based on risk assessment. Kyler Hammond, senior vice-president of the American Cancer Society and the

leader at the time in the field of risk assessment, advised me, when I asked about this requirement, that I should not worry, that the back of an envelope and a pencil stub would be adequate. By 1984, however, they were not enough. Mr. Ruckelshaus gave his famous speech at the National Academy of Sciences on how use of risk assessment would speed up the regulatory process. In reviewing a draft of his speech I told him that this would not speed up regulation.

Four years later the federal government is now spending $50 million a year on quantitative risk assessment research, and regulation is not speeded up. And $5 million of this money came from the National Science Foundation, diverted directly from basic research. Risk assessment cannot and will not speed up regulation, and it is a perfect example of how funds are unwisely diverted into the hands of people who have some immediate political needs, who reshape the science itself. *Homo sapiens* has not made his case.

In 1944 a famous BBC broadcast involved an interview between the nuclear physicist Polanyi and the philosopher Bertrand Russell, whose principles of mathematics formed part of the basis for quantum physics. The discussion was on nuclear energy research: was the work begun with an application in mind? Would there be a practical result? Both Polanyi and Russell said no. Thirty days later the bomb dropped on Hiroshima.

There are two lessons here. One, there was no way to forecast that there would be an application. Two, there was long-term support from the public for that kind of research. Remember that Einstein's paper on special relativity was given at a conference sponsored by a mining company, which expected no practical application of the theory but sponsored the conference regardless — because of fascination and curiosity, and because of the realization that the good and the true is often indistinguishable from the beautiful. The public is us. We are not basically different from the banker or the baker.

We need to do a better job of making our case for science qua science, and also to use our political powers to protect our programs. At the same time we need to do a better job of understanding what other disciplines are contributing to an understanding of the problems for which we can supply only one facet of the solution. We need only remember a principle of science that we constantly assume — the continuity of nature. It makes sense to look at cancer in fish as evidence of human carcinogenesis. It makes sense to look at the work environment for evidence of what may occur in other species. It makes sense to model the Bay, using techniques borrowed from the physical sciences. It makes sense to use techniques of population study ecologists have developed to analyze the dynamics of human populations. It makes sense first because it is heuristic, from a scientific point of view, and second because the public perception is of the whole, and we must respect that perception if we expect public support to be forthcoming.

QUESTIONS FROM CONFERENCE PARTICIPANTS

Q: There has been a lack of concern here for the edges of the Bay, the wetlands. In the Chesapeake Bay there is an important gap that would tie the biological and energy contributions of our marshes, both fresh and salt, to the productivity of the Chesapeake. This is important right now, because at least in Maryland extensive mosquito control operations, replacing pesticides with upper marsh management, are destroying many of the natural marshes around the Chesapeake; and I am not aware of anyone anywhere monitoring this kind of activity. It is justified by those doing it as being better than the pesticide alternatives, but someone should be taking some interest in the fate of our salt marshes as a result of these destructive management practices.

A: As Ed Houde pointed out, this particular conference did not cover the whole Chesapeake world. Our embryonic plan is, if a conference is held in 1990, to cover a range of different topics.

A: NSF is funding studies of marshes, and we have made progress in understanding them, but it is true that this work has not been done in the Chesapeake.

Q: Most of the work NSF has funded has been in the coastal regions of Georgia, South Carolina, and New England – areas not suffering declines – and these studies have shown the nutrient dynamics and documented the value of marshes to coastal food chains. We need similar studies in the Chesapeake Bay, especially in the mid-Bay region where rapid changes are occurring. We need to know what these changes mean for resources in the Bay.

Q: Another topic omitted from this conference is dredging and its many impacts.

A: A call for papers went out around the Bay, and if topics have not been addressed here it is partly because the people doing the work are not in attendance here. If everyone were aware that a Chesapeake-based conference would be scheduled every two years, people would have a better chance in the future to contribute to the conference and get their interests discussed.

A: Under the Bay Agreement, a comprehensive research plan is to be developed, and scientists around the Bay are integrally involved in contributing ideas for it.
Q: How can we institutionalize the acquisition of long-term datasets?
A: Give money to the lab directors and trust them.
A: The question of institutionalizing data sets is one that concerns a lot of us. This country is not using its computers and networks to make data really available. We need a central, independent, quasi-governmental data organization. The kind of data that people here collect is directly relevant to human ecology and should not be hidden away in specialized databanks.
A: One thing that would help support the long-term monitoring effort would be an agreement at least between the governors of Virginia and Maryland that would commit those states to monitor in the long term.

CONCURRENT SESSIONS AND POSTER SESSION:

LIVING RESOURCES

Chairs:

William Hargis and Linda Schaffner
Virginia Institute of Marine Science
College of William and Mary

Understanding the Estuary: Advances in Chesapeake Bay Research. Proceedings of a Conference. 29-31 March 1988. Baltimore, Maryland. Chesapeake Research Consortium Publication 129. CBP/TRS 24/88.

Abstract only

Effect of Changing pH and Salinity on Chesapeake Bay Striped Bass Larvae

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A laboratory study was conducted to examine the effects of rainfall phenomena on striped bass (Morone saxatilis) larvae under conditions known to occur in Chesapeake Bay striped bass spawning grounds. The two variables under observation were salinity dilution and pH depression as they affected hatchery-reared 10, 13, 16, 19, 22 and 30 day old (post-hatch) larvae from Choptank River, a tributary of the Chesapeake Bay, broodstock. A continuous flow dosing system was developed to expose larvae to three pH depression regimes by metering three pH-controlled test media (pH 7.5, 6.5 and 5.5) into the exposure chambers. Initial pH for all exposures was 7.5. Final pH in the exposure chambers after 24 hours was 7.5, 6.8 and 6.2 for each pH depression regime respectively. The system also controlled salinity maintenance at 1.1 ppt as well as dilution from 1.1 ppt to 0.6 ppt over a 24-hour exposure period. Tests were conducted at 18°C. Test media consisted of "soft reconstituted" bioassay water with a hardness of 42-45 mg/L (as $CaCO_3$).

Statistical analysis revealed that age, pH and salinity were all significant variables affecting larval mortality. Salinity reduction significantly increased mortality in fish exposed at pH 7.5, 6.8 and 6.2 at 10 days of age, pH 6.8 and 6.2 at 13 days, and pH 6.2 at 16 days. Although not statistically significant, elevated mortality in salinity reduced exposure groups was empirically observed in all age groups (tapering off with age) throughout all pH regimes. Organisms in all age exposure groups maintained at 1.1 ppt salinity were not significantly affected by changes in pH. Further, pH depression did not significantly affect mortality in larvae exposed to reduced salinity at 19, 22 and 30 days of age.

The data indicates that survival of 10 day old striped bass larvae is dependent on maintenance of 1.1 ppt salinity independent of pH. This is supported by elevated mortalities obtained at all three pH regimes under salinity reduction. At reduced salinity conditions pH appears to be important to larval survival at 13 and 16 days of age. At 1.1 ppt the effects of pH on mortality is negated. Mortality due to salinity reduction was significantly decreased at pH 7.5, 6.8 and 6.2 with 13, 16, and 19 day old larvae, respectively. Heavy rainfall events temporally occurring during critical periods of larval development may significantly affect young-of-the-year recruitment.

Understanding the Estuary: Advances in Chesapeake Bay Research. Proceedings of a Conference. 29-31 March 1988. Baltimore, Maryland. Chesapeake Research Consortium Publication 129. CBP/TRS 24/88.

Abstract only

Grazing and Egg Production by Chesapeake Bay Zooplankton in Spring and Summer

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Grazing of primary production by zooplankton (> 200 um) was measured in the surface water of the mesohaline portion of Chesapeake Bay in May and August of 1986 and by zooplankton (> 64 um) in March, May and August 1986. Egg production rates were also measured for the dominant copepod species Acartia tonsa in May and August of 1987. Weight specific filtering rates of zooplankton populations, averaged over 24h, were higher in August (333 and 841 ml/mgC/h for 1986 and 1987 respectively) than in May (205 and 57 ml/mgC/h). Average water column grazing pressure was up to an order or magnitude higher in August (3.6 and 14.2 $L/m^3/h$ for 1986 and 1987 respectively) than in May (0.8 and 0.4 $L/m^3/h$), due to increased zooplankton biomass during August for both years. Copepod nauplii had the highest grazing impact of zooplankton > 64 um in March (0.5 $L/m^3/h$) and August (3.0 $L/m^3/h$) of 1987 while polychaete larvae had the highest grazing rates in May (0.06 $L/m^3/h$) of that year. We estimate that zooplankton > 64 um can remove 23% of the primary production in March, 7% in May and 79% in August. Water column egg production rates of Acartia females averaged 0.37 $mgC/m^3/d$ and were not significantly different between May and August, even though nauplii biomass and numbers were twice as high in August. Therefore, the discrepancy between production and standing stock of nauplii in the mesohaline portion of the Bay during May 1987 represents a loss to predation or advection.

Understanding the Estuary: Advances in Chesapeake Bay Research. Proceedings of a Conference. 29-31 March 1988. Baltimore, Maryland. Chesapeake Research Consortium Publication 129. CBP/TRS 24/88.

The Effects of Suspended Sediments on Microzooplankton Grazing in the Patuxent River, A Subestuary of the Chesapeake Bay

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INTRODUCTION

As a eutrophic temperate zone estuary, Chesapeake Bay is typified by high phytoplankton densities and chlorophyll concentrations throughout the system. Nano- and microphytoplankton densities exceed 10^6 cells/l and chlorophyll concentrations range from several to several hundred μg/l. The abundant primary producers should support an active planktonic suspension feeding community, ultimately passing the abundant primary production to higher trophic levels in the nekton and benthos. In Chesapeake Bay, however, fish and shellfish stocks have dramatically declined over the last several decades suggesting some limitation in the Bay from the top (e.g. over-fishing) or from below through reductions in food levels and quality, habitat, etc. (Verity 1987).

Because phytoplankton standing stocks are high and cell size and quality of the Bay's eucaryote phytoplankton is similar to phytoplankton diets supporting high zooplankton growth in other estuaries and cultures, we hypothesized that some factor might limit the quantity of phytoplankton carbon ingested by the planktonic secondary producers in the watershed, i.e. that feeding in the plankton might be reduced resulting in lower secondary production and ultimately biomass of higher trophic levels.

Over the last fifty years, water clarity has declined, as documented by a decrease in secchi disc depth (Heinle et al. 1980). The

decline is attributed to an increase in total suspended particulate material as phytoplankton and inorganic suspended sediments. The latter particulate pool, suspended sediments, has increased due to population growth in the watershed. For example, Schubel (1981) has estimated that there has been a 100-fold increase in sediment loading in the Bay since colonial times due to urbanization and suburbanization. Land use also dramatically alters sediment loading rates to the Bay with minimal input from forested lands (0.5-49 lbs/acre/yr) and highest contributions from croplands and residential areas (11-2461 lbs/acre/yr; EPA 1982).

Ambient concentrations of suspended sediment range from 1-600 mg/l in the Bay and its tributaries (Schubel 1968; Roberts and Pierce 1976; Bennett 1983; OEP 1984). This mixture of non-nutritious inorganic material and phytoplankton provides a suite of potential food items for the planktonic suspension feeders and, potentially, inadequate energy intake should sediments interfere with ingestion or digestion of the planktonic autotrophs.

Deleterious effects of high and moderate concentrations of suspended sediments on mesozooplankton have been reported (Arruda et al. 1983; McCabe and O'Brien 1983) though little research has been conducted on the effects of suspended sediments on microzooplankton. The size of suspended sediments in the Chesapeake Bay (>90% less than 3um, Schubel 1969) overlaps the optimal food size for many microzooplankters (Spittler 1973; Heinbokel 1978a,b; Fenchel 1980; Rassoulzadegan and Etienne 1981; Rassoulzadegan 1982). The presence of these predominantly non-nutritious fine particles could interfere with the feeding and, subsequently, growth of these organisms. If microzooplankton feed only on the basis of size and not "taste" (suggested by Fenchel 1980) and these inert particulates represent a significant portion of ingestion, then reduced growth should occur. Ciliates are known to ingest inert particles, e.g., plastic beads (Fenchel 1980; Borsheim 1984; Albright et al. 1987) and even iron particles (Rifkin and Ballentine 1976). Microzooplankton could also respond to non-nutritious particulate matter by reducing their feeding rates and therefore total ingestion also resulting in reduced growth. If some microzooplankton also exhibit chemoreception, as noted by Stoecker et al. (1981) for a large tintinnine ciliate, ingestion of significant quantities of the silt particles might be avoided, but time spent in handling and rejecting inert particles would decrease ingestion rates of nutritious particles.

Positive effects of suspended particulates on ciliate feeding and growth can also be postulated. Because sediment particles accumulate dissolved organic material and microbial (e.g. bacterial) populations, the organic-inorganic aggregates might supply needed C,N and P resulting in an increase in ciliate growth rates. The association of bacteria and suspended particles has been repeatedly described (e.g. Goulder 1977; Pedros-Alio and Brock 1983). For some species of ciliates grown in axenic and organically rich culture conditions, inert beads must be added to the media to promote growth; beads concentrate nutrients on their surfaces and provide organic-rich particulate material for ingestion. Albright et al. (1987) found that ciliates isolated from marsh floc feed more

efficiently on bacteria associated with particulates (beads) while ciliates from a tidal creek showed preferential feeding on unattached bacteria. With these alternative hypotheses in mind, studies were designed to examine the effects of suspended sediments on microzooplankton.

METHODS

The effects of increasing concentrations of suspended sediments on microzooplankton grazing rates were studied at the Benedict Estuarine Research Laboratory, Benedict, Maryland from winter, 1985 through summer, 1986. All species examined and water samples used in the experiments were collected from the laboratory pier. The sediment concentrations tested ranged between ambient (19-62) to 246 mg/l. Radioisotope labeling techniques were used in short term experiments (4-10 h) initially using a single label (^{14}C-bicarbonate) and subsequently using dual labels (^{14}C-bicarbonate and ^{3}H-methyl thymidine) so that clearance rates for both phytoplankton (^{14}C) and bacteria (^{3}H) could be measured.

Test Organisms

Test organisms included representatives from the three dominant microzooplankton taxa in the Chesapeake Bay, i.e. rotifers, tintinnine ciliates, and oligotrichine ciliates. The rotifer employed, Synchaeta sp., had a mean length (205 μm) just slightly greater than its width in the preserved state. Morphologically this species was very similar to S. baltica though approximately half its size. The tintinnine ciliate studied was Eutintinnus angustatus which has a hyaline lorica with length and oral diameter of 205 and 60 μm, respectively. The oligotrichine ciliate Strombidium sp. was also studied. This ciliate was rather rounded with a mean length and width of 46 and 43 μm, respectively, and the length of its preoral groove was 60% of its total length.

Sediment Preparation

Sediments were collected from farmland north of Harwood, Maryland. Sediments were ground with mortar and pestle until fine, mixed with Patuxent River water, and allowed to settle for 9 minutes; the upper two thirds was subsequently decanted through a series of screens (505, 202 and 153 μm mesh nytex nylon). The resulting slurry was mixed with unfiltered river water to obtain the desired final sediment concentrations.

Experimental Protocol

The dual labeling method of Lessard and Swift (1985) was used with minor modifications. Initially, only ^{14}C was used but subsequent studies used dual isotopes as discussed above. Lessard and Swift's method involves: (1) gentle concentration of the microzooplankton with a fine mesh net which should concentrate the microzooplankton without affecting the available food concentration; (2) addition of label to the concentrate; (3) incubation (4 h); and (4) isolation of the organisms from the radioactive milieu by rinsing then micropipetting individuals into scintillation vials for subsequent estimation of activities.

Adapting this method, originally used in oligotrophic waters, to the relatively eutrophic Chesapeake Bay required some modifications. In preliminary experiments, it was found that the concentration step (1) not only concentrated microzooplankton but the larger algae as well. The carbon fixation of the concentrate was two fold higher than the rate in unconcentrated water. To eliminate this problem net concentrated microzooplankton were micropipetted through at least one wash in unfiltered river water and then into the experimental vessels (55 ml test tubes) containing unfiltered river water. The tubes were spiked with isotope, mixed and placed in a water cooled incubator equipped with flourescent lighting. Following incubation, individual animals were micropipetted through a series of filtered seawater washes. An analysis of the radioactivity remaining in water from each of six rinses of the organisms following an experiment suggested that 6 rinses were necessary to remove residual non-fixed label. In addition, a time-zero control was introduced to control for surface attachment of the label and feeding during the rinse period. The controls were set up in an identical manner to the experimental tubes; however, after the addition of label the tubes were immediately sacrificed.

RESULTS

The relationship between sediment concentration and clearance rate for Synchaeta sp. was determined in a set of experiments conducted on February 5, 1986 (4.4 °C, 12.7 °/oo). Though the clearance rates were relatively low, a significant increase in grazing rate was found at moderate sediment concentrations (38 mg/l) relative to ambient (Figure 1). Clearance rates at 120 and 136 mg/l were not significantly different from ambient while clearance rate at the highest sediment concentration was significantly lower than at any other sediment concentration.

Concurrent with this experiment, an incorporation experiment was conducted in which the rotifers, after having fed on labeled food for eight hours, were placed in unlabeled food and the loss of label monitored approximately every 2 hours for 10 hours. The results indicated a consistent 5% loss rate per hour.

In a later experiment (July 23, 1986, 29.0 °C, 12.8 °/oo), the dual label method was used to compare the response of clearance rate on phytoplankton and bacteria to increasing suspended sediment concentrations. A tintinnine ciliate Eutintinnus angustatus was the test organism in these experiments. Clearance rates on both phytoplankton and bacteria were stimulated at moderate sediment concentrations and decreased at the highest levels (Figure 2). However, the peak in grazing rate on phytoplankton occurred at 74 mg/l suspended sediment and the highest sediment concentration resulted in a clearance rate significantly less than that at the ambient sediment concentration; in contrast, the clearance rates on bacteria peaked at 99 mg/l and the rate at the highest sediment level was significantly greater than the rate at the ambient suspended sediment level. This may be due to the ingestion of bacteria adhered to sediment particles.

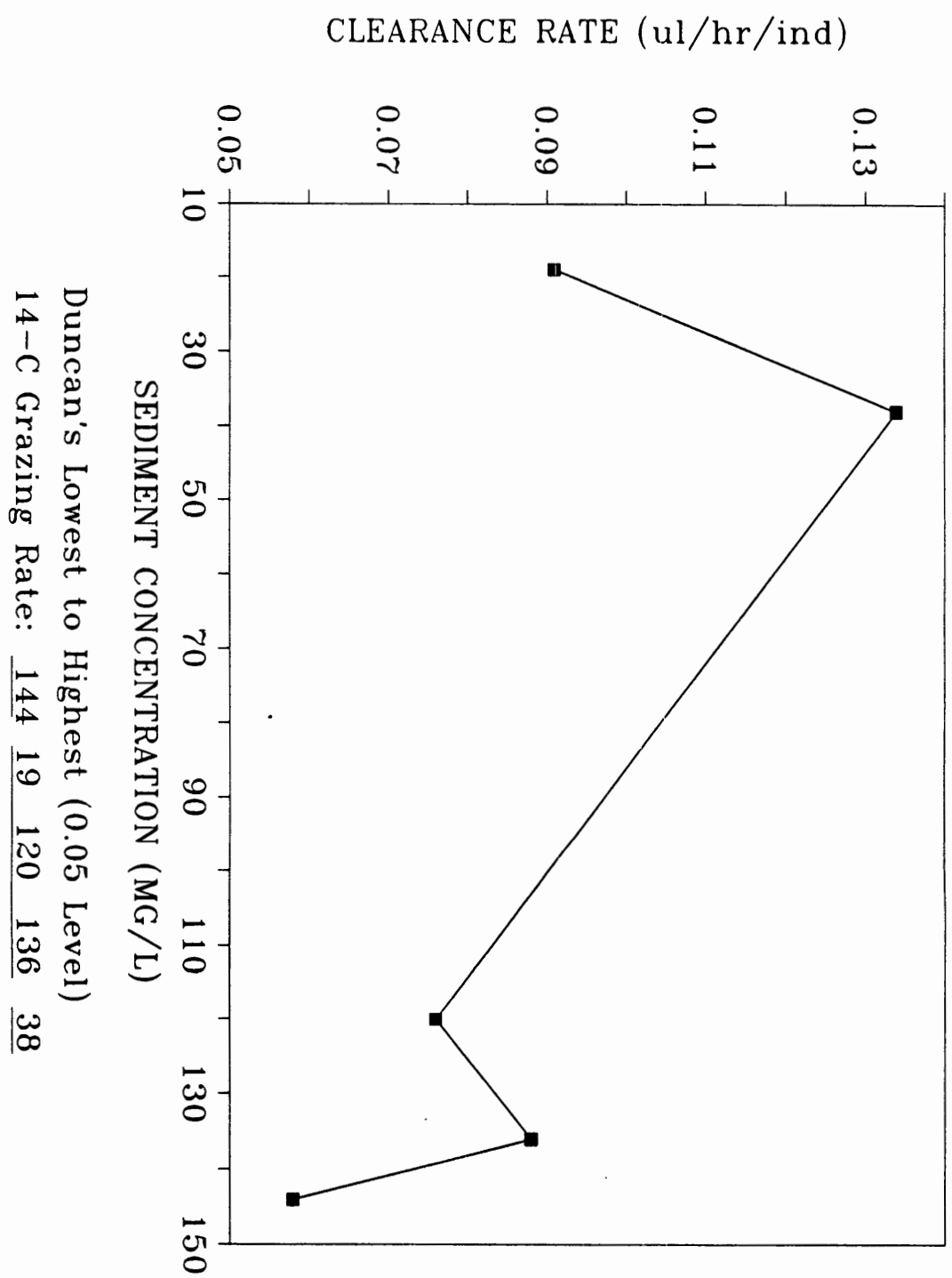
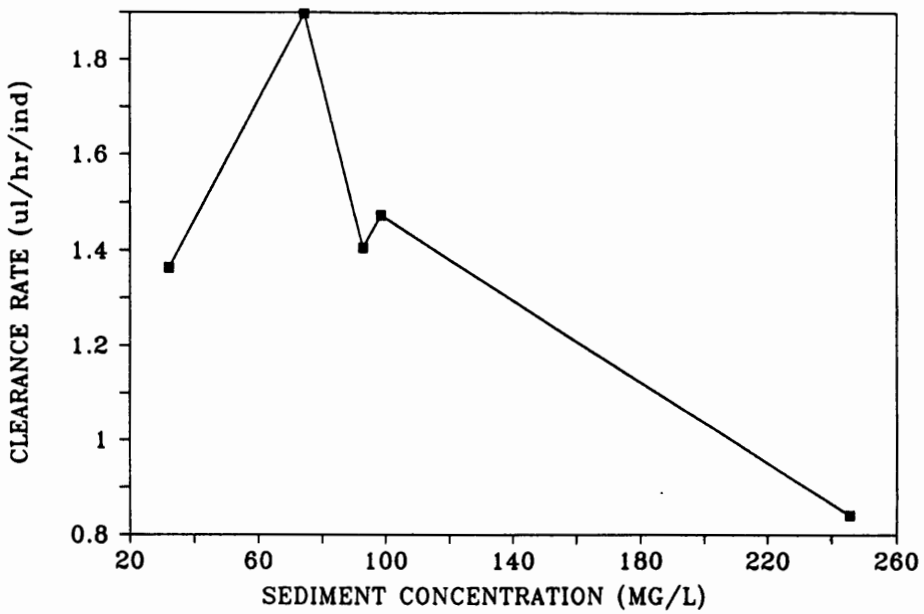
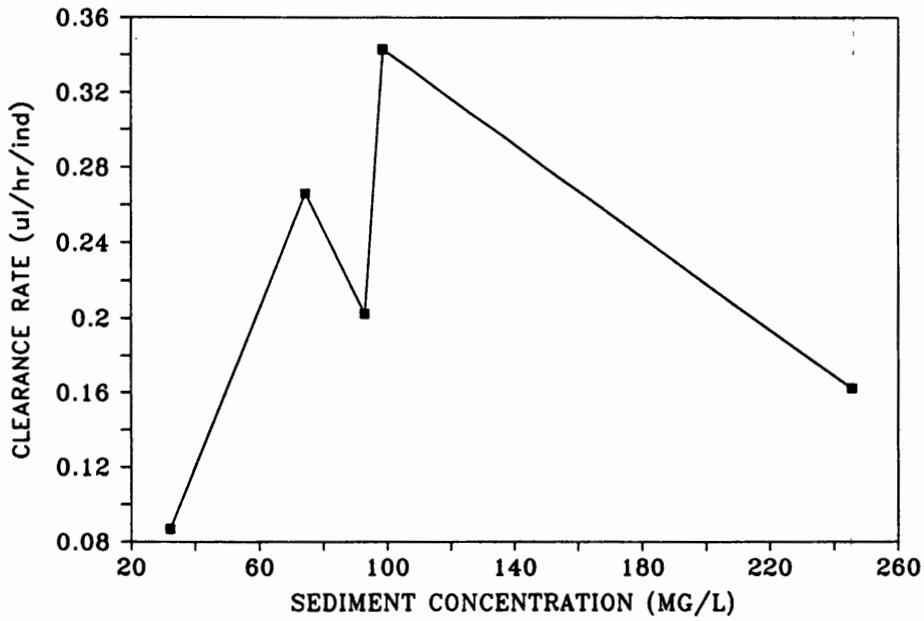


Figure 1. Relationship between suspended sediment concentration and clearance rates of phytoplankton by the rotifer Synchaeta sp. In the Duncan's test, a break in the underline denotes a significant difference.



Duncan's Lowest to Highest (0.05 Level)
14C Grazing Rate: 246 32 93 99 74



Duncan's Lowest to Highest (0.05) Level)
3H Grazing Rate: 32 246 93 74 99

Figure 2. Relationship between suspended sediment concentration and clearance rate on phytoplankton (above) and bacteria (below) by the tintinnine ciliate Eutintinnus angustatus. In the Duncan's test, a break in the underline denotes a significant difference.

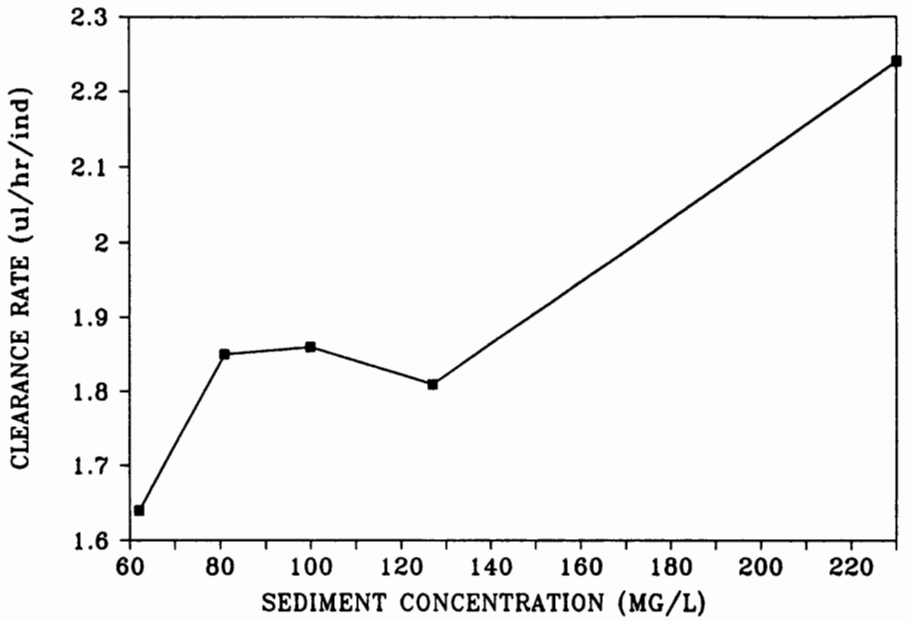
A similar study was conducted for Strombidium sp., an oligotrichine ciliate (August 6, 1986, 29.6 °C, 11.8 o/oo). Clearance rates on bacteria followed a similar pattern to that demonstrated by E. angustatus feeding on bacteria with highest clearance rates at about 100 mg/l suspended sediment (Figure 3). However, clearance rates on phytoplankton showed a divergent pattern with rates increasing sharply between 62 (ambient) and 81 mg/l, similar rates between 81 and 127 mg/l, and a dramatic increase at 230 mg/l. In addition, behavioral changes were noted with increasing sediment concentration. At the highest level of suspended sediment, this species showed errant swimming behavior: continuous rapid jumping and twirling was obvious at the highest turbidity levels.

DISCUSSION

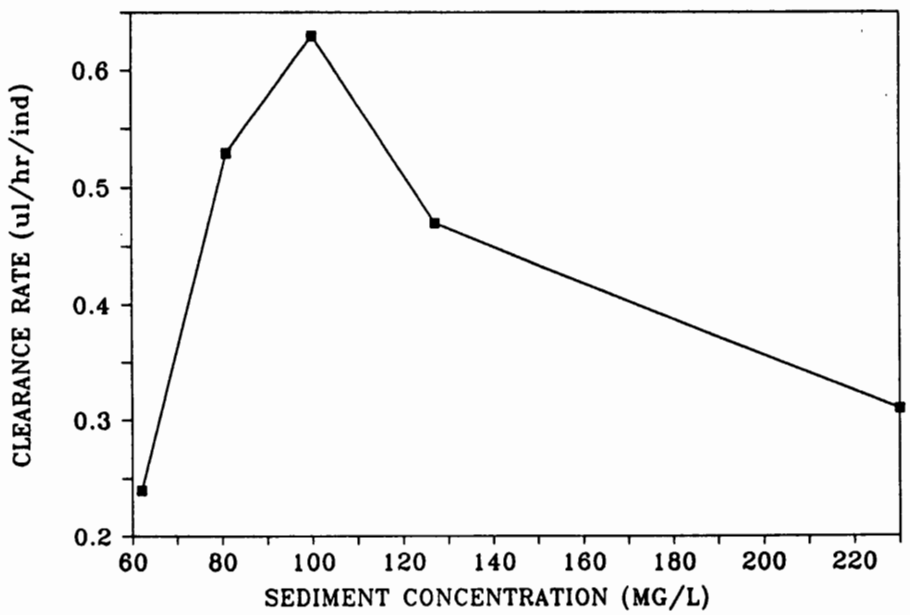
The stimulation of microzooplankton clearance rates at moderate levels of suspended sediment and the depression of these rates at higher levels has also been observed for other zooplankters including the copepod Acartia tonsa (Sellner et al. 1987 a,b). Suspended sediment levels observed in Chesapeake Bay and its tributaries, therefore, might prove beneficial to particle (and energy) intake in small protozoa, perhaps favoring enhanced growth and reproduction. However, previously reported data in studies of sediments and planktonic crustaceans do not support one consistent pattern with all taxa. In some studies, even moderate levels of suspended sediment resulted in decreased reproduction and growth of mesozooplankton (Arruda et al. 1983; McCabe and O'Brien 1983); however, this pattern is not always observed as Sellner et al. (1987a,b) saw no decrease in growth or reproduction until very high suspended sediment concentrations were employed.

The clearance rates observed in the present study are similar to rates previously reported. For rotifers feeding on phytoplankton, clearance rate can vary over two orders of magnitude (Starkweather and Gilbert 1977) depending on food concentration, high levels of food resulting in a low clearance rate. The rates we obtained for phytoplankton (0.05-0.13 µl/h/ind) are characteristic of high food concentrations and low temperatures (Bogdan and Gilbert 1982; Gilbert and Bogdan 1984). During the grazing experiments with Synchaeta, chlorophyll-a concentrations were relatively high (24.4 µg/l) and temperature low (4.4 °C), consistent with the relatively low grazing rates observed.

When feeding on phytoplankton or bacteria clearance rates for the tintinnine ciliate E. angustatus (0.84-1.9 and 0.09-0.34 µl/h/ind, respectively) compared well with published values (phytoplankton - Spittler 1973, 0.5-8.5; Heinbokel 1978a,b, 0.5- 9.0; Capriulo and Carpenter 1980, 1-85; Capriulo 1982, 2-65; Lessard and Swift 1985, 0-32; bacteria - Hollibaugh et al. 1980, 0.042; Lessard and Swift 1985, 0-11 µl/h/ind. Heinbokel (1978a) found that high food densities resulted in lowered clearance rates (e.g. from 5 to 1 µl/h/ind when offered low and high food densities, respectively). Chlorophyll-a concentrations during the E. angustatus study were high (25 µg/l) and thus clearance rates on the lower side of these ranges would be expected.



Duncan's Lowest to Highest (0.05 Level)
14-C Grazing Rate: 62 81 100 127 230



Duncan's Lowest to Highest (0.05 Level)
3H Grazing Rate: 62 230 127 81 100

Figure 3. Relationship between suspended sediment concentration and clearance rate on phytoplankton (above) and bacteria (below) by the oligotrichine ciliate Strombidium sp. In the Duncan's test, a break in the underline denotes a significant difference.

Strombidium sp. exhibited clearance rates on phytoplankton (1.6- 2.2 µl/h/ind) that are within the range of reported values for other oligotrichine ciliates (Rassoulzadegan 1982, 2.3-8.9; Scott 1985, 0.083-1.5; Lessard and Swift 1985, 0-213 µl/h/ind). The rates on bacteria (0.24-0.63 µl/h/ind) are generally lower than those reported (Lessard and Swift 1985, 0-7.6; Rivier et al. 1985, 0.6-73 µl/h/ind) but are within the values reported for tintinnine ciliates (see above).

The decreased clearance rates at high sediment concentrations for Synchaeta and E. angustatus could be the result of impaired feeding or due to a behavioral response (i.e. decreased feeding) in the presence of elevated levels of suspended sediment. The later possibility may be an adaptation to survive short periods of high turbidity. The reason for the opposite response found at high sediment concentrations for clearance rates on phytoplankton and bacteria by Strombidium is unknown. One possible explanation might be that the erratic behavior of this species observed at the highest sediment concentration might have been associated with rejection of sediment particles and capture of phytoplankton cells. Though the movements appeared energetically expensive, this behavior may be an alternative mechanism for surviving in high suspended sediment concentrations for short periods of time.

One of the assumptions made in estimating clearance rates from these types of experiments is that significant elimination of label (excretion, defecation, respiration) does not occur during the time course of the experiment. This assumption may be violated when using the 4 h incubation time. Little information exists to determine how long all of the label might remain within the various taxa of microzooplankton. However, the results of the incorporation study with Synchaeta suggest that the label has already obtained a constant rate of elimination by two hours. As the duration of all experiments was greater than or equal to 4 hours, it is suggested that measured clearance rates may be better interpreted as assimilation or incorporation rates. However, the incubation period does not alter the effects of suspended sediments on relative clearing rates on bacteria and phytoplankton.

Future studies are anticipated to determine the effects of suspended sediments on growth and reproduction of microzooplankton. The question remains as to whether or not stimulation in clearance rate at moderate suspended sediment concentrations will result in increased growth (representing increased ingestion of nutritious particles along with sediments), decreased growth (the harmful effects of ingesting large quantities of inorganic particles outweighing any beneficial effect of ingesting more prey), or equivalent growth (clearance rates are increased to counterbalance the decreased ingestion of nutritious particles due to interference in feeding caused by the suspended sediments). If deleterious long-term effects are observed, then the increases in sediment loads to the Chesapeake Bay and its tributaries could interfere with the transfer of carbon to higher trophic levels directly through reducing assimilation and therefore reproduction in zooplankton or

indirectly by altering the species composition of the plankton. If less phytoplankton are grazed, more would remain to settle to bottom waters for subsequent decomposition and oxygen demand in bottom waters.

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LITERATURE CITED

Albright, L.J., E.B. Sherr, B.F. Sherr and R.D. Fallon. 1987. Grazing of ciliated protozoa on free and particle-attached bacteria. Mar. Ecol. Prog. Ser. 38: 125-129.

Arruda, J.A., G.R. Marzolf and R.T. Faulk. 1983. The role of suspended sediments in the nutrition of zooplankton in turbid reservoirs. Ecology 64: 1225-1235.

Bennett, J.P. 1983. Nutrient and sediment budgets for the tidal Potomac River and estuary. In: Dissolved loads of rivers and surface waters: Quantity/quality relationships. Proc. Hamburg Sympos., August, 1983. IAMS Publ. No. 141, 217-227.

Bogdan, K.G. and J.J. Gilbert. 1982. Seasonal patterns of feeding by natural populations of Keratella, Polyarthra, and Bosmina: Clearance rates, selectivities, and contributions to community grazing. Limnol. Oceanogr. 27: 918-934.

Borsheim, K.Y. 1984. Clearance rates of bacteria-sized particles by freshwater ciliates, measured with monodispersed fluorescent latex beads. Oecologia 63: 286-288.

Capriulo, G.M. 1982. Feeding of field collected tintinnid micro-zooplankton on natural food. Mar. Biol. 71: 73-86.

Capriulo, G.M. and E.J. Carpenter. 1980. Grazing by 35-202 μm micro-zooplankton in Long Island Sound. Mar. Biol. 56: 319-326.

EPA (U.S. Environmental Protection Agency). 1982. Chesapeake Bay technical studies: A synthesis. Washington, DC. 634 pp.

Fenchel, T. 1980. Suspension feeding in ciliated protozoa: Feeding rates and their ecological significance. Microb. Ecol. 6: 13-25.

Gilbert, J.J. and K.G. Bogdan. 1984. Rotifer grazing: In situ studies on selectivity and rates. Pages 97-133 in: D.G. Meyers and J.R. Strickler (eds.), Trophic interactions within aquatic ecosystems. AAAS, Washington, D.C.

Goulder, R. 1977. Attached and free bacteria in an estuary with abundant suspended solids. J. Appl. Bacteriol. 43: 399-405.

Heinbokel, J.F. 1978a. Studies on the functional role of tintinnids in the Southern California Bight. I. Grazing and growth rate in laboratory cultures. Mar. Biol. 47: 177-189.

Heinbokel, J.F. 1978b. Studies on the functional role of tintinnids in the Southern California Bight. II. Grazing rate of the field populations. Mar. Biol. 47: 191-197.

Heinle, D.R., C.F. D'Elia, J.L. Taft, J.S. Wilson, M. Cole-Jones, R.B. Caplins and L.E. Cronin. 1980. Historical review of water quality and climatic data from Chesapeake Bay with emphasis on

effects of enrichment. U.S. EPA Chesapeake Bay Prog. Final Rept., CRC Public. No. 84, Annapolis, MD. 128 pp.

Hollibaugh, J.T., J. Fuhrman and F. Azam. 1980. Radioactively labeling in natural assemblages of bacterioplankton for use in trophic studies. Limnol. Oceanogr. 25: 172-181.

Lessard, E.J. and E. Swift. 1985. Species-specific grazing rates of heterotrophic dinoflagellates in oceanic waters, measured with a dual-label radioisotope technique. Mar. Biol. 87: 289-296.

McCabe, G.J. and W.J. O'Brien. 1983. The effects of suspended silt on feeding and reproduction of Daphnia pulex. Amer. Midland Natural. 110: 324-337.

OEP (MD Office of Environmental Programs). 1984. Patuxent estuary water quality survey. 1983 data summary. MD Dept. Health and Mental Hygiene, Technical Rept. No. 3, Baltimore, MD. 57 pp.

Pedros-Alio, C. and T.D. Brock. 1983. The importance of attachment to particles for planktonic bacteria. Arch. Hydrobiol. 98: 354-379.

Rassoulzadegan, F. 1982. Dependence of grazing rate, gross growth efficiency and food size range on temperature in a pelagic oligotrichous ciliate Lohmanniella spiralis Leeg., fed on naturally occurring particulate matter. Inst. Oceanogr. 58: 177- 184.

Rassoulzadegan, F. and M. Etienne. 1981. Grazing rates of the tintinnid Stenosemella ventricosa (Clap. and Lachm.) Jorg. on the spectrum of the naturally occurring particulate matter from a Mediterranean neritic sea. Limnol. Oceanogr. 26: 258-270.

Rifkin, J.L. and R. Ballentine. 1976. Magnetic filtering of the ciliated protozoan Tetrahymena pyriformis. Trans. Amer. Micros. Soc. 95: 187-197.

Rivier, A., D.C. Brownlee, R.W. Sheldon and F. Rassoulzadegan. 1985. Growth of microzooplankton: A comparative study of bactivorous zooflagellates and ciliates. Mar. Microb. Food Webs 1: 51-60.

Roberts, W.P. and I.W. Pierce. 1976. Deposition in upper Patuxent estuary, Maryland 1968-1969. Est. Coastal Mar. Sci. 4: 267-280.

Schubel, J.R. 1968. Turbidity maximum of the northern Chesapeake Bay. Science 161: 1013-1015.

Schubel, J.R. 1969. Size distributions of the suspended particles of the Chesapeake Bay turbidity maximum. Neth. J. Sea Res. 4: 283-309.

Schubel, J.R. 1981. The Living Chesapeake. Johns Hopkins, Baltimore, MD. 113 pp.

Scott, J.M. 1985. The feeding rates and efficiencies of a marine

ciliate, Strombidium sp., grown under chemostat steady-state conditions. J. Exp. Mar. Biol. Ecol. 90: 81-95.

Sellner, K.G., M.H. Bundy and J.W. Deming. 1987a. Influences of suspended sediments on feeding and production of the copepod Acartia tonsa. EOS 68: 1732, Abst. (31G-10).

Sellner, K.G., J.W. Deming and M.H. Bundy. 1987b. The effects of suspended sediment on copepods and carbon transfer in Chesapeake Bay. Abstract and presentation at the 50th Annual Meeting of the American Society of Limnology and Oceanography, June 14-18, 1987.

Spittler, P. 1973. Feeding experiments with tintinnids. Oikos 15(Suppl.): 128-132.

Starkweather, P.L. and J.J. Gilbert. 1977. Feeding in the rotifer Brachionus calyciflorus. 2. Effect of food density on feeding rates using Euglena gracilis and Rhodotorula glutinis. Oecologia 28: 133-139.

Stoecker, D., R.R.L. Guillard and R.M. Kavee. 1981. Selective predation by Favella ehrenbergii (Tintinnina) on and among dinoflagellates. Biol. Bull. 160: 136-145.

Verity, P.G. 1987. Factors driving changes in pelagic trophic structure of estuaries, with implications for the Chesapeake Bay. pp 35-56. In: M.P. Lynch and E.C. Krome (eds.), Perspectives on the Chesapeake Bay: Advances in estuarine sciences. CRC Publication No. 127. Chesapeake Research Consortium. Gloucester Point, Virginia.

Understanding the Estuary: Advances in Chesapeake Bay Research. Proceedings of a Conference. 29-31 March 1988. Baltimore, Maryland. Chesapeake Research Consortium Publication 129. CBP/TRS 24/88.

Abstract only

Temporal and Spatial Variations in Zooplankton of the Mesohaline Portion of Chesapeake Bay

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As part of the NOAA-Sea Grant Program on Anoxia in Chesapeake Bay, we conducted measurements of the short-term (tidal, diel, < 10-day) variability in the distribution of zooplankton (> 64 um) in the mesohaline portion of Chesapeake Bay during May and August 1986 and 1987. Hydrographic and biological parameters were measured along with the vertical distribution of zooplankton at both 30 h anchor stations in the mid-Bay channel and along a 5-station transect across the Bay. During both years, there was a 2-5 fold increase in zooplankton biomass between May and August. There was an order of magnitude increase in the abundance of nauplii in the surface waters between May (x = 10/L) and August (x = 100/L). Considerable short-term variability occurred during both months. Zooplankton samples collected in mid-Bay every 1.5 h for 30 h varied by a factor of 5 in May and 10 in August. In general, we found that significant diel vertical migration occurred in August but not in May. Transect stations across the Bay showed that in general: there were more zooplankton present in the middle of the Bay than on the flanks, and that there were greater concentrations of zooplankton on the western side of the Bay as compared to the eastern side of the Bay. Short-term variability in the cross-Bay distribution of zooplankton can occur as a result of pycnocline "tilting" in response to wind events.

Understanding the Estuary: Advances in Chesapeake Bay Research. Proceedings of a Conference. 29-31 March 1988. Baltimore, Maryland. Chesapeake Research Consortium Publication 129. CBP/TRS 24/88.

Simulating the Vertical Motion of Nekton in the Estuarine Environment - Scientific Considerations and Speculations

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ABSTRACT

The vertical and horizontal distributions, and resulting patterns of living particles (plankton and nekton) are modulated by biological and physical characteristics within the estuary. The transition reach in estuaries is the location where spawning of anadromous species occurs. This compartment within the estuary is also the location where wind, river flow, the upper estuarine salinity regime, and tidally induced vertical mixing and advection interact with radiative transfer processes, in the optical and thermal regions of electromagnetic energy spectrum, to provide an environment for nekton growth and development. These environmental processes are conceptualized as the stimuli affecting the statistical characteristcs of nekton swimming (speed and direction) during their early life stages. Particular processes are also related to providing the environment for sustaining food sources for the nekton. We discuss the formulation and demonstration of an approach to describe the vertical motion of nekton in estuaries with reference to the Upper Chesapeake Bay. Specifically, a simulation model with deterministic and stochastic components is applied with reference to Morone saxatilis eggs and larvae. Typical model output and an approach for sensitivity analysis are presented which provide insight into selected environmental processes and swimming characterisitcs which may be related to larval retention

and recruitment. Preliminary results suggest that statements concerning "active versus passive transport" of living particles can be refined to "horizontally passive and vertically active" in particular marine environments. This conceptual refinement can be assessed within the marine environment by examining a dimensionless number we call Pm (a particle number), which is obtained from scaling and dimensional considerations using a form of the backwards diffusion equation and conservation of probability. The research methods used in this work may provide insight into processes that should be monitored and considered in the development of living resources management strategies which attempt to protect the spawning habitat for anadromous species within subestuaries of Chesapeake Bay.

INTRODUCTION

The management of estuarine living resources requires knowledge of their relative abundance and distributions. The population dynamics of other species which support and interact with the target estuarine species must also be understood in at least a conceptual manner to discern their role as predator or prey.

The habitat and environmental quality that directly impact the early life stage success and recruitment of the living resources also needs to be understood. An important indicator of habitat quality in estuaries is the underwater light field (ULF). The changes in the underwater light field and factors affecting it is sometimes termed "subsurface light climatology". The characteristics of the subsurface irradiant light field has been studied by various researchers (Burt 1953, Champ 1980, Clarke and Backus 1956), but there remains considerable uncertainty as to its role as an "environmental stimulus" which nekton (small organisms exhibiting motility or swimming) respond to, and use in their behavioral strategy to live, especially in turbid estuarine water. How nekton may use specific charateristics of the temporal and spatial variations in the optical portion of the electromagnetic spectrum may have important implications. These may include aspects concerning vertical motion, migration and patchiness. These factors may ultimately affect their horizontal translocation and/or retention within estuaries or particular spawning or nursery habitats.

TECHNIQUE DEVELOPMENT

Scientific Considerations

In view of the above, the scientific approach taken is, by necessity, interdisciplinary and involves the application of numerical techniques. The mathematical conceptualizations

are a result of bio-ecological knowledge, estuarine environmental processes, applied optics and physical oceanographic principles.

The techniques employed depend upon laboratory studies which describe the swimming characteristics of nekton or "living particles" in terms of speed and direction statistics. The techniques involve implementation of expertiments in a laboratory setting as well as in the field, concerning the settling characterisitcs (Sunby 1983) of the egg stage of the living particles or nekton. Qualitative information collected by researchers and resource management agencies assist in discerning vertical and horizontal distributions of this class of particles.

At the same time one must recognize the lack of precision and accuracy of current field collections of distribution and abundance data during the early life stages. Our current technology prevent accurate descriptions because montioring programs are not typically designed to assess the relative importance of space and time scales which affect early life stage success or "natural mortality". To do so would be too costly, therefore resort to research and numerical modeling techniques is necessary.

Figure 1 indicates the spatial and temporal scales of selected physical and biological processes in the estuarine environment. Qualitatively, this shows the spatial and temporal frequency upon which scientific and management data should be considered for developing research and monitoring strategies. Currently, monitoring during "selected windows" is conducted. For example, no data except secchi disc is collected on a routine basis concerning the ULF in Chesapeake Bay by management agencies. This should be given serious consideration by upper scientific and government management officials since light energy is a fundamental physical quantity used by the phytoplankton community which respire, decay and contribute to low dissolved oxygen. In addition, associated light attenuation is one, if not a key parameter used in driving water quality simulations for management purposes. However, it does exhibit wavelength dependence.

Research has demonstrated the importance of phytoplankton, nekton density or primary particle concentrations upon the attenuation of light in the marine envrionment. These concentrations are amplified or can decrease with migration from prey (zooplankton), swimming, and specific growth rates (Huntley, Marin and Escritor 1987).

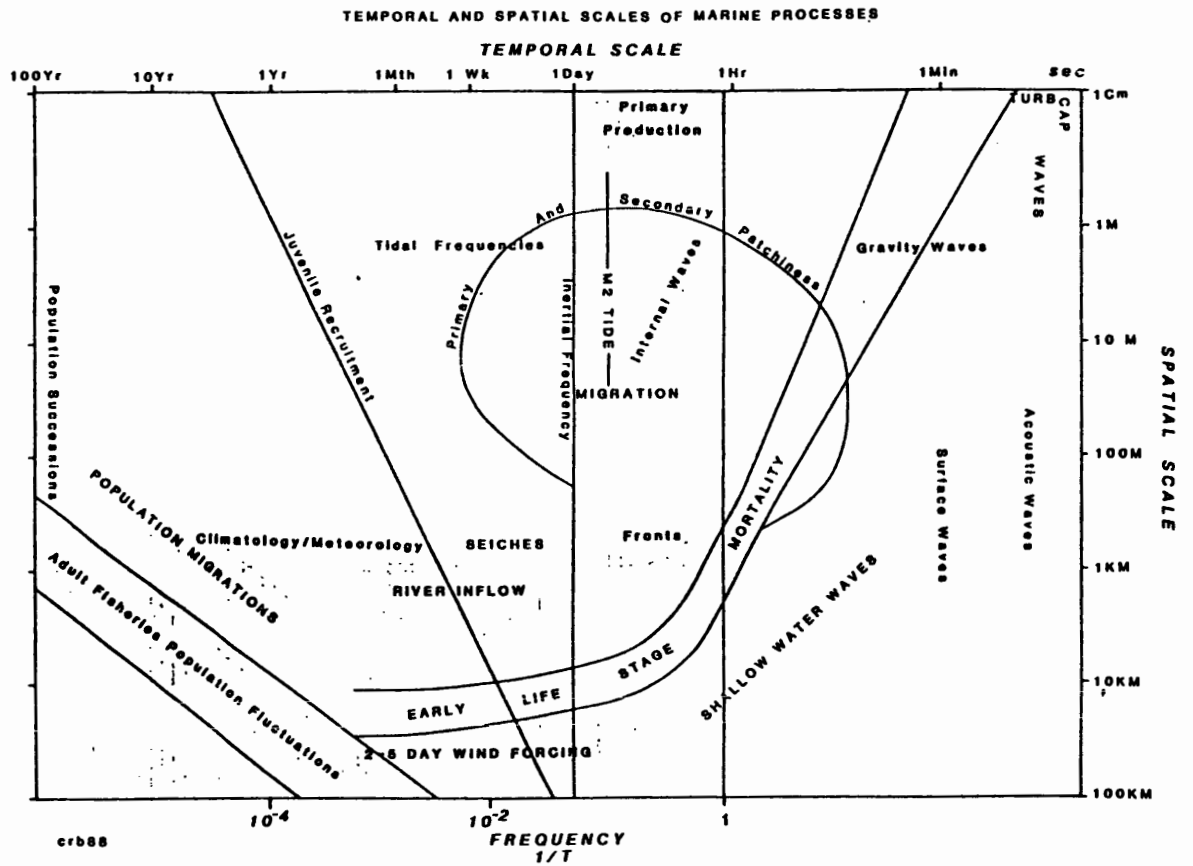

Figure 1. Selected spatial and temporal scales of estuarine and marine processes.

Previous Similar Research

Previous researchers such as Kamykowski 1976 have examined the relationships between small scale circulation processes such as internal waves and the vertical motion of organisms. Recent research has resulted in estimates (Kamykowski 1988) concerning swimming characterisitcs of dinoflagellate swimming for possible coupling with particle transport processes.

Other efforts have included the development of simplified vertical migration mechanisms such as attempts to relate larval distributions to circulation in the North Sea. Bartsch 1988 reported on coupling a three dimensional circulation model of the North Sea with particle tracing tecnqniques. In his effort, the vertical migration response (an active transport mechanism) was shown to strongly affect the predicted horizontal translocation of herring larvae. Without vertical migration, the final passive particle locations did not effectively match the field collection data.

Zooplankton research results presented at this conference suggested vertical migration and possible interaction of active vertical transport with internal waves, crossbay seiches and potential upwelling within the mainstem segments of Chesapeake Bay. Research at Virginia Institute of Marine Science has addressed the vertical motion of oyster larvae and their interaction with frontal dynamics (A. Kuo 1988, pers. comm.). Previous efforts concerning striped bass larvae in the Potomac estuary have indicated they maintain their early life stages in the complex estuarine transiton reach (Setzler-Hamilton et. al. 1981).

Current Speculations

One hypothesis that may help to explain the maintenance of larval densities in or just above the estuarine transition reach (salinity 0-10 ppt) is vertical migration. One can invoke the simplified basin equations (Knauss, 1978) and estimate a dimensionless scale which represents the difference between the time it would take for a hypothetical particle to move from the middle of the upper layer, to the middle of the lower layer and vice versa, under specified salinity and freshwater inflow conditions. This scale and its variability can be assesed for different salinity and freshwater inflow regimes given sufficient field data and by assuming a migration period. This dimensionless scale is given by T'/T where $T' = P-T$, and it can be shown that:

$$T=P/[(Q/A)/(Q'/A')+1] \quad . \qquad (1)$$

The primed terms denote mean lower layer values of cross-sectional average fluxes (Q, Q′) derived from the basin equations which make use of river inflow and layer average salinities. P is representative of a migration period and A, A′ are crossectional areas for the upper and lower layers respectively. These terms can be estimated from vertical salinity information from shipboard measurements. Adjustments in annual cycles of swimming behavior has been demonstrated for other fresh and brackish water finfish (Eriksson 1984).

Using available field data for the extreme Upper Chesapeake Bay, one notes the change in this scale during the high and low flow years of 1984, 1985 respectively, as indicated in Figure 3. One can speculate that in order for organisms to maintain a particular longitudinal position in an estuarine reach and if migration was used as the behavioral mechanism, this behavior may have to be rather adaptable or the organisms could be flushed into higher salinity waters that may affect survival. Alternatively, higher freshwater inflows following storm events or higher flows are associated with dramatic changes in the underwater light climatology as indicated by changes in light attenuation. The effect upon the ULF can be dramatic and can be asessed quite readily from field data as well as from satellite data, where the irradiance reflectance ratio is shown to be dependent upon light attenuation coefficients (DiToro 1978) which vary with wavelength. Figure 2 shows the changes in the effective radiance from three wavelengths in the visible portion of the light spectrum from the Landsat, Thematic Mapper multispectral scanner for the Upper Chesapeake Bay. The color variations are related to changes in light attenuation and the estuarine processes of scattering and absorption.

Swimming Characteristics

Vertical swimming speeds can be assessed from video image processing techniques as depicted schematically in Figure 4. Under laboratory conditions, swimming behavior can be statistically assessed and used to derive speed and direction information. This framework is used for assessing the vertical swimming response of nekton. For example, Figure 5 indicates the vertical swimming or "geotaxis" of striped bass larvae shortly after they hatch. At a very early age striped bass larvae swim vertically, followed by sinking and then then swimming vertically again. As these nektonic organisms grow, they increase their swimming speed and directions. Their swimming speed distributions can be assessed by using statistical distributions as shown in Figure 6. One can then use Monte-Carlo stochastic simulation techniques (Nelson 1982, Ripley 1987) to model a particular taxis to an environmental process. We have

Figure 2. Thematic Mapper image representative of "effective" light attenuation in Upper Chesapeake Bay.

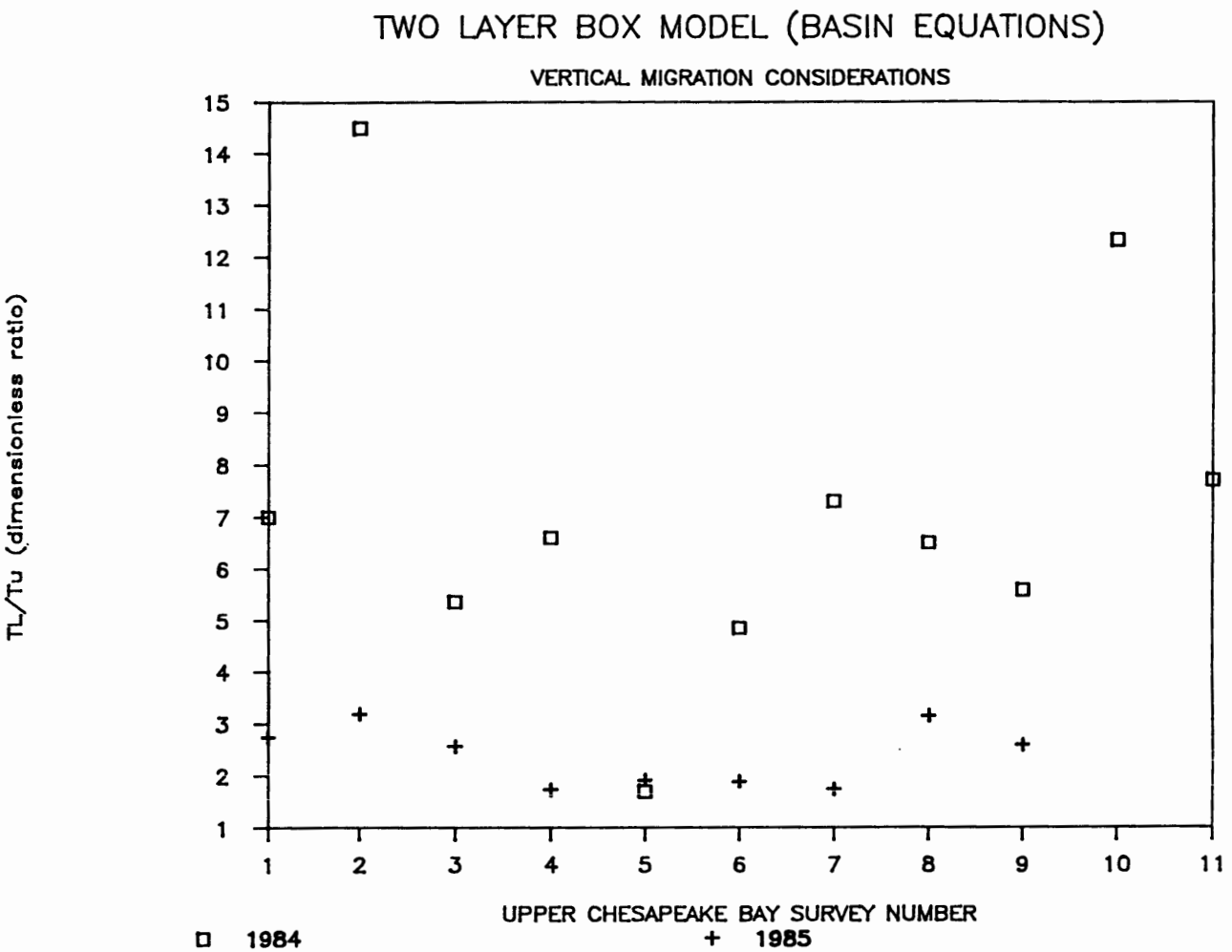

Figure 3. Estimates of a dimensionless steady state vertical time scale during 1984(high flow) and 1985 (low flow) spawning seasons for the Upper Chesapeake Bay.

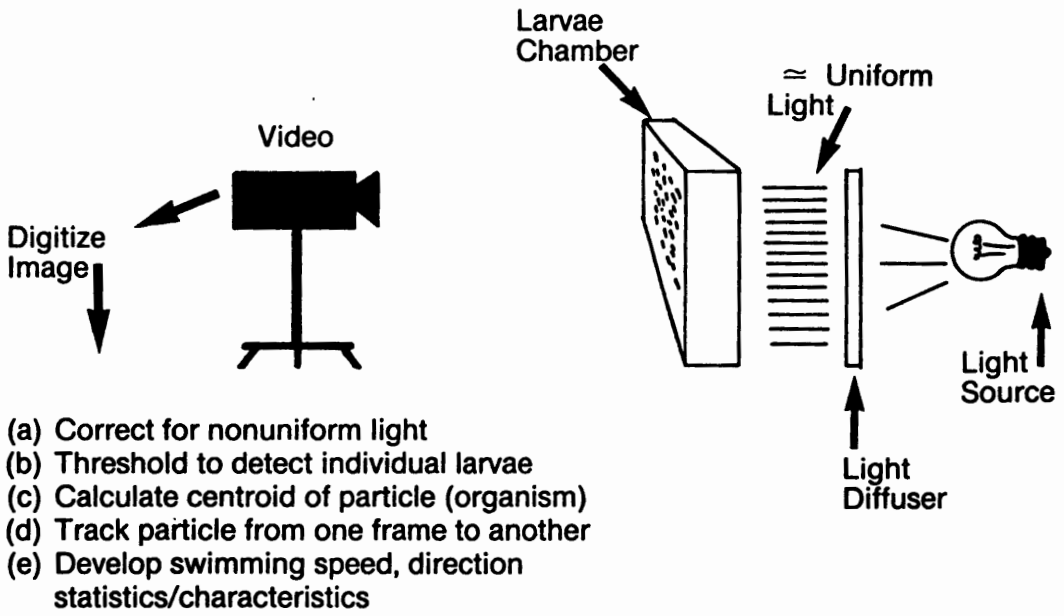

Figure 4. Schematic representation of a laboratory procedure used to estimate swimming speed statistical distributions of nekton.

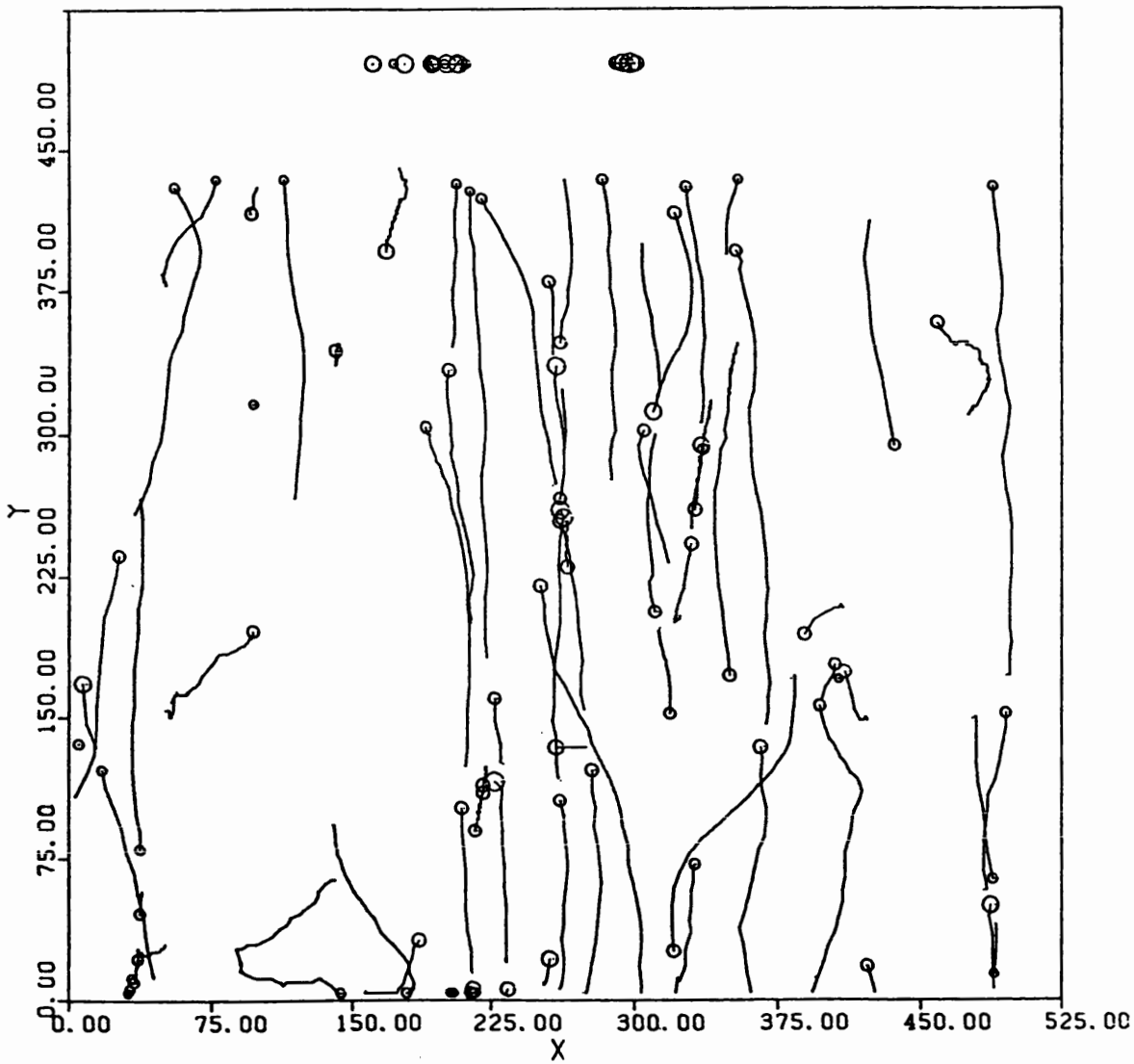

Figure 5. Vertical motion "tracks" from a laboratory experiment demonstrating "geotaxis" in striped bass larvae within 12 hours from hatching.

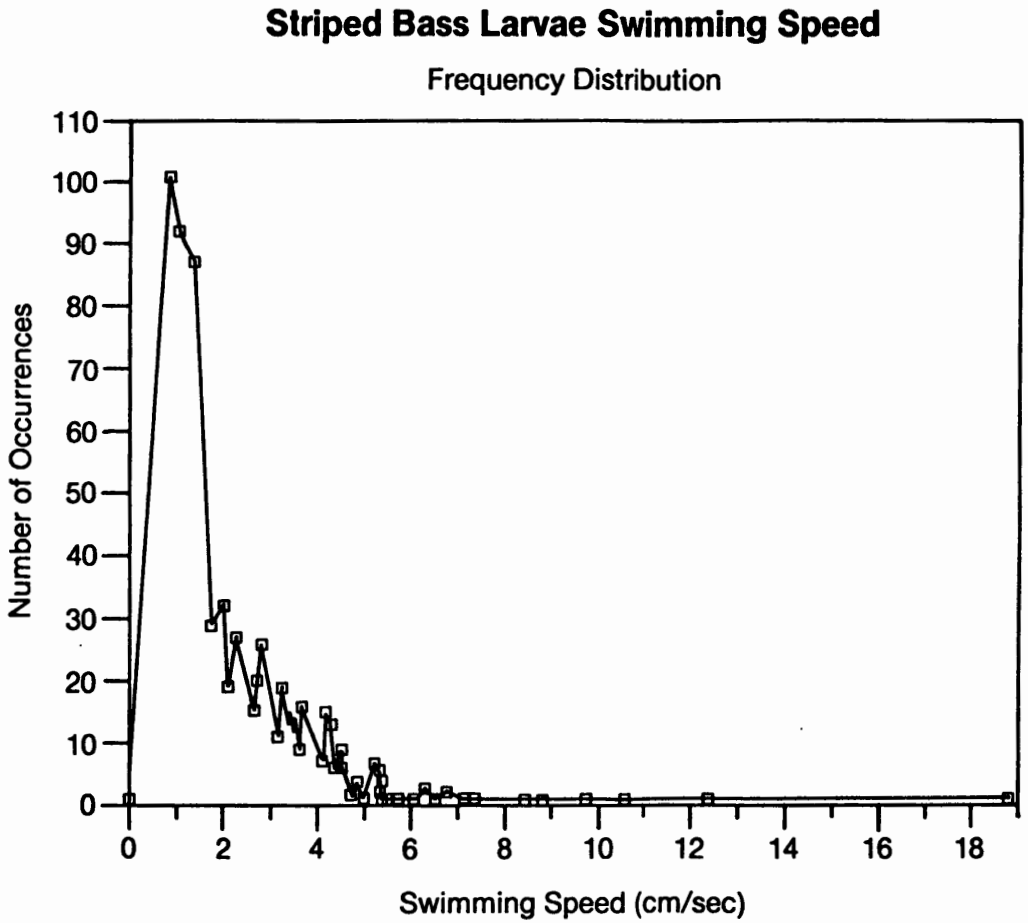

Figure 6. Example statistical distribution of swimming speeds for 5 day old larvae.

chosen to use stochastic approaches to model the swimming process. When these numerical techniques are used in conjunction with analytical or numerical characteristics of an estuarine physical environment (vertical flux or vertical velocties), one can estimate the time rate of change of abundance or "vertical motion" of the nekton or living particles.

NUMERICAL SIMULATIONS

Figure 7 indicates a simple example of the model output. As indicated above, we couple time variable vertical swimming speed and direction characterisitcs estimated from application of stochastic Monte-Carlo techniques with estimates of vertical water motion. These latter estimates can be derived from a variety of methods in estuaries (Bostater 1987). Our current understanding needs to be dramatically improved in order to make better predictions as well as to improve scientific understanding concerning the effects of bottom topography and channel characterictics (including bottom stress), river inflow, wind stress and estuarine fronts upon vetical mass flux. In any event, existing scientific judgement suggests that vertical velocity scales are small (depending upon the transport process and the averaging period used).

Typical estimates (Officer 1975, Dyer 1975, Pritchard 1953) indicate velocity scales on the order of 0.1 near fronts to 0.0001 cm/sec in open water environments. Recent attempts to measure vertical velocites using acoustic doppler profilers (Bostater, et. al. 1987) indicate the difficulties involved, however this technique provides promise for future studies of the interaction of small scale physical and biological processes. In fact, a very important scientific question is: to what degree does living particle patchiness influence recent atempts to measure small scale physical processes that make use of acoustic doppler techniques, which are based upon acoustic particle backscatter ?

In any event, this estuarine process model starts with an initial distribution of particles. In this case the particles represent striped bass eggs and larvae with specified particle ages. The physical characteristics of the estuarine compartment includes parameters such as temperature, salinity, river inflow, turbulent eddy coefficient and wind stress. The simulation steps forward in time and the coupling of vertical swimming, settling velocities (for the egg stage) and water velocities are used to estimate the vertical particle motion. In this simulation example the results of two runs are indicated. First, the upper layer particles (near the surface) are repostioned from their uniform horizontal starting positon

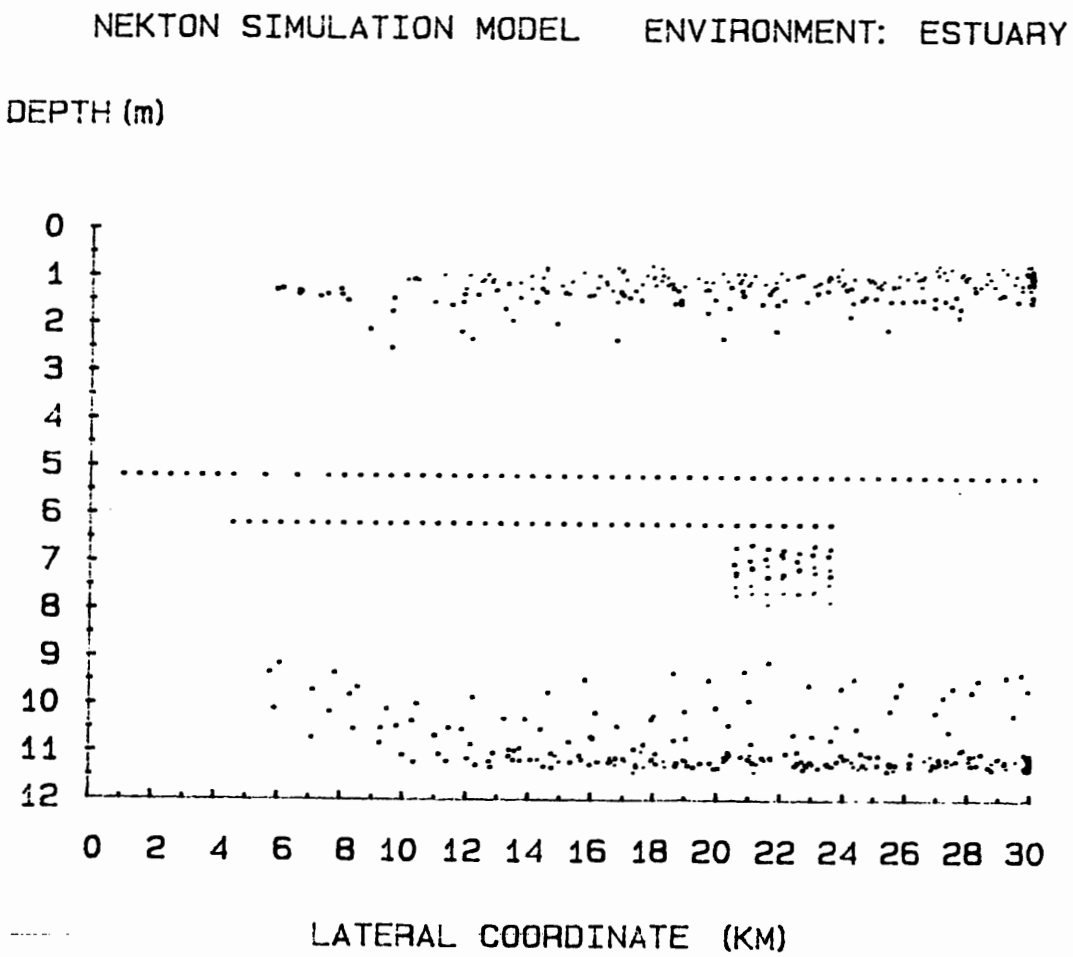

Figure 7. Example nekton simulation model output. Particles initially located at mid depth near 5 meters move upwards or downwards within the estuarine compartment depending upon the migration pathway specified. Lateral migration is also indicated as well as settling of eggs .

near 5 meters. The active particles numerically move (swim) upwards in the water column due to a stimulus such as light. Swimming can be initiated by defining the "quantum swimming yield" (Sq), a dimensionless number we define as a "threshold" for the swimming response. This ratio is the light level at a given depth to the light level that stimulates a swimming response. The ratio can be estimated from knowledge of downwelling light above the water surface and which is attenuated below the surface due to light attenuation processes which are a function of suspended sediment concentrations, inorganic and organic substances such as chlorophyll.

Figure 7 also indicates the change in the centroid of particle location. Particles are relocated very close to the bottom and to the right side of the estuarine compartment or segment. This response is again produced by defining a migratory pathway. We have also developed an algorithm which simulates swarming, simple schooling or "particle aggregation". This figure also demonstrates the lateral movement across the estuary which is produced by the horizontal swimming characteristics which can be conceptualized as larval or juvenile migration towards shallow water environments. The group of 8 particles in the middle of the figure demonstrate particles which only settle. The model can simulate more than one type of species or particle type.

Sensitivity Analysis

We are currently exploring optional modes of running the model processes to simulate particle motion in estuarine compartments or segments. We calculate a sensitivity analysis parameter from:

$$S(O,P)=\frac{(O'-Os)/Os}{(P'-Ps)/Ps}, \qquad (2)$$

where S, the sensitivity scale, is a measure of the change in vertical distribution of particles. O′ is the same measure for a particular model run and Os is the standard output (from a comparison run) of the same measure. P′ is a parameter value for a specified model run and Ps is the value of P for the standard or comparison run.

This parameter is helping us to explore various scaling and dimensional analysis functions derived from basic partial differential equations which can be used to estimate the relative importance of advective and diffusive processes in estuaries for passive and active particle dynamics. The current research has helped to derive a set of particle

transport scales which indicate the conditions that may control particle motion. These conditions include parameters which describe advection and/or turbulent eddy motion, amplitude of the tidal current, tidal (M2) height, topographic characteristics such as water depth, settling and active transport stimulated by environmental processes. Our results indicate a biophysical regime where nekton may be considered as being horizontally passive and vertically active when one scales a mathematical description of the conservation of probability. This regime is described by Pm, a dimensionless number.

SUMMARY AND FUTURE RESEARCH

The role nekton (swimming organisms) play in maintaining and sustaining economically important finfish and shellfish species cannot be over emphasized. This class of biota forms an ecological organization above the plankton. Established procedures have previously been developed by scientists for modeling various aspects of plankton in order to support the development of management strategies. This newly conceptualized area of process modeling and research may help to provide a missing link in the refinement of ecological models for practical use. This type of modeling can help to assess questions concerning: (a) early life stage distributions, (b) larval retention mechanisms, (c) the effect of living and non-living particles upon the irradiance reflectance ratio, (d) transport of particles in estuaries, and (e) statistical sampling protocols for sampling the vertical distributions of living particles in the water column. We have developed a method to couple circulation models with "active particles" and have shown how laboratory testing procedures (taxis studies) can begin to be implemented in order to investigate estuarine porcesses. We believe practical implications from this research will be utilized as 3 dimensional circulation models of Chesapeake Bay are applied in order to enhance our understanding of particle dynamics in estuaries.

References

Bartsch, J., 1987, Numerical Simulation of the Advection of Vertically Migrating Herring Larvae in the North Sea, Reports On Marine Research (Berichte der Deutschen Wissenschaftlichen Kommission fur Meeresforschung), December, Paper Submitted to AAAS Annual Meeting, February 12, 1988, Boston., 15 pp.

Burt, W. V., 1953, Extinction of Light by Filter Passing Matter in Chesapeake Bay Waters, Science, 118:386-387.

Bostater, C. R., Smullen, J., Biggs, R., 1987, Vertical Velocities Measured by an Acoustic Doppler Current Profiler and Comparisons to Analytical Models of Estuarine Circulation, EOS, 68(44):1305.

Clarke, G., Backus, R., 1956, Measurements of Light Penetration in Relation to Vertical Migration and Records of Luminescence of Deep-Sea Animals, Deep-Sea Research, 4:1-14.

Champ, M. A., et.al., 1980, Characterization of Light Extinction and Attenuation in Chesapeake Bay, August, 1977, V. S. Kennedy, ed., In: Estuarine Perspectives, Academic Press, pp. 263-277.

DiToro, D. M., 1978, Optics of Turbid Estuarine Waters: Approximations and Applications, Water Research, 12:1059-1068.

Dyer, K. R., 1973, Estuaries: A Physical Introduction, Wiley-Interscience, pp. 140.

Erikson, T. 1984, Adjustments in Annual Cycles of Swimming Behaviour in Juvenile Baltic Salmon in Fresh and Brackish Water, Trans. Amer. Fisheries Society, 113:467-471.

Kamykowski, D., 1976, Possible Interactions Between Plankton and Semidiurnal Internal Tides. II. Deep Thermoclines and Trophic Effects, Journal of Marine Research, 34:499-509.

Kamykowski, D., McCollum, S. A., Kirpatrick, G. J., 1988, Observations and a Model Concerning the Translational Velocity of a Photosynthetic Marine Dinoflagellate Under Variable Environmental Conditions, Limnol. Oceanogr., 33:766-78.

Huntley, M. E., Marin, V., Escritor, F., 1987, Zooplankton Grazers as Transformers of Ocean Optics: A Dynmaic Model, Journal of Marine Research, 45:911-945.

Knauss, J. A., 1978, Introduction to Physical Oceanography, Prentice Hall, p. 99.

Nelson, W., 1982, Applied Life Data Analysis, John Wley & Sons, New York, 634 pp.

Officer, C. B., 1976, Physical Oceanography of Estuaries, Wiley-Interscience, New York, 465 pp.

Pritchard, D. W., Kent. R. E., 1953, Tech. Rept. VI, Chesapeake Bay Institute, Johns Hopkins University.

Ripley, B. D., 1987, Stochastic Simulation, John Wiley & Sons, New York, 237 pp.

Setzler-Hamilton, E., Boynton, W. R., Mihursky, J. A., Polgar, T. T., Wood, K. V., 1981, Spatial and Temporal Distribution of Striped Bass Eggs, Larvae, and Juveniles in the Potomac Estuary, Trans. Amer. Fisheries Society, 110:121-136.

Sunby, S., A One-Dimensional Model for the Vertical Distribution of Pelagic Fish Eggs In the Mixed Layer, Deep-Sea Research, 30:645-661.

Understanding the Estuary: Advances in Chesapeake Bay Research. Proceedings of a Conference. 29-31 March 1988. Baltimore, Maryland. Chesapeake Research Consortium Publication 129. CBP/TRS 24/88.

Abstract only

Effects of Natural Environmental Fluctuations on Defense-related Oyster Hemocyte Activities

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Epizootic diseases caused by Haplosporidium nelsoni and Perkinsus marinus have inflicted serious mortalities on Chesapeake Bay oysters (Crassostrea virginica). Although present in the Virginia portion of the Bay for thirty years, recent drought conditions have allowed the agents to invade normally low salinity regions of Maryland.

Blood cells, or hemocytes, are the primary line of internal defense for oysters. To phagocytose and encapsulate parasites and disease agents, hemocytes must be able to recognize foreignness, spread to an ameboid shape, and locomote to the intruder. Hemocytes of oysters from estuarine and oceanic habitats differed in their defense-related activities after acute, short-term, and annual changes in salinity and temperature.

Monitoring and laboratory studies have detailed two major responses of hemocytes to acute salinity change regardless of ambient salinity; 1) increased salinity retards activities, and 2) decreased salinity has no effect until 12 ppt or less. Acclimation to a new salinity can require several hours or weeks, depending on the activity measured. Hemocytes from estuarine oysters in a low (1984) and a high (1985) salinity year were retarded with acute increases in salinity. Yet, acute decreases to 6 ppt retarded activity much more in 1985 than 1984, probably due to the higher ambient salinity. Oyster hemocytes from the oceanic habitat, with a consistent ambient salinity, responded identically in both years.

All hemocytes were more active after an acute increase in temperature, except that estuarine oyster hemocytes showed a summertime high-temperature stress (retardation) which was amplified when the oysters spawned in June. Oceanic oyster hemocyte activities were not retarded until they spawned in July, even though temperature regimes at both sites were nearly identical. This indicated that spawning and/or high temperature and low salinity conditions reduced defense-related activities of oyster hemocytes.

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Evidence for Loss of Suitable Benthic Habitats for Oysters in Tributaries of the Chesapeake Bay

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INTRODUCTION

During the 1980 and 1981 spawning and spat setting seasons of the oyster Crassostrea virginica , a combined study of the physical hydrography, oyster larvae distributions and phytoplankton distributions in the Choptank River system was made in order to understand the general mechanism for the transport and retention of planktonic oyster larvae in the tributary estuaries of the Chesapeake Bay (Seliger et.al., 1982). We were fortunate in that these years corresponded with atypically high concentrations of larvae and specific spat sets (spat per bushel of dredged oyster shell) in the seed bed areas of the Choptank River and its tributaries, Broad Creek and the Tred Avon River. The concomitant measurements of biological, physical hydrographic and meteorological parameters, combined with the focus upon quantification of the three dimensional distributions of the larvae of a single species, C. virginica , in a specific river system were essential to developing a description of the complex interplay of factors involved in the success of this species in the estuary.

A relation between water circulation patterns and the delivery and retention of meroplanktonic oyster larvae has been proposed and investigated by a number of authors (J.Nelson 1911; Pritchard 1951;1952; Manning and Whaley 1954). The fact that high concentrations of larvae and high spat set success were still possible in the river system, although the latter had not been observed over the previous decade (see Table 1 of Seliger et. al., 1982), suggested that oyster success in the system was much more sensitive to physical and climatic forcing functions than in the past. Based on older charts and anecdotal descriptions of the extents of oyster harvesting areas, there was an obvious and significant reduction of the areal extents where commercial oyster harvesting was being carried out. It appeared important to determine the relationship between high larval concentrations giving rise to high specific spat sets on the one hand and the availability of suitable bottoms which these successful planktonic stages could colonize and grow (ca 3 years) to mature adults. It was conceivable that in addition to the effects of eutrophication, toxic chemicals and oyster

Table 1. Summary of oyster bar areas in 1980 and in 1912

Bar No.	km2 in 1980	Bar No.	km2 in 1912	
		Chester River		
1	0	1		
2*	0.5	2	10	
3	0	3		
Total	0.5	Total	10	5% Remaining
		Broad Creek		
1	0.81	21	2.74	
2	0.1	20	2.07	
3*	0.91	19	1.48	
4	0.06			
5	0.02	18	0.97	
6*	0.36			
7*	0.11	17	2.33	
8	0.02	16	0.88	
9*	0.02	11	0.39	
10*	0.5	3	0.12	
Total	2.91	Total	10.98	26% Remaining
		Tred Avon River		
1*	0.38	22	1.49	
2*	0.08	21	1.07	
3	0.02	20	0.34	
4	0.03	19	0.82	
5	0.11	18	0.6	
6	0.1	17	0.74	
7	0.08	16	0.51	
Total	0.8		5.57	14% Remaining

* Site of State subsidized cultch deposition

Table 1 Summary of the previously measured areas of oyster bars (1912), the present (1980) shell areas, and the percent remaining, for the Chester R., Broad Creek and the Tred Avon R.

disease on the long term success of oyster harvests in the tributaries of the Chesapeake Bay, the gradual silting over of significant areas of suitable hard bottoms for oysters had resulted in an areal limitation of oyster production.

In the present paper we report a quantitative acoustical and SCUBA diver examination of oyster bed areas in the Broad Creek and Tred Avon R. tributaries of the Choptank R. in the central bay as well as in the Chester R. of the northern bay. We report major siltation over previously described oyster beds and a correlation between strong bathymetric gradients and the residual locations of exposed shell. Based upon this small sampling it appears that elimination of oyster beds by sedimentation may be the major cause of the decline of oyster populations in the Chesapeake Bay.

EXPERIMENTAL

The study areas of the Chester R. and the Choptank R. system are shown in Figure 1a and 1b respectively. The echo sounder was a Simrad EY-N operating at 70 kHz, developed for measurement of fish concentrations by quantitation of swim bladder echoes. It was calibrated for bottom sediment composition by comparison of recorder signals with samples both dredged and collected by SCUBA diver. The transmitter produced a 0.6 ± 2% msec pulse of 75 Watts with a cone angle of 27°. The receiver gain control had fixed steps of 3 ± 0.3 dB and an adjustable time-varied gain (TVG) which provided a linear increase in gain as a function of time subsequent to the initiation of the transmission pulse. The TVG had the effect of correcting for transmission losses and therefore of making echo trace intensities from reflecting surfaces independent of depth. The option of using "Dynaline" recording enables precise bottom discrimination by blocking strong echoes momentarily after the return from the sea bottom; the profile appears as a sharp line on the echogram followed by a white gap. A ceramic transducer, protected by a streamlined PVC blister, was suspended from the ship on a V-fin depressor which transmitted the pulse perpendicular to the bottom. The pitch and roll effects of a hull mounted system were thereby minimized. A portable oscilloscope (Telequipment D 32) was used to check signal levels and to avoid saturation f the output signals. The primary and echo pulses were recorded on magnetic tape simultaneously (2 channels) during the soundings.

The bathymetries for the Chester R. and the Broad Creek-Tred Avon R. study areas were determined from cross channel transects at 930m and 250m separations respectively. Each transect was initiated and terminated at the 2m depth contour. The transects were located by Loran C, Raydist coordinates (Radio Navigation System), radar fixes and compass bearings, and agreed with NOAA navigational charts of the areas. The ship speed and echo sounder chart speed were held constant throughout the transects and were checked separately against measured courses. Each sounding was made at constant gain and recorder settings.

During the transect anchor buoys were dropped overboard without stopping at 1-meter depth increments determined from the chart recorder and marked on the chart by an event recorder pen. Subsequent to the complete transect, bottom grab samples were taken at each buoy location with Van Veen and Ponar type samplers, depending upon the hardness of the bottom. The collected sediments were mixed and stored in labelled jars for silt/sand analysis. The silt/sand compositions of the sediments were determined with a 63 μm mesh (Tyler No.230) sieve. Sand was arbitrarily defined as particles > 63 μm in any 2-dimensions (Wentworth 1922). A uniformly mixed portion (10--20 g) of each grab sample was sieved until the distilled water used for washdown remained clear. The silt and remaining sand were transferred to pre-weighed beakers, dried at 76° for 2-3 days, then re-weighed. The horizontal scale for plotting bathymetry was calculated from the ship speed and the recorder chart speed and converted to 1:40,000 horizontal scale for contour plotting. Observations of locations of commercial oyster tongers were marked, as well as locations of oyster dredge sampling.

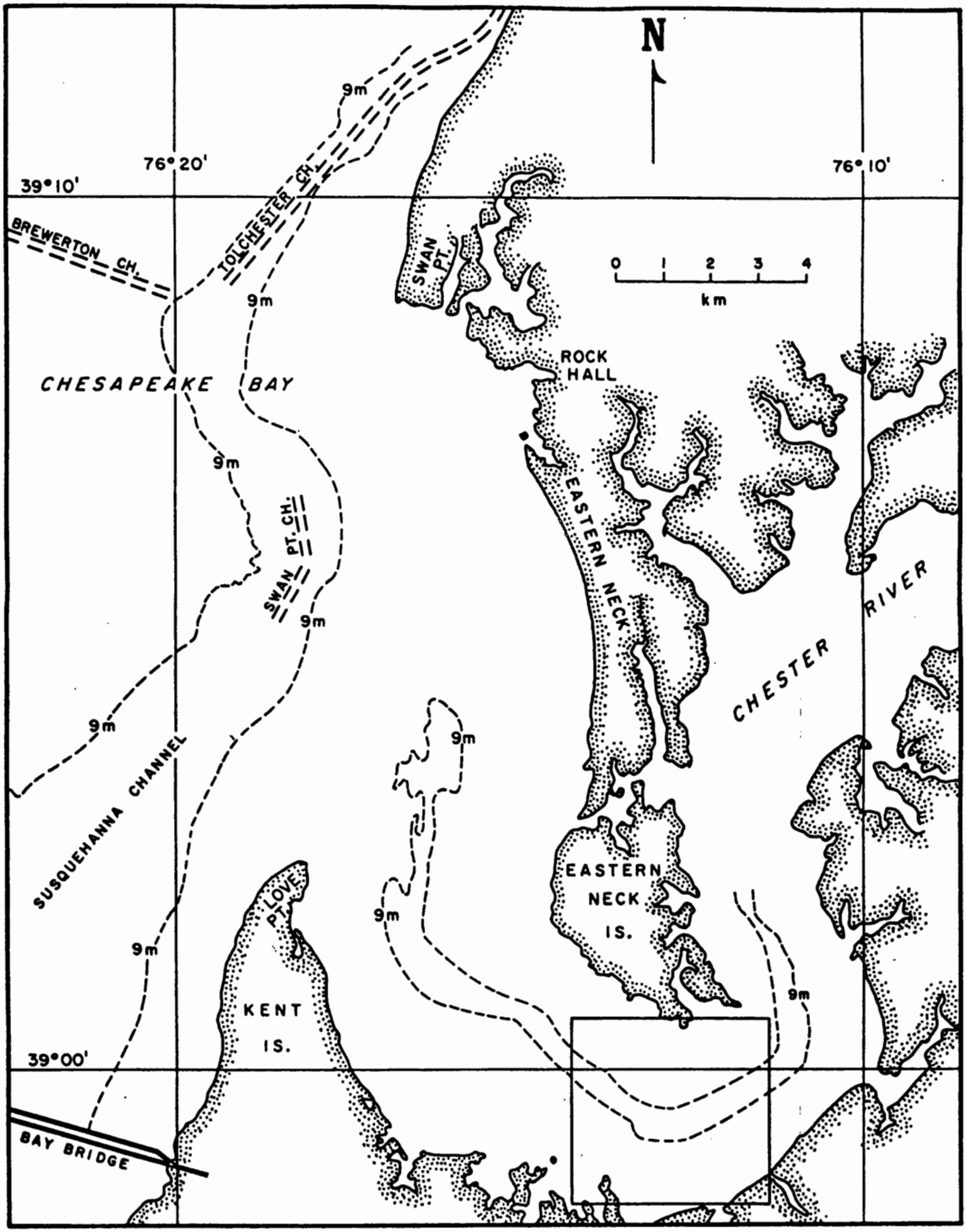


Fig. 1(a) The Chester River System, with Study areas enclosed is boxes.

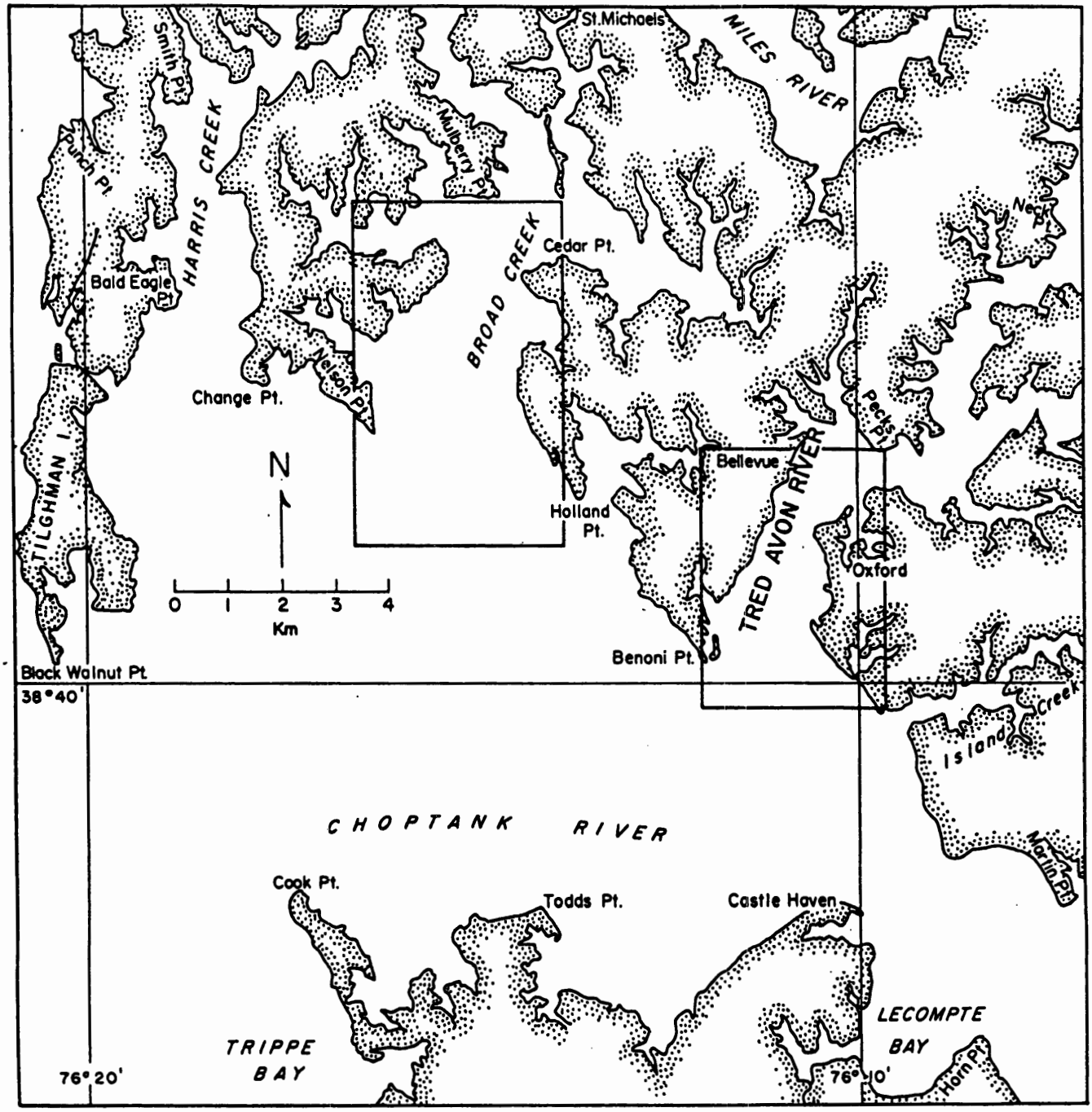


Fig. 1(b) The Choptank River System, with study areas enclosed is boxes.

The areal determinations of the previously defined oyster bars and the present shell beds were measured from the contour charts with a Graf/Pen, GP6-50, (Science Accessories Corp.) digitizer. Bathymetry gradients were calculated from isobaths (depth,z) by measuring the projected distances (r) normal to the isobaths, expressing the gradient as (dz/dr) x 103.

RESULTS

Calibrations were carried out over an active oyster bar in the Chester R. (Buoy Rock Bar) whose dimensions are shown in Figure 2a. S--->N and W--->E echogram transects were made directly across the oyster bed (Figure 2b) and a S--->N echogram transect just west of the oyster bed (Figure 2c). The water surface and the bottom depths are indicated by the arrows labelled "Surface" and "Dynaline", respectively. From Figure 2b the echograms of hard sand and soft mud relative to oyster shell can be distinguished in terms of the densities of their respective acoustic returns. Viable oyster beds can also be identified by the distinctive light space in the secondary reflection, shown by the rectangles. The echogram in Figure 2c shows the absence of shell bed. The gradations in densities of acoustic returns were correlated with silt/sand or mud composition of sediments determined by physical sampling and more precisely by SCUBA diver. In our small study area which contained one of the two remaining commercially active oyster bars in the Chester R., less than 0.5 km^2 of oyster shell area, of an original 10 km^2 remain (Yates 1912; Merritt 1977). The original extent of Buoy Rock Bar, 2.6 km^2 has been reduced to 0.5 km^2. From acoustic assay and corroborated by SCUBA diver, Blunt Point Bar (SE of Buoy Rock Bar) and Long Point Bar (SW of Buoy Rock Bar) have been completely silted over. The maximum depth for the oyster bar appears to be correlated with the depth of the bay pycnocline during summer (Seliger et.al., 1980; 1984), because of anoxia that develops in waters below the pycnocline during summer.

Two W--->E cross channel echogram transects, 500m apart, encompassing an actively worked oyster bar (Great Bar) in Broad Creek are shown in Figure 3a and 3b, illustrating from west to east: a) mixed silt/sand (thin primary and secondary traces); b) soft mud (broad primary trace); c) mixed silt/sand; d) shell bed (white space, see rectangle); broad tertiary trace); e) location of a commercial oyster tonger and f) hard sand (thin secondary and tertiary traces) on the eastern shore.

The locations and extents of present exposed oyster shell bars in Broad Creek and the Tred Avon R. are shown by the cross hatched areas in Figure 4a and 4b respectively. The dotted lines represent the 6 foot (ca. 2m) and the 18 foot (ca 6m) depth contours. Peripheral areas where sediment has partially encroached upon shell, i.e., Bed 1 of Figure 4a and Bed 7 of Figure 4b, are labelled "patchy". Areas of soft mud are usually at depths > 6m. In general the beds follow the 6 foot (2m) depth contour and, where they extend into deeper water, are sharply delineated by the 18 foot (6m) depth contour.

The present oyster shell locations (cross hatched) in the Broad Creek and the Tred Avon R., together with the areas where the bathymetric gradient (Dz/Dr) x 103 (solid black) are shown in Figure 5a and 5b respectively. In most instances these solid black areas are contiguous with the locations of present oyster shell areas. The only saving feature for the presently remaining area (18%) of Buoy Rock Bar is the very steep bathymetric gradient at its southern end (not shown in the figures). The previous areas of active oyster beds in these three tributaries measured by Yates (1912) are shown in Figure 6a, 6b and 6c. Table 1 summarizes the previous bars, the present areas in km2 and the percent of shell area remaining. In the Chester R. and the Tred Avon R., oyster bed areas have been reduced to 5% and 14% respectively, of their previous measured extents.

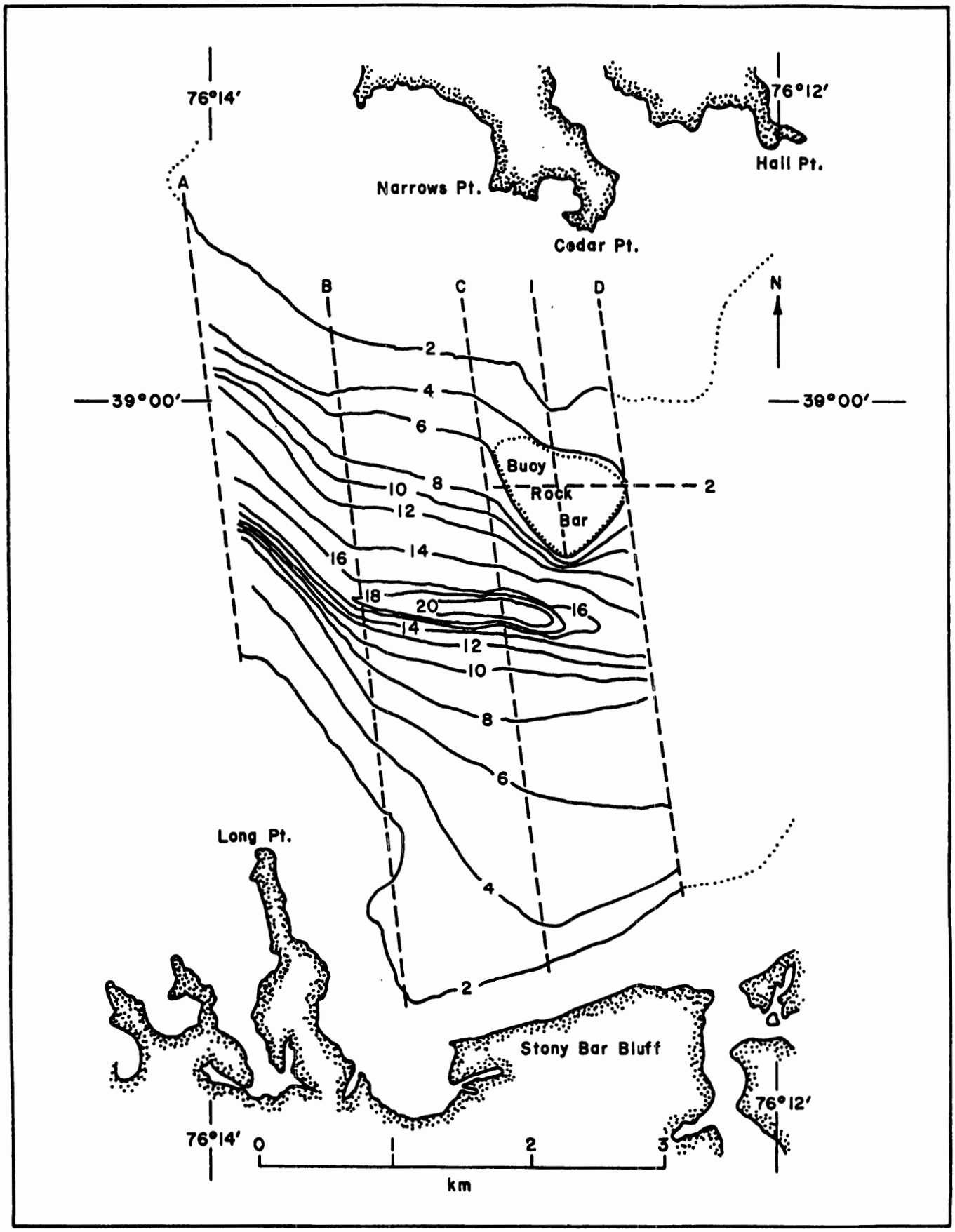


Fig. 2 (a). Chart of the Chester R. study area showing the location of Buoy Rock Bar. The bathymetry measured is shown in 2m increments (solid lines). The transects occupied are shown as dashed lines.

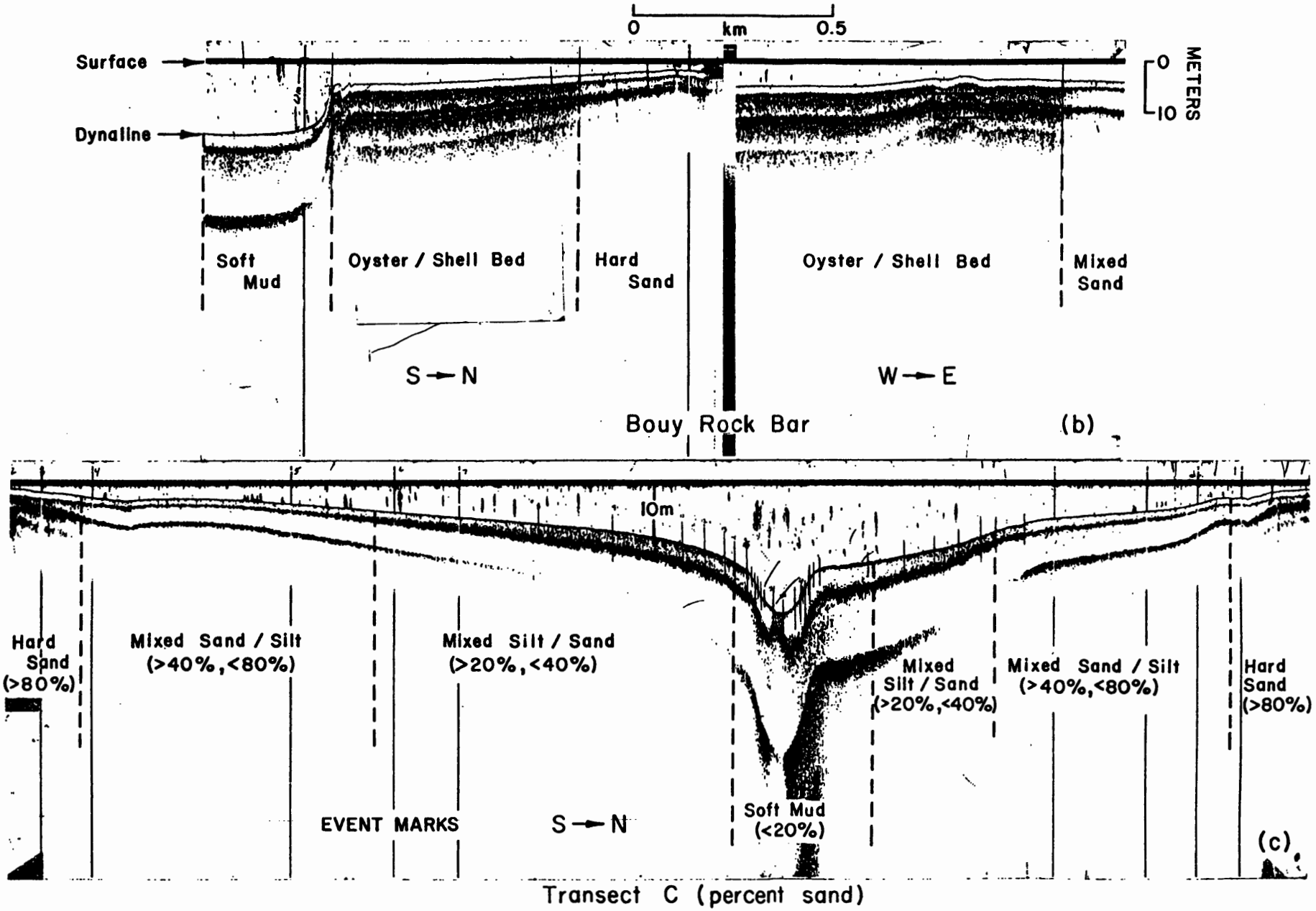


Fig. 2 The transects occupied are shown as dashed lines. (b) Sonograms from transects 1 (S--->N) and 2 (w--->E) across Buoy Rock Bar. (c) Sonogram from transect C (S--->N) across the Chester R just west of Buoy Rock Bar. In both (b) and (c) the bottom compositions corresponding to the sonogram tracings were determined by grab sampling and percent sand from wet sieve analysis.

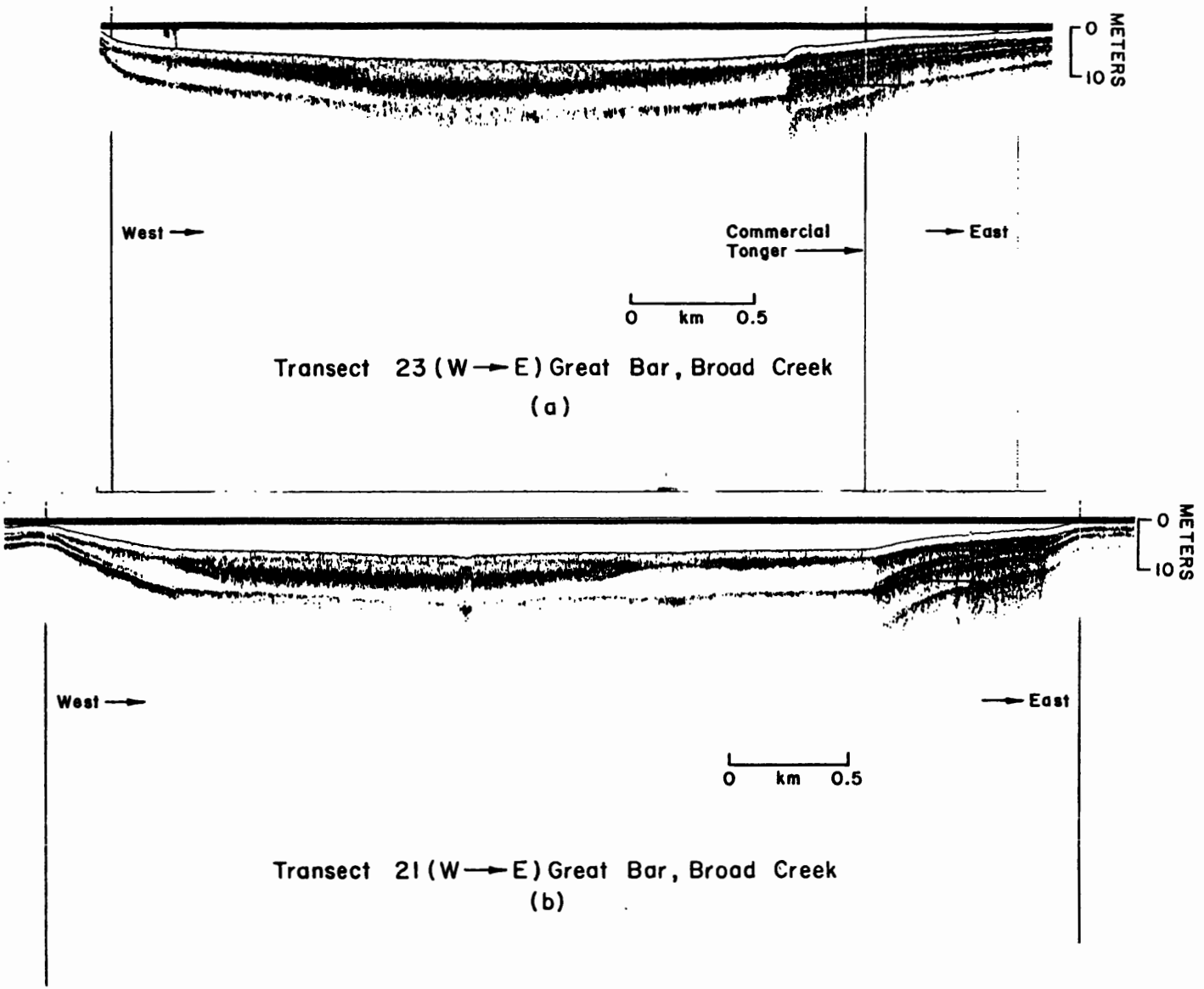
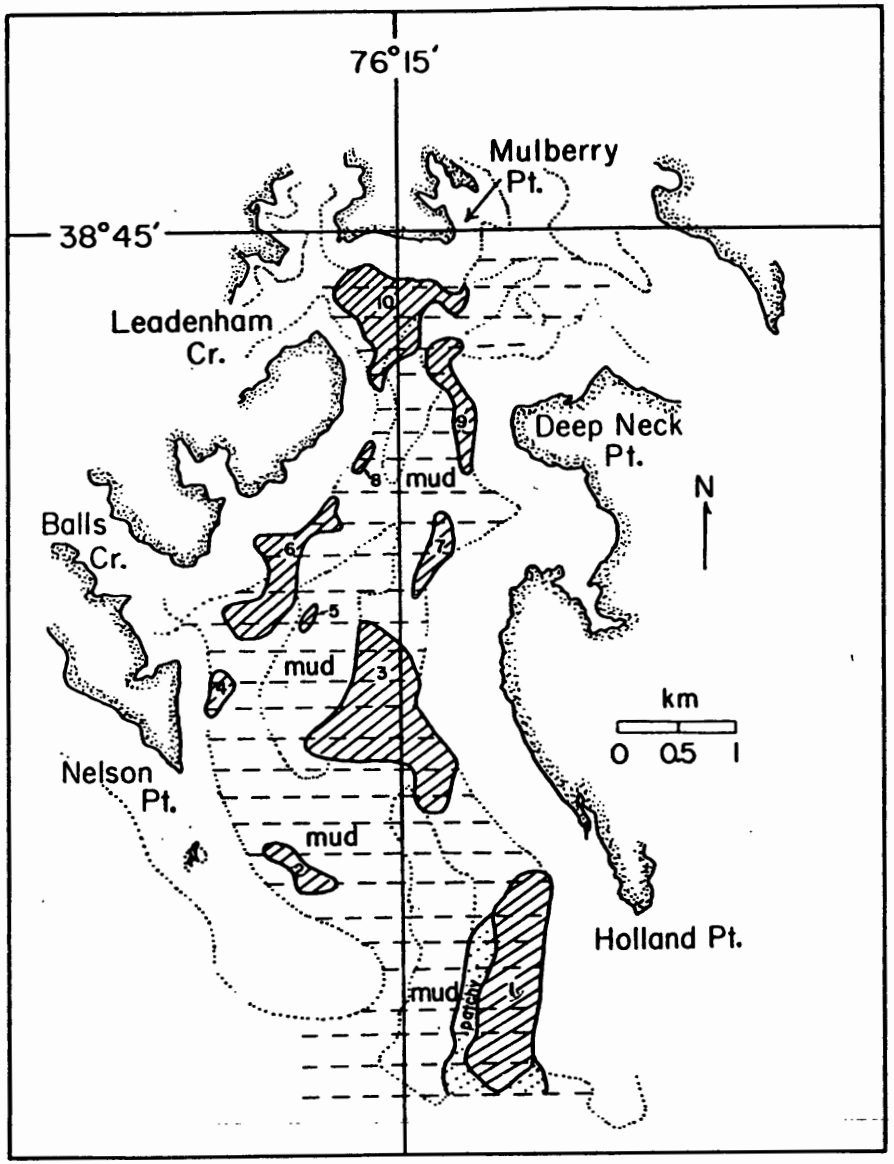
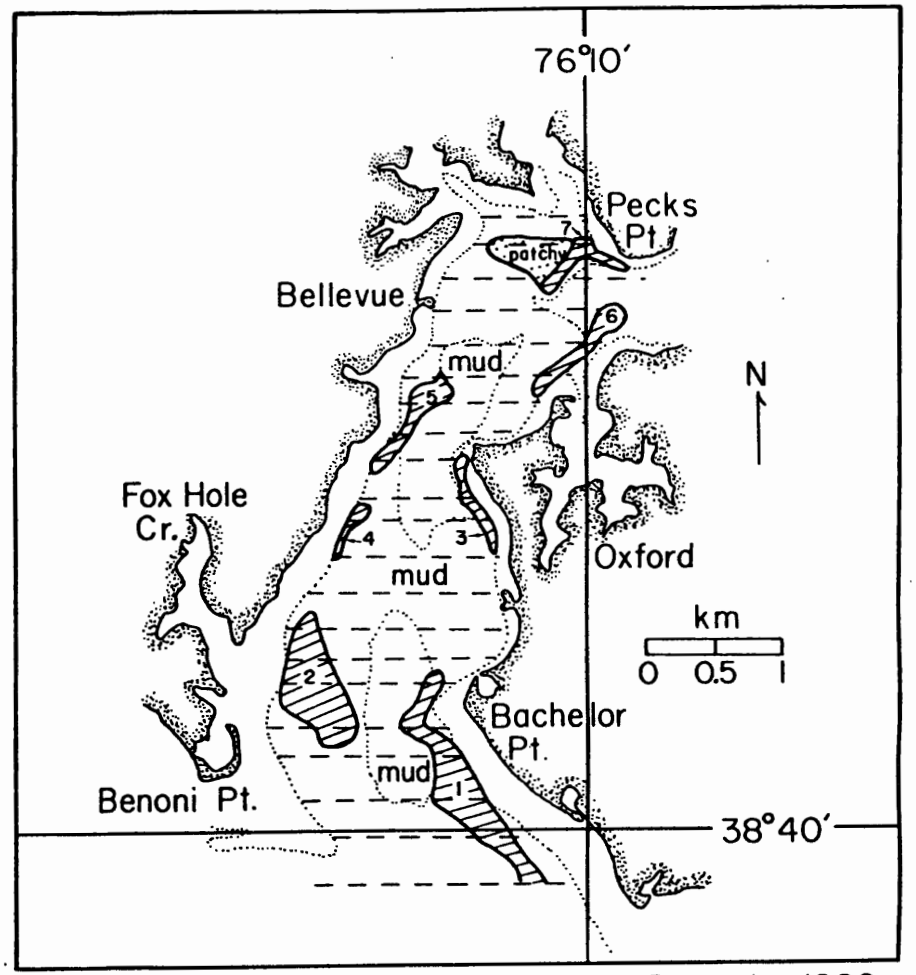


Fig. 3 (a) Sonogram from transect (W--->E) over Great Bar in Broad Creek showing bottom compositions (from left to right in the figure) as mixed silt/sand, soft mud, mixed silt//sand, oyster bed shell (commercial tonger at event mark) and hard sand inshore of the shell bed. (b) A parallel transect 500m upstream of the transect in (a) above with sonar responses {see enclosed rectangles in both (a) and (b)} characteristic of viable oysters.

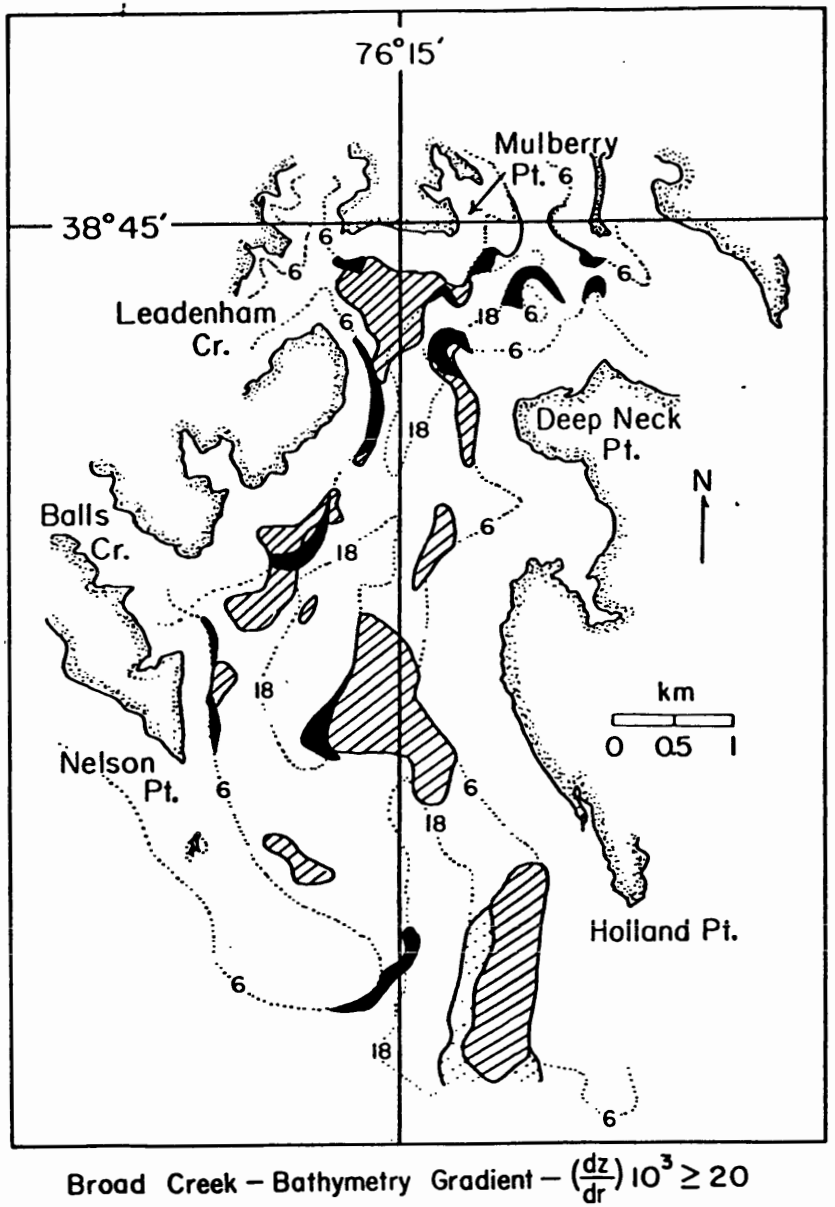


Broad Creek - Shell-Bed Locations - December 1980

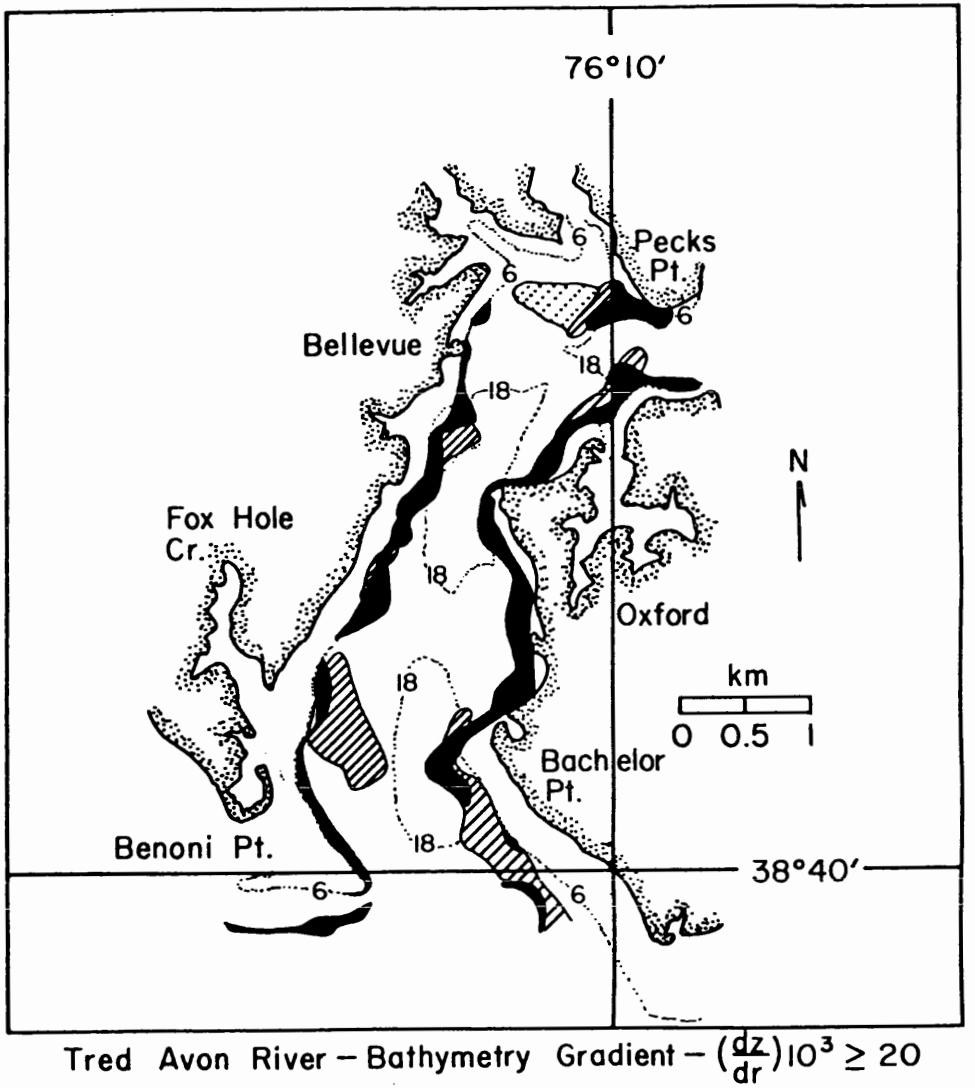


Tred Avon River - Shell-Bed Locations - December 1980

Fig. 4 Charts of the present locations of shell beds as cross hatched areas (a) in Broad Creek; (b) in the Tred Avon R.



Broad Creek – Bathymetry Gradient – $(\frac{dz}{dr})10^3 \geq 20$



Tred Avon River – Bathymetry Gradient – $(\frac{dz}{dr})10^3 \geq 20$

Fig. 5 (a) and (b) Charts of the same shell bed areas as in 4(a) and 4(b) except that superimposed upon the cross hatched shell bed areas are solid black areas representing the locations of bathymetric gradients where dz/dr x 103 was greater than 20.

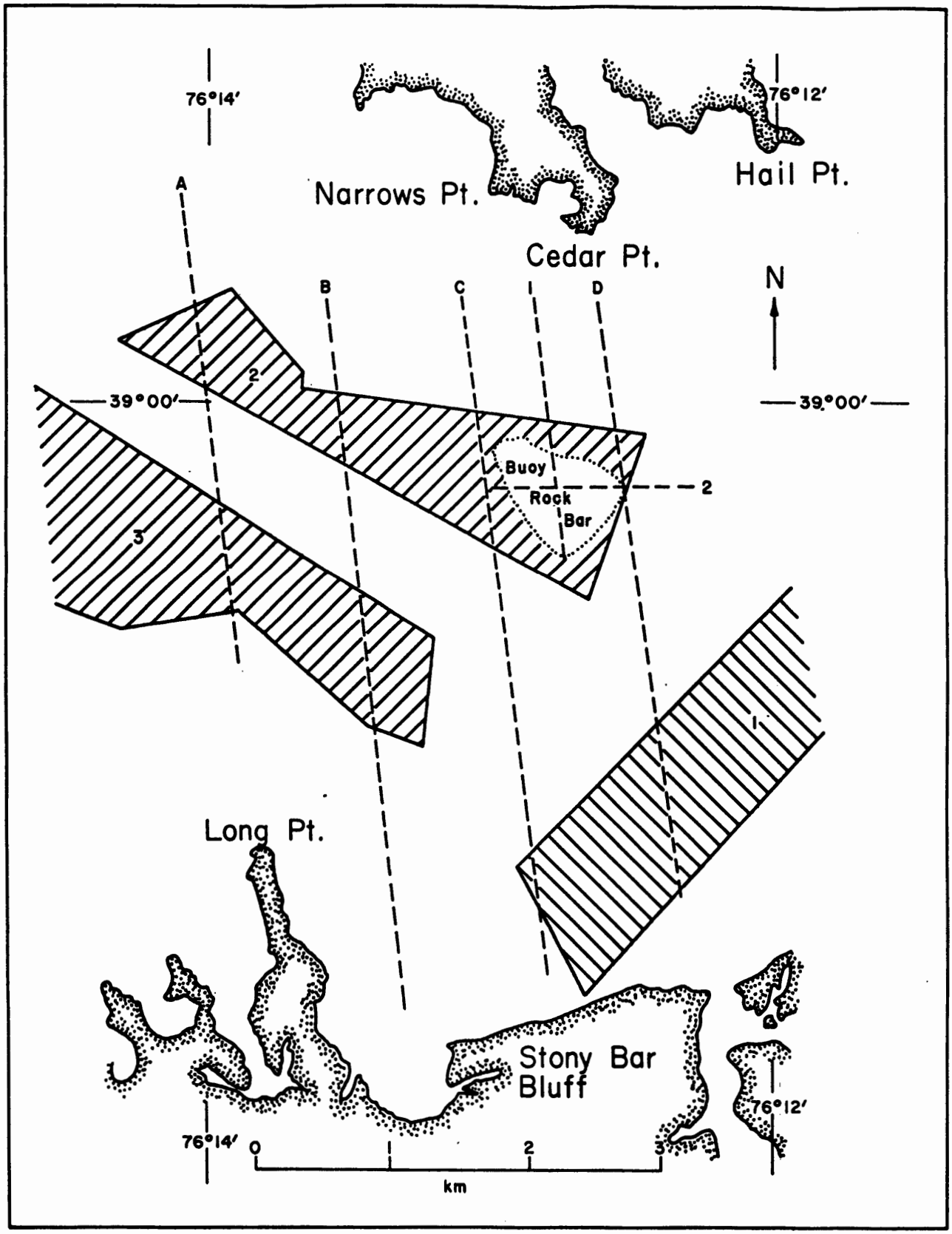


Fig. 6(a) Previously measured extent of oyster bars on the Chester River.

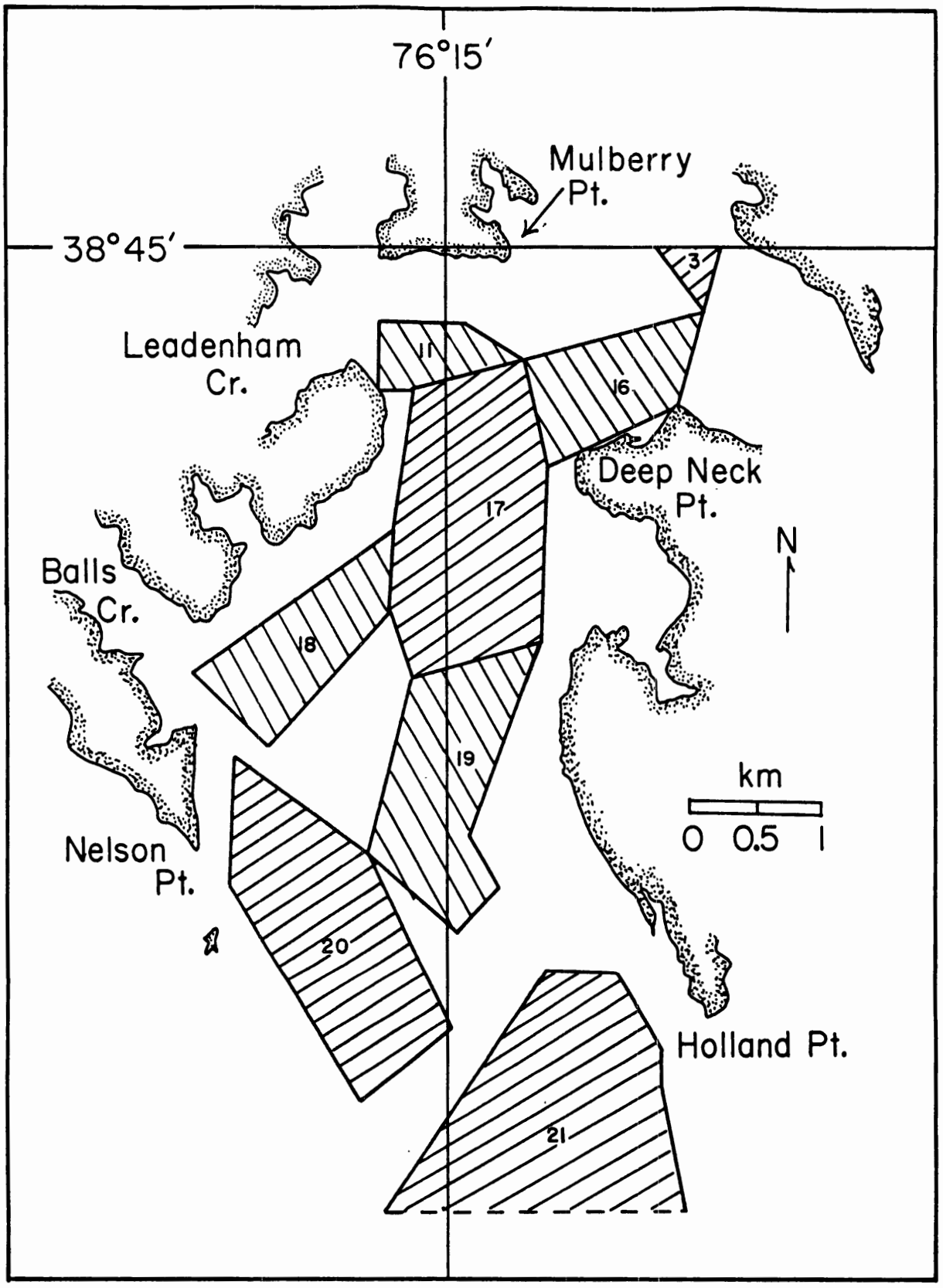
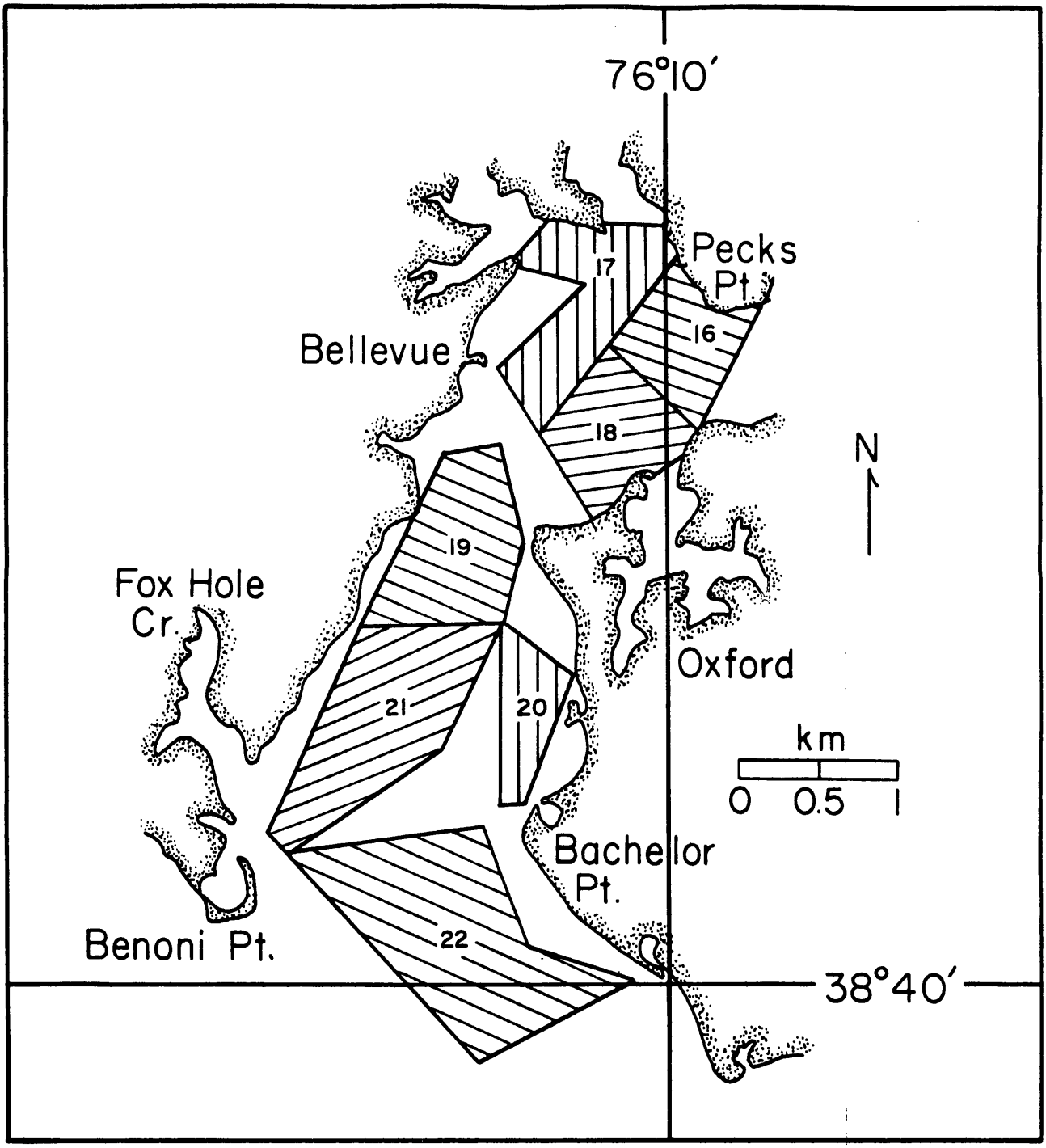


Fig. 6(b) Previously measured extent of oyster bars on the Broad Creek.



Tred Avon River - Oyster - Bar Locations - Yates, 1912

Fig. 6(c) Previously measured extent of oyster bars on the Tred Avon River.

DISCUSSION

It is obvious that biotic as well as abiotic factors influence the success of sessile oyster populations and their planktonic and meroplanktonic life cycle stages. Only a small portion of these factors has been addressed in the present paper. From the success of the planktonic and settling stages in 1980 and 1981 it appears that water circulation patterns and food sources for the larval stages were still adequate. The successful commercial re-harvesting in the early 1980's of seed oysters "planted" in previous years on both Buoy Rock Bar and Ferry Bar through the State-subsidized re-seeding program is inferential evidence that, barring climatic extremes, there were still sufficient food sources for the growth of adult oysters. It appears therefore that for each of the oyster bars investigated, sedimentation (the covering over of previously viable oyster beds) has been the major factor in the reduction of oyster harvests. Many formerly productive oyster bottoms along the Atlantic coast of the United States have been destroyed by high sedimentation rates (Galtsoff 1964). The filling of embayments with sediments is a general phenomenon along the Texas coast and is particularly pronounced in Laguna Madre and near the Colorado R. in Matagorda Bay, where approximately 24-28 km^2 of oyster reefs are under 14 feet of mud (Norris 1953). In the York R. in the Chesapeake Bay faulty erosion control practices,deforestation and population growth were identified 50 years ago as increasing erosion of soil from the watershed into the river (Brown et. al., 1939). In the upper Chesapeake Bay aproximately 38000 m^2 of shoreline (0.33×10^6 metric tons) are eroded and enter the bay annually (Schubel 1968).

The data reported in this paper represent the application of a rapid profiling and bottom composition analysis technique to identification and areal assay of oyster beds. Our previous physical hydrographic and biological study of transport and retention of oyster larvae (Seliger et.al. 1982) indicates that the planktonic life cycle stages of oysters have not been affected as much as the sessile benthic stages. Silting over of upstream seed bed areas and viable oyster beds along the shorelines may be the major factor in the loss of oyster harvests in the Chesapeake Bay. The study has not been applied to examine conditions outside of the mouths of the tributaries. However from the high larval concentrations and high spat sets measured in the Choptank R. system in 1980 and 1981, and from the absence of expected mature oyster harvests in the years following the high spat sets (1983, 1984), it appears that a limiting factor to oyster production in the tributaries of the Chesapeake Bay has been the silting over of suitable hard bottoms.

It is apparent from Table 1 and comparisons of Figures 2a,4a and 4b with Figure 6 that there have been severe reductions in all three tributaries in exposed shell areas suitable for the setting of spat. The pattern of sedimentation and consequent reduction of exposed shell bed area is most obvious in the Tred Avon R.)Figure 4b), where the entire central channel is now essentially soft mud. The present percentages of shell bed remaining are actually higher than they would be in the absence of the Oyster Management Program of the Maryland Department of the Environment. It has annually subsidized extensive deposition of fossil oyster shell cultch in all three tributaries in order to assist commercial harvesters.

The association of the remaining shell areas with steep bathymetric gradients (Figures 5a and 5b) is presumed to represent areas of highest tidal frictional turbulence and therefore the areas least likely to be impacted by siltation. The locations of the steep bathymetric gradients also represent areas where tidal shear fronts are observed. It follows that the areas immediately contiguous to these steep bathymetric gradient areas (Figure 5) would represent ideal locations for the deposition of artificial and fossil cultch, in order to increase the areas of oyster beds with minimum siltation effects. Since the major streamflow and sediment runoff into the tributaries occurs during the spring freshets, it would appear more efficient to deposit cultch material for new beds during early June, just

prior to the oyster spawning season, in order to minimize siltation on these new surfaces for spat settling.

In a companion paper in this volume (Seliger and Boggs, "Long Term Pattern of Anoxia in the Chesapeake Bay"), we present an hypothesis that excess sedimentation in spring runoff results in extremes of light limitation in the upper (Maryland) portion of the bay. This inhibits photosynthesis and thus nutrient assimilation, permitting runoff nutrients in the surface plume to be delivered to the southern (Virginia) portion, where blooming occurs. A similar process occurs in each of the tributaries of the bay; excess sediment results in a transfer of production to downstream regions. Concomitant with the effects upon primary production in the water column this same sediment loading results in the progressive deposition of sediment upon and the loss of upstream seed bed areas for oysters and the shallow areas outside of the mouths of the tributaries. These areas are specifically the regions of the tributaries which are not subject to transient incursions of anoxic bottom waters from the central bay during summer. In addition the salinities in the upstream seed bed regions of the Maryland tributaries are low, minimizing the incursions of high salinity-requiring oyster diseases. The loss of the upstream seed bed areas by siltation removes from the system the resevoir of potential production, i.e., of larvae for re-colonization subsequent to extremes of anoxic events or of disease. The loss by siltation of the shallow oyster beds contiguous with the shorelines removes from the system the mature oyster resevoir least impinged upon by anoxia of bay bottom waters. What remains are oyster bed areas in the deeper waters, areas which are most sensitive to anoxic incursions when, as in 1984, anoxia in bay bottom waters rose to a depth of 6 meters (Seliger et. al., 1985). Under these conditions oyster populations are not replenished sufficiently even following years when spawning and setting conditions are excellent. Much of this sediment deposition is irreversible. Management procedures for optimizing oyster spat settlement during good years can be inferred from the paper: a) implementation of severe restrictions on further sedimentation; b) building up of cultch and artificial hard bottoms in specific contiguous areas of steep bathymetric gradients; c) deposition of both cultch and seed oysters after the spring freshets, thus avoiding major siltation; d) utilization of acoustic techniques for following the progress of oyster bed replenishment.

Brown, Carl B., Seaney, L.M. and Rittenhouse, G. Advance report on an investigation of silting in the York River, Virginia, October 25-November 5, 1938. Sedimentation Studies, Division of Research, SCS-SS-32, U.S. Department of Agriculture, Soil Conservation Service, Washington, D.C. 12 pp (1939).

Galtsoff, Paul S. The American Oyster, *Crassostrea virginica* (Gmelin). Fishery Bulletin **64:** 480 pp (1964).

Manning, J.H. and Whaley, H.H. Distribution of oyster larvae and spat in relation to some environmental factors in a tidal estuary Proc. Natl. Shellfish Assoc. **45**, 56-65 (1954).

Merritt, D.W. Oyster spat set on natural cultch in the Maryland portion of the Chesapeake Bay (1939-19975). UMCES Special Report No. 7 (1977).

Nelson, J. Report of the Biologist. Ann. Rept. Dept, Biol. for 1910, N.J. Agric. Coll. Exper. Sta. pp183-218 (1911).

Norris, R.M. Buried oyster reefs in some Texas bays. Journal of Paleontology **27** (4): 569-576 (1953).

Pritchard, D.W. Distribution of oyster larvae in relation to hydrographic conditions, Proc. Gulf and Caribbean Fish Inst. Nov. 1952 pp 1-10.

Pritchard, D.W. The physical hydrography of estuaries and some applications to biological problems, Trans. 16th N. Amer. Wildlife Conf. Wildlife Management Inst., Wash. D.C. pp 368-376 (151).

Schubel, J.R. Suspended sediment of the northern Chesapeake Bay. Chesapeake Bay Institute Technical Report 35 Reference 68-2 (1968).

Seliger, H.H., Boggs, J.A. and Biggley, W.H. Catastrophic anoxia in the Chesapeake Bay in 1984, Science **228**, 70-73 (1985)

Seliger, H.H., Boggs, J.A., Rivkin, R.B., Biggley, W.H. and Aspden, K.R.H. The transport of oyster larvae in an estuary, Marine Biology **71**, 57-72 (1982).

Seliger, H.H., McKinley, K.R., Biggley, W.H., Rivkin, R.B. and Aspden, K.R.H. Phytoplankton patchiness and frontal regions, Marine Biology **61**, 119-131 (1981).

Wentworth, C.K. A scale of grade and class terms for clastic sediments, Journal of Geology **30**, 377-392 (1922).

Yates, C.C. Surveys of oyster bars, U.S. Coast and Geodetic Survey Publication, U.S. Government Printing Office (1912).

Understanding the Estuary: Advances in Chesapeake Bay Research. Proceedings of a Conference. 29-31 March 1988. Baltimore, Maryland. Chesapeake Research Consortium Publication 129. CBP/TRS 24/88.

Stabilized Coal Ash as Substratum for Larval Oyster Settlement: A Pilot Field Study

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INTRODUCTION

Chesapeake Bay is one of the primary oyster production areas in the United States. In recent years oyster production has dropped dramatically due to disease, poor water quality (eutrophication, pollution), and a lack of suitable substrata for oyster spat set. This study funded by Baltimore Gas & Electric Company and the Delaware Research Partnership examines the feasibility of using stabilized coal waste materials in the form of pucks as a suitable substratum for oyster spat set and growth.

METHODS

Stabilized coal-ash was constructed by the engineering firm of KBK, Inc. of Atlanta, Georgia. Bottom and fly ash were supplied to KBK by Baltimore Gas and Electric Company; the stabilized ash was shaped in the form of 3-inch and 6-inch diameter discs approximately two inches thick and allowed to cure. The substrata was then shipped to each of two sites: 1. the College of Marine Studies, University of Delaware in Lewes for placement in the Broadkill River, and 2. the eastern shore of Maryland for placement in the Little Choptank River and Broad Creek, prime oyster producing tributaries of the Chesapeake Bay. Stabilized ash and oyster shell control substrata were placed at the Maryland sites in early July, 1987; the Delaware portion was placed in holding tanks to which hatchery reared oyster larvae were added. After oyster settlement was ascertained, the Delaware substrata were placed in the Broadkill River.

The Chesapeake study was conducted at two sites in the eastern Chesapeake Bay area, one in Broad Creek near Neavitt, Maryland, and the other in the Little Choptank River near Madison, Maryland. Both sites are known to be oyster spawning grounds and were recommended by Dr. George Krantz, Maryland Department of Natural Resources in Oxford, Maryland.

Two substrata were used at each site to study oyster spat set; 1) coal ash/concrete 6 and 3 inch diameter pucks, and 2) oyster shell. Coal ash/concrete pucks were made of a mixture of bottom ash, fly ash, and cement in a ratio of 59:33:8. Oyster shell was dredged fossil oyster shell provided by the Langenfelter Company which supplies the State of Maryland shell for replenishing oyster setting areas.

At each site three plots were set up. Each plot contained nine squares 2 meters by 2 meters, and each plot was separated by the length of one plot (Figure 1). Coal ash pucks and oyster shell were the treatment variables in this replicated randomized block design, and each variable appeared once in each block. The "natural bottom" squares were included in the design to provide access to the "ash" and "shell" squares and were not sampled.

The replicated randomized block design was used here to account for any variability between blocks possibly due to currents, patchiness of oyster larvae in the water column, bottom differences, etc. (thus our one random blocking effect might be any of these unknown factors).

The same replicated randomized block design was used at both sites and each square was coded for identification of sample origin. Plots of both sites were measured and staked out before substrata planting. Substrata were planted at the Broad Creek site on 7/2/87 and at the Little Choptank site on 7/7/87.

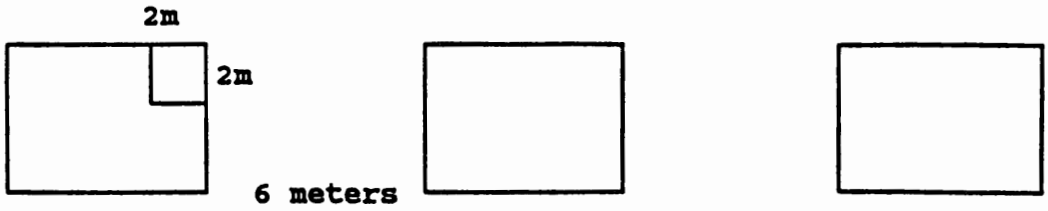
Substrata were ferried to the sites in burlap sacks by the oyster boat "Miss Molly" and transferred to two motor boats as needed for planting. To avoid confusion one boat planted only oyster shell and the second only pucks. Sacks of the substrata were opened and poured from the boats evenly into designated squares and each square was raked with a clam rake to ensure even substratum deposition.

Twelve sacks of oyster shell were deposited in each "shell" square, and six sacks of 6" pucks and six sacks of 3" pucks were deposited in each "puck" square. Sacks were chosen haphazardly for depositing, and the order in which the squares were planted was also haphazard. Three sample bags of each substratum (chosen haphazardly) were volumed to determine the approximate amount of substratum deposited in each square.

Four 10 cm x 10 cm asbestos plates were hung at each site as an indicator of intensity of spat set.

Sampling of substrata began on 7/15/87 and continued on a weekly basis until 8/19/87 after which sampling was monthly until 11/16/87 when sampling ended for the winter. One 6" puck was sampled from each "puck" square and at least four shells from each "shell" square.

Figure 1: Arrangement of the Latin Square design plots at both sites on the Chesapeake Bay.



Coded Latin Square design plots for the Broad Creek site and the Little Choptank site.
nb= natural bottom

Broad Creek Site

1.1.1 puck	1.1.2 shell	1.1.3 nb
1.1.4 shell	1.1.5 nb	1.1.6 puck
1.1.7 nb	1.1.8 puck	1.1.9 shell

1.2.1 puck	1.2.2 nb	1.2.3 shell
1.2.4 nb	1.2.5 shell	1.2.6 puck
1.2.7 shell	1.2.8 puck	1.2.9 nb

1.3.1 nb	1.3.2 puck	1.3.3 shell
1.3.4 puck	1.3.5 shell	1.3.6 nb
1.3.7 shell	1.3.8 nb	1.3.9 puck

Little Choptank Site

2.1.1 puck	2.1.2 shell	2.1.3 nb
2.1.4 shell	2.1.5 nb	2.1.6 puck
2.1.7 nb	2.1.8 puck	2.1.9 shell

2.2.1 puck	2.2.2 nb	2.2.3 shell
2.2.4 nb	2.2.5 shell	2.2.6 puck
2.2.7 shell	2.2.8 puck	2.2.9 nb

2.3.1 nb	2.3.2 puck	2.3.3 shell
2.3.4 puck	2.3.5 shell	2.3.6 nb
2.3.7 shell	2.3.8 nb	2.3.9 puck

Substratum from each square was placed in a coded Ziploc plastic bag, sealed, and placed in ice in an ice chest and delivered to the College of Marine Studies, University of Delaware, Lewes, Delaware for analysis.

Each substratum was sampled haphazardly from its square by diving until 11/16/87 when sampling was done from the boat using clam rakes. Sampling of the squares was according to the code of the sites (square 1.1.1. sampled first, then 1.1.2., etc.). On each sampling date the spat collectors were retrieved and stored in cool bay water and brought to the lab for analysis and replaced by clean spat collectors until 9/15/87, the end of the spawning season.

Analysis of the samples consisted of examining both flat sides of the puck (the rim surface was not included in the analysis) and both sides of a shell under an American Optical stereo microscope at 14x to 84x and noting oyster spat and measuring spat length from hinge to outer lip to the closest millimeter. Beginning with the 9/15/87 samples dead oyster spat were noted separately from live oyster spat. Length and width of each shell substrate was measured to the nearest millimeter and spat set per square centimeter was calculated. For direct comparison spat set per square centimeter was also calculated for "puck" surfaces and asbestos spat collectors. Density data (spat per centimeter squared) was loaded into LOTUS worksheets and analyzed statistically using STATGRAPHICS and MYSTAT on an IBM PC computer.

Oysters from both the Maryland eastern shore sites and the Delaware site were harvested for metals analysis from the coal-ash pucks and oyster shell controls, frozen at -70 C, freeze-dried for 48 hours and ground to powder consistency. Samples were then prepared for analysis via atomic absorption spectroscopy as follows: sample aliquots were weighed out, pre-digested in 10 ml concentrated nitric acid at room temperature for 24 hours, digested at 65 C for 4 hours and diluted to 50 ml with 1% nitric acid. Digests were then filtered through quantitative ashless filters and analyzed for metal concentrations using a Varian SpectrAA-20 atomic absorption spectrophotometer (flame mode). Blanks were included in the digestion procedure; no standard oyster reference material was available from the National Bureau of Standards therefore no outside tissue standard was included in these preliminary analyses. We are attempting to find a source of standard tissue for future analyses. Where necessary, samples were appropriately diluted to a measurable concentration.

Data Analysis

Two-way mixed model ANOVAs were computed for the Broad Creek data and the Little Choptank data separately with "substratum" as the fixed factor and "block" or "plot" as the random blocking factor (Sokal and Rohlf, 1981). Least significant differences and residuals of the means of the factors in the ANOVA were plotted. All statistical tests were run at an alpha level of 0.05.

RESULTS

Statistical comparisons of setting intensity on shell versus coal ash show that shell is slightly but not overwhelmingly selected for setting by oyster larvae. Although the setting density is about 2:1 in favor of shell the results are not statistically different based on samples collected during the summer and on November 16, 1987 (Table 1). Growth and mortality are statistically equal on shell and coal ash based on this and other samples taken during the growing season. By comparison, growth was statistically greater on coal ash in two of three grow out situations in the Delaware Broadkill River experiment. Mortalities and setting densities were statistically indistinguishable on shell and coal ash in the Delaware experiment (Table 2).

Live oyster abundance was quite low on shell and coal-ash substrata retrieved from the Maryland sites. Tissue for metals analysis was harvested and pooled in an attempt to obtain at least 0.1 g final dry weight per sample. Three shell-grown and three coal-ash-grown oysters samples were available from the Little Choptank, and only one shell-grown and one coal-ash-grown sample were available from the Broad Creek site. The paucity of tissue was not apparent until shells and pucks had been returned to the laboratory and individual oysters were actually shucked. Many apparently "live" oysters were filled with sediment; had this been known during the November collection period, more substrata would have been collected.

Tissue availability from Broadkill River, Delaware oysters was conversely, quite good. Both shell-grown and coal-ash-grown oysters were readily abundant; tissue was harvested and pooled as above. A total of eight shell-grown and eight coal-ash-grown samples have been analyzed to date.

Broadkill River oysters

Analyses of the following metals have been completed: zinc, iron, copper, and cadmium. Copper and iron concentrations were higher in Broadkill River shell-grown oysters than in coal-ash-grown oysters (Table 3). Non-parametric statistical analysis (Mann-Whitney U test, = 0.05) found both metals to be present in significantly higher concentrations in shell-grown oysters than in coal-ash-grown oysters. Zinc, on the other hand, was present in significantly higher concentrations in coal-ash-grown oysters than in shell-grown oysters. Cadmium concentrations were not detectable using flame atomic absorption methods and it is possible that the digestion procedure employed allowed for the volatilization of cadmium from the samples; quality control experiments are being conducted to ascertain if volatilization does occur during the digestion procedure.

Little Choptank River and Broad Creek oysters

No statistical tests were employed to analyze these data due to the small number of replicates. The range of concentrations of copper, zinc, and iron in these oysters is presented in Table 3. The values for zinc and iron concentrations seem to be quite different from the concentrations determined for the Broadkill River oysters. These differences are most probably due to a difference in water quality and food supply at the Maryland sites.

Table 1

Chesapeake Bay data sampled on Nov. 16, 1987. Shell Height data are mean ± Standard Deviation in millimeters. * denotes significant difference from shell $p < .05$ (n in parentheses).

	BROAD CREEK	LITTLE CHOPTANK
SHELL	22.22 ± 7.19 (9)	12.03 ± 4.53 (35)
COAL	16.33 ± 7.45 (6)	7.62 ± 3.84 (13)

Chesapeake Bay data sampled on Nov. 16, 1987. Setting Density (#Spat/Cm2), and Percent Mortality (Number of dead oysters/total number set) X 100.

	Setting Density	Percent Mortality
Broad Creek	Shell = .0122	Shell = 59.4
	Coal = .0075	Coal = 77.6
Little Choptank	Shell = .0626	Shell = 64.4
	Coal = .0317	Coal = 84.7

Table 2

Broadkill River, Delaware data sampled on November 18, 1987. Shell Height data are mean ± Standard Deviation in millimeters. * denotes significant difference from shell $p < .001$ (n in parentheses).

	BENTHIC	TANK	HANGING
SHELL	10.20 ±.824 (112)	4.92 ±.173 (284)	17.31 ±.596 (177)
COAL	19.61 ±.659 (175)*	9.84 ±.218 (178)*	14.28 ±.343 (533)1*

Broadkill River, Delaware data sampled on November 18, 1987. Setting Density (#Spat/Cm2), and Percent Mortality (Number of dead oysters/total number set) X 100.

	Setting Density	Percent Mortality
Benthic	Shell = .218	Shell = 51.5
	Coal = .134	Coal = 42.5
Tank	Shell = 1.10	Shell = 28.5
	Coal = .079	Coal = 6.32
Hanging	Shell = .224	Shell = 2.2
	Coal = .235	Coal = 5.7

Table 3. Metal concentrations in tissue of oysters (Crassostrea virginica) grown on stabilized coal-ash and oyster shell substrata. Analyses were performed using atomic absorption spectroscopy, flame mode. Concentrations are ug/g dry weight of tissue; Broad Creek, Maryland values are ranges, while Broadkill River, Delaware are means ± one standard deviation.

ELEMENT	SITE	TISSUE	CONCENTRATION (ug/g)
Zinc	Broadkill R. (n=8/treatment)	shell-grown	$\bar{X}$ = 2437 + 406
		coal-ash-grown	$\bar{X}$ = 2901 + 232
	L. Choptank and Broad Creek (n=4/treatment)	shell-grown	875 to 1103
		coal-ash-grown	0 to 945
Iron	Broadkill R. (n=8/treatment)	shell-grown	$\bar{X}$ = 286 + 48
		coal-ash-grown	$\bar{X}$ = 218 + 16
	L. Choptank and Broad Creek (n=4/treatment)	shell-grown	307 to 505
		coal-ash-grown	304 to 2058
Copper	Broadkill R. (n=8/treatment)	shell-grown	$\bar{X}$ = 116 + 11
		coal-ash-grown	$\bar{X}$ = 96 + 11
	L. Choptank and Broad Creek (n = 4/treatment)	shell-grown	55 to 92
		coal-ash-grown	49 to 118

There are several metals yet to be determined in these samples; the volatile elements (arsenic, selenium, and mercury) will require a separate digestion procedure than the one currently used. Quality control experiments are being conducted to determine the most appropriate method for these elements. Also, several elements are present in such low concentrations that the atomic absorption furnace mode of detection will be required.

DISCUSSION

The setting, growth, and mortality data from the Chesapeake ad Delaware studies indicate that stabilized coal ash is an acceptable alternative substratum for oyster settlement and growth; these results concur with preliminary laboratory oyster setting studies (Price, 1987). Differences do exist; however, between the three sites chosen for this pilot study. The Broadkill River, Lewes, Delaware is, historically, a "good" area for growing oysters; there is a plentiful supply of food for filter feeding bivalves and tidal flushing provides a constant source of necessary nutrients. The suspended sediment in the River has also been implicated in stimulating/enhancing feeding in oysters (Ewart, 1985). The differences in oyster setting, growth, and mortality between the two Chesapeake sites most probably are due to water and food quality at each site. As corroborative evidence, our calculated setting intensity on the shell and coal-ash substrata at Broad Creek was similar to the setting intensity determined for Broad Creek by researchers at the Horn Point Laboratory of the University of Maryland.

Preliminary metals analyses provide evidence that there is a difference in metals concentrations between coal-ash growth and shell-grown oysters. Coal-ash is known to adsorb ions from water and may be depleting the water in the immediate vicinity of a filtering oyster (Andren et al., 1980); given that such a depletion may occur, there is no evidence at present to determine that differences in tissue metals concentrations results in deleterious effects in juvenile oysters.

In conclusion, stabilized coal ash appears to be an acceptable alternative cultch material for the oyster Crassostrea virginica. The second year of this study will provide additional setting, growth, and mortality data, and hopefully, additional oyster tissue for metals accumulation analyses.

LITERATURE CITED

Andren, A., M. Anderson, N. Loux and R. Tabbot. 1980. Element flow in aquatic systems surrounding coal fired power plants. U.S. Document EPA-600/3-80-076, July 1980.

Ewart, J. W. 1985. Effect of suspended silt on growth of Crassostrea virginica and ultrastructural condition of ingested algal food in feces. Master's thesis, University of Delaware, College of Marine Studies, Lewes, Delaware.

Price, K. S. (ed.). 1987. Project ASHREEF. A report on a stabilized coal waste fish reef on Delaware subaqueous lands. Electric Power Partners Program, College of Marine Studies, University of Delaware, Lewes, Delaware.

Sokal, R. R. and F. J. Rohlf. 1981. Biometry, Second Edition. 859 pp. W. H. Freeman and Company, New York, New York.

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Abstract only

The Sedimentary History of Submerged Macrophytes in the Upper Chesapeake Bay

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Seeds preserved in sediments show spatial and temporal distributions of 12 species of submerged macrophytes in 11 Upper Bay embayments and tributaries. Three of the species are known only from the seed record. The time spanned in the sediment cores includes a warm interval from 1000 to 1200 A.D., when temperatures are believed to approximate the mean annual temperature from 1930 to 1960; a cool period estimated to be a degree C lower than the 1930 to 1960 mean annual temperature extending from the 13th century into the 18th century; and European occupation, beginning in the mid-17th century and accompanied by extensive land clearance. For at least 1000 years, the geographic gradients of species composition and diversity have remained constant. Seeds of Vallisneria americana are most common where salinities are ≤1‰, and seeds of Zannichellia palustris where salinities are >3‰. Species diversity decreases latitudinally as sedge and cordgrass marshes become more extensive, with no macrophytes present in those tributaries surrounded by Cyperaceae and Spartina alterniflora. Similarly, seeds are not found in the Pocomoke River which is also surrounded by Taxodium distichum swamps. In one area representing the longest record, species diversity increased from a community consisting only of Zannichellia palustris when water levels were low to a community of 7 species, as water depth increased with rising sea level. Significant increases and decreases in populations of individual species occurred within the broad geographic gradients, with the majority of increases occurring during an interval immediately after European settlement and the most significant decline within the last two decades. The post-European increase is interpreted as a response of the plants to increased nutrient input with deforestation and agriculture. However, as turbidity increased with erosion and sedimentation, increases in populations were erased and sporadic declines culminated in the most recent near extinction of all species in the majority of upper Bay tributaries.

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Alternative Sampling Strategies for a Survey of Submerged Aquatic Vegetation in the Chesapeake Bay

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ABSTRACT

Complete aerial censuses of Submerged Aquatic Vegetation (SAV) in the Chesapeake Bay including digitizing and mapping SAV beds have been conducted in 1978, 1984, 1985, 1986, and 1987. This approach provides the best information, but it is expensive. Digitizing and mapping a fraction of the Bay each year provides a less expensive alternative for monitoring SAV. Annual estimates of the total area of SAV in the Bay would be available from sampling for evaluating Bay recovery. For example, a 20% sample of the Bay could estimate the total SAV area for the whole Bay within $\pm$ 15% and a 50% sample within $\pm$ 2%. The SAV beds in the entire Bay could be periodically digitized and mapped for reviewing wetland permit applications and for following changes on specific areas.

BACKGROUND

SAV is important to Chesapeake Bay because it provides important habitat for many species, enhances water clarity, binds sediments, and enhances nutrient cycling (Orth, *et al.* 1986?). Complete aerial censuses in 1978, 1984, 1985, 1986, and 1987 have provided detailed information but annual censuses are expensive (about $160K per year) and quality control of the large amount of data is difficult. The total area of SAV beds and the area of four SAV density classes are compiled for the Bay, for 3 zones (upper, middle, and lower bay), for 21 sections (e.g. watersheds) and for 164 U.S. Geological Survey quadrangles (135 with SAV for at least one year). Complete SAV maps are also used to evaluate wetland permit applications and to follow SAV changes for specific areas. Species composition is also estimated.

This paper will develop alternative sampling strategies for estimating total SAV area. Total SAV area is the most important value and is widely believed to be an excellent indicator of

water quality. I will discuss alternative survey designs, the definitions of sampling units and the advantages of stratification, and show the relevance of each to an effective sampling strategy. Confidence interval widths are projected for several sample sizes and alternative stratifications. Statistical methods for designing and analyzing sample surveys of SAV are presented and illustrated with a simple example.

ALTERNATIVE SURVEY DESIGNS

Sampling a portion of the Bay at a fraction of the cost of a complete mapping will allow annual estimates of the total area of SAV and of changes in total area over time. Each year digitizing, mapping, and reporting costs about $100K. This cost could be proportionally reduced by sampling. Because aerial photographs are relatively inexpensive ($60K) compared to the cost of digitizing and mapping SAV beds, the entire Bay could be photographed each year. Then non-sampled areas in any given year could be digitized and mapped if the need arose in the future. I will refer to a sample of part of the Bay as a survey and refer to a complete coverage as a census. Three alternative sampling designs will be considered:

1. Complete annual censuses.
2. Annual sample surveys with rotating sampling units. Quarter quadrangles (for example) are randomly divided into say five groups. One group is digitized and mapped each year on a rotating basis so that the entire Bay will have been mapped at the end of five years. A two year, three year or other rotation could also be used.
3. Annual sample surveys with fixed sampling units. A random sample of quarter quadrangles (for example) is selected and the same quadrangles are digitized and mapped each year.

Design 1 provides the best information but is the most expensive. Current SAV maps are always available to evaluate wetland permit application and to follow small scale SAV changes. The large amount of data makes quality control difficult. Designs 2 and 3 also provide annual estimates of the total area of SAV for evaluating Bay recovery, but they are less precise than the estimates from design 1. Designs 2 and 3 are equivalent for estimating the annual total SAV area. They differ in the estimation of changes in the total SAV area between two years. Design 2 is analogous to a unpaired t-test while design 3 is analogous to a paired t-test. Design 3 is more precise than design 2 for estimating changes in total SAV area between years. Design 2 has the advantage of providing a composite map for evaluating wetland permit applications although different parts of the map would be digitized in different years. Design 3 requires a periodic complete survey to provide maps. Estimated changes would still be based on the fixed sampling units.

A ratio estimate should probably be used to analyze a SAV sample survey (designs 2 and 3) because it exploits the correlation between the current sample survey and a previous complete census to increase the precision of the estimates. The correlations among the annual SAV areas for quadrangles ranged from 0.98 to 0.74, based on the 1978, 1984, 1985, and 1986 complete SAV area censuses.

SAMPLING UNITS AND STRATIFICATION

Sample surveys require a careful definition of sampling units and strata. Sampling units are the plots on which the SAV areas are measured. Sampling units must not overlap and must completely cover all areas of the Bay where SAV may occur. Sampling unit boundaries must be unambiguous, identifiable, and unchanging from year to year. I have used quadrangles as the sampling unit for this paper because data from previous surveys were not easily available for smaller units.

Estimates can be reasonably made for the three zones using quadrangle sampling units but the large size of quadrangles makes it impractical to estimate SAV areas for the 21 sections of the

Bay. A minimum of two sampling units are needed in each of the 21 sections to estimate the variance. This requires a minimum of 42 quadrangles (31% of the Bay) to obtain section estimates. These estimates would not have the advantage of stratification on the SAV density which would increase the precision of the estimates. Two SAV density strata would require 82 quadrangles (4 per section for 62% of the Bay). Quarter or ninth quadrangles may be better sized sampling units. Even if section estimates are not required, quarter or ninth quadrangles would provide more flexibility for stratification. Other sampling units are possible.

Stratification is the subdivision of the Bay into homogenous regions that are called strata. It can dramatically increase the precision of the estimates with the same sample size or decrease the cost of a survey by reducing the sample size. The precision of the SAV estimates for the Bay or its subdivisions is estimated by comparing the SAV area of the sampling units in the same stratum. Because differences among strata do not contribute to the sampling error, grouping sampling units into homogenous strata can greatly increase the precision and decrease the variance of the estimates. For example, the variance of 1, 2, 3, 11, 12, and 13 is 31 with 5 degrees of freedom. Dividing the numbers into two strata, [1, 2, 3] and [11, 12, 13], reduces the variance to 2 with the loss of 1 degree of freedom. The stratification could be based on the area of SAV in each sampling unit as measured during a recent census. If accurate measurements are not available for the sampling units, they can be assigned to strata on the basis of a visual estimate without biasing the estimates. This stratification reflects the current distribution of SAV. Alternative stratifications based on area of water with the appropriate depth or other criterion may better reflect the historic distribution of SAV and may better reflect future SAV distribution.

The strata do not have to be equally sized or have the same number of samples. An optimal allocation of samples to strata increases the precision of the whole Bay estimates. The number of samples in a stratum should be proportional to the size of the stratum, proportional to the standard deviation in the stratum, and inversely proportional to the square root of the cost to measure the stratum. At least two sampling units must be selected from each stratum in order to estimate the variance, but the sampling rate in unimportant strata may be decreased by increasing the size of those strata. Important areas can be designated as separate strata and completely sampled each year.

SURVEY ESTIMATES

Statistical methods for designing and analyzing sample surveys of SAV are presented in the "survey estimates," "sample size," and "example" sections. These sections are intended as a tutorial for those who may have to implement a sample survey and may be skipped by those who wish an overview.

With a complete census (design 1) the total SAV area is directly measured. There is no sampling error and consequently the variance and confidence interval widths are zero. The following discussion relates to sample surveys (designs 2 and 3). The total SAV area for the Bay or one of its subdivisions is the sum of the SAV area in each stratum (combination of zone and SAV density strata)

$$A = \sum A_k = \sum N_k \bar{a}_k \qquad (1)$$

where A is the mean per unit estimate of the total SAV area for the Bay, A_k is the SAV area in stratum k, N_k is the number of sampling units in stratum k (including those that were not measured), and $\bar{a}_k = \sum_{i=1}^{n_k} a_{ki}/n_k$ is the mean SAV area for the n_k measured sampling units in stratum k. With design 2 (rotating sampling units), separate estimates are made each year using (1) and compared independently by observing if their confidence intervals overlap. With design 3 (fixed sampling units) annual estimates of the total SAV area are also made

independently using (1), but tests for changes in the total SAV area between two years uses the differences in the SAV areas between the two years similar to a paired t-test. In this situation, $\bar{a}_k$ in (1) is defined to be the mean difference in SAV area instead of the mean SAV area.

The variance of the total SAV estimate is the sum of the variances for each stratum

$$v(A) = \sum v(A_k) = \sum N_k^2 \left(1-\frac{n_k}{N_k}\right) \frac{s_k^2}{n_k}. \quad (2)$$

The estimate of the stratum variance for a total is N_k^2 times the stratum variance of a mean $\frac{s_k^2}{n_k}$ where $s_k^2 = \frac{1}{(n_k-1)} \sum_{i=1}^{n_k} \left(a_{ki}-\bar{a}_k\right)^2$. Use the σ_{n-1} definition on a calculator to estimate the sample standard deviation s_k. The estimate also includes the finite population correction $\left(1-\frac{n_k}{N_k}\right)$. A 95% confidence interval for A is $A \pm t_{0.05,f} \sqrt{v(A)}$, where $t_{0.05,f}$ is the value from a t-table for 5% significance level with f degrees of freedom,

$$f = \left[\sum v(A_k)\right]^2 / \sum \frac{v^2(A_k)}{n_{k-1}}. \quad (3)$$

If f is not an integer, use the next lowest integer.

A ratio estimate exploits the correlation between the current sample survey and a previous complete census to increase the precision of the estimates. The total SAV area from an earlier complete census A' is multiplied by the average change per sampling unit. That average change is estimated by the ratio R of the mean SAV areas from the current sample survey $\bar{a}_k$ to the mean areas on the same sampling units for the earlier complete census $\bar{a}'_k$.

$$A^* = A' R = A' \frac{\sum N_k \bar{a}_k}{\sum N_k \bar{a}'_k} \quad (4)$$

The variance of the ratio estimate is

$$v(A^*) = \sum \left[\frac{N_k^2 \left(1-\frac{n_k}{N_k}\right)}{n_k}\right] \left(s_k^2 + R^2 s_k'^2 - 2 R \rho_k s_k s'_k\right). \quad (5)$$

where s'_k is the standard deviation of SAV areas for the earlier complete census using the same sampling units as the current sample survey, and ρ_k is the correlation between the SAV area on the current survey and on the earlier complete survey. More information on survey estimates may be found in Steel and Torrie (1980), Snedecor and Cochran (1980), and Cochran (1977).

SAMPLE SIZE

I have projected the precision of estimates for the total area of SAV in the Bay and in each zone for several sample sizes and stratifications using data from the 1978, 1984, 1985, and 1986 censuses provided by the Annapolis Field Office, U.S. Fish and Wildlife Service. I have used the mean per unit estimate instead of the ratio estimate for planning because the correlations between an earlier complete survey and the current sample survey may decrease over time, decreasing the precision of the ratio estimate. Four SAV density strata were arbitrarily defined for the sample size projections presented in this paper using the mean SAV area in the quadrangles for the four censuses (density 1: 0-9 ha/quad, 2: 10-99, 3: 100-499 and 4: 500 and more). Density strata 3 and 4 were combined in zone 2 because density stratum 4 had only a single quadrangle. Quadrangles for which SAV areas were not available for any of the four surveys are excluded.

The number of sample quadrangles in each zone and density stratum was determined using an optimal allocation which minimizes the variance of the estimate given the total sample size n:

$$n_k = n \frac{N_k s_k}{\sum N_k s_k}. \quad (6)$$

I have treated all quadrangles as if the cost to measure the area of SAV were the same. If estimates of the cost to measure the SAV in a quadrangle are available (perhaps proportional to the area of SAV) an alternate allocation formula is available. Say the total cost is

$$C = c^* + \sum c_k n_k \quad (7)$$

where c^*= fixed cost, and c_k= cost of measuring the SAV area in a quadrangle in stratum k. Then

$$n_k = (C\text{-}c^*) \frac{N_k S_k / \sqrt{c_k}}{\sum N_k S_k \sqrt{c_k}}. \quad (8)$$

For planning estimates of SAV area, I estimated $v(\bar{a}_k)$ separately for each survey year and used the mean of the four estimates. The precision of estimates of the annual total SAV area will be the same for designs 2 and 3. The variance for changes between two annual total SAV estimates with design 2 (rotating sampling units) is twice the variance of an annual estimate. For changes with design 3 (fixed sampling units), I formed the differences between successive years (1978-1984, 1984-1985, and 1985-1986), estimated $v(\bar{a}_k)$ separately for each difference, and used the mean of the three estimates. Confidence interval widths are expressed as the percentage of the estimate that must be added to and subtracted from the estimate

$\frac{t_f \sqrt{v(A)}}{A}$ 100 where t_f is the tabular t value for f degrees of freedom (f = number of quadrangles minus the number of strata).

EXAMPLE

To illustrate the use of these equations, consider the following example with 5 quadrangles in the first stratum and 6 in the second. The SAV area from one year is given in the column headed "Complete Census." The SAV area for a sample of the quadrangles for the next year is listed in the column headed "Sample Survey."

First Stratum			Second Stratum		
No. of Quad.	Complete Census	Sample Survey	No. of Quad.	Complete Census	Sample Survey
10	11.60	9.72	37	223.91	
15	10.10	7.70	33	97.90	36.57
18	0.00		36	346.69	164.37
16	12.89		14	132.99	
27	0.00	0.52	9	439.96	369.54
	34.59		26	586.96	295.21
				1828.41	
Mean*	7.23	5.98		367.88	216.42
Variance*	40	23		42177	21570
Correlation*	0.9958			0.8615	
Variance	41			35701	

* using only those sampling units that were measured in the sample survey.

Estimates for the sample survey are:

$A_1 = 5 * 5.98 = 29.90$ $A_2 = 6 * 216.42 = 1298.52$ $A = 29.90 + 1298.52 = 1328.42$

$v(A_1) = 5^2 \left(1-\frac{3}{5}\right) \frac{23}{3} = 77$ $v(A_2) = 6^2 \left(1-\frac{4}{6}\right) \frac{21570}{4} = 64710$ $v(A) = 77 + 64710 = 64787$

$f = \left(77 + 64710\right)^2 / \left[\frac{77^2}{2} + \frac{64710^2}{3}\right] = 3$ degrees of freedom.

95% confidence interval for A is $1328 \pm 3.182 \sqrt{64787} = (518 - 2138)$
The ratio estimate, using the total SAV area from the complete survey $34.59 + 1828.41 = 1863$, is

$$A^* = 1863 \frac{5*5.98 + 6*216.42}{5*7.23 + 6*367.88} = 1863 * 0.5921 = 1103$$

$$v(A^*) = \left[\frac{5^2\left(1-\frac{3}{5}\right)}{3}\right]\left(23 + 0.5921^2*40 - 2*0.5921*0.9958\sqrt{23}\sqrt{40}\right) +$$

$$\left[\frac{6^2\left(1-\frac{4}{6}\right)}{4}\right]\left(21570 + 0.5921^2*42177 - 2*0.5921*0.8615\sqrt{21570}\sqrt{42177}\right) = 16760$$

95% confidence interval for A^* is $1103 \pm 3.182 \sqrt{16760} = (691 - 1515)$ with 3 degrees of freedom. In this example, the ratio estimate A^* is much more precise than the mean per unit estimate A because of the large correlations between the surveys.

With equal costs, the optimal sample size for the first stratum using a total sample of 7 quadrangles is $n_1 = 7 \frac{5\sqrt{41}}{5\sqrt{41} + 6\sqrt{35701}} = 0.03$. There must be at least 2 samples in each stratum to estimate a variance, so take $n_1=2$ and $n_2 = 7 - 2 = 5$.

RESULTS AND DISCUSSION

Design 1, a complete census, provides the best estimates but is the most expensive. Because the whole bay is digitized and mapped, there is no sampling error. Consequently, confidence intervals have zero width. The annual total SAV area estimates have the same precision with design 2 (sample survey with rotating sampling units) and design 3 (sample survey with fixed sampling units). Designs 2 and 3 differ in the precision of the estimated change between two annual estimates (Table 1).

An 18% sample of the quadrangles will provide estimates of the annual total SAV area with a 95% confidence interval width of $\pm$ 27.2% of the estimate with designs 2 and 3 (Table 1). Not using SAV density strata greatly increases the width to $\pm$ 66.0% with a 19% sample. The width is decreased to $\pm$ 15.1% with a 19% sample if zone estimates are not required. Confidence intervals are quite wide for zone estimates ($\pm$ 46.4%, $\pm$ 132%, and $\pm$ 40.4% for an 18% sample with four SAV density strata).

A 48% sample of the quadrangles will provide estimates of the annual total SAV area with a 95% confidence interval width of $\pm$ 2.0% of the estimate with designs 2 and 3 (Table 1). Using two instead of four SAV density strata increases the width to $\pm$ 4.6% with a 48% sample. Not using SAV density strata greatly increases the width to $\pm$ 29.8% with a 49% sample. There is little change in the width without zone estimates ($\pm$ 2.2% with 49% sample) because there are adequate samples in each zone. Confidence intervals are wider for zone estimates ($\pm$ 5.8%, $\pm$ 11.1%, and $\pm$ 1.4% for an 48% sample with four SAV density strata).

An 18% sample of the quadrangles will estimate changes between two annual total SAV areas with a confidence interval width of $\pm$ 38.5% of the annual estimate with design 2 (rotating sampling units) and $\pm$ 19.2% with design 3 (fixed sampling units). Design 3 is more precise because changes on the same sampling units are followed over time, but a periodic complete census is required to digitize and map all parts of the Bay. Not using SAV density strata increases the widths to $\pm$ 93.3% and $\pm$ 35.8% (19% sample) with designs 2 and 3 respectively. The widths are decreased to $\pm$ 21.4% and $\pm$ 16.0% (19% sample) if zone estimates are not required.

Table 1. Optimal allocation of sample quadrangles to upper (1), middle (2), and lower (3) zones of the Bay and four SAV density strata. Projected 95% confidence interval widths are expressed as the percentage of the estimate that should be added to and subtracted from the estimate. Confidence intervals widths are given for estimates of annual total SAV area (same for designs 2 and 3) and for the change in SAV area between two successive annual total SAV areas with designs 2 (rotating sampling units) and 3 (fixed sampling units).

	Den.	SAV	Var.	SAV		Number of Sample Quadrangles							
Zone	str.	mean	mean	total	Quads	16%	18%	24%	32%	40%	48%	59%	69%
1	1	5	50	96	19	2	2	2	2	2	2	5	9
1	2	35	2319	277	8	2	2	2	2	4	7	8	8
1	3	216	30014	1298	6	2	2	3	5	6	6	6	6
1	4	1108	672258	2216	2	2	2	2	2	2	2	2	2
2	1	3	26	111	35	2	2	2	2	2	3	7	13
2	2	29	782	261	9	2	2	2	2	2	5	9	9
2	3	292	54915	2041	7	2	2	4	7	7	7	7	7
3	1	3	29	52	16	2	2	2	2	2	2	3	6
3	2	41	1556	405	10	2	2	2	2	4	8	10	10
3	3	248	16345	3716	15	2	2	5	10	15	15	15	15
3	4	940	236342	7518	8	2	4	8	8	8	8	8	8
Bay total quadrangles					135	22	24	33	44	54	65	80	93
95% conf. int. ± % (total)						37.2	27.2	13.3	6.5	3.4	2.0	1.0	0.6
95% conf. int. ± % (change 2)						56.2	38.5	18.8	9.2	4.8	2.8	1.4	0.8
95% conf. int. ± % (change 3)						21.3	19.2	13.1	6.7	3.8	2.6	1.4	0.8
Zone 1 total quadrangles					35	8	8	8	11	14	17	21	25
95% conf. int. ± % (total)						46.4	46.4	46.4	19.2	9.4	5.8	2.7	1.7
95% conf. int. ± % (change 2)						65.6	65.6	65.6	27.2	13.3	8.2	3.8	2.4
95% conf. int. ± % (change 3)						62.2	62.2	62.2	23.2	10.5	6.9	3.6	2.3
Zone 2 total quadrangles					51	6	6	8	11	11	15	23	29
95% conf. int. ± % (total)						132	132	60.9	18.9	18.9	11.1	5.2	3.3
95% conf. int. ± % (change 2)						187	187	86.1	26.7	26.7	15.7	7.4	4.7
95% conf. int. ± % (change 3)						89	89	44.5	21.9	21.9	13.9	7.3	4.7
Zone 3 total quadrangles					49	8	10	17	22	29	33	36	39
95% conf. int. ± % (total)						66.3	40.4	13.8	7.8	2.9	1.4	0.8	0.4
95% conf. int. ± % (change 2)						93.8	57.1	19.5	11.0	4.1	2.0	1.1	0.6
95% conf. int. ± % (change 3)						32.5	24.0	11.2	6.9	3.1	1.8	1.2	0.8

Table 1. Continued.

Combined SAV density strata.

Zone	Den. str.	SAV mean	var. mean	SAV total	Quads	Number of Sample Quadrangles 9%	13%	21%	30%	39%	48%	59%	70%	79%
1	12	15	1035	411	27	2	2	2	2	5	9	14	19	23
1	34	439	291304	3514	8	2	3	6	8	8	8	8	8	8
2	12	10	365	452	44	2	2	2	2	5	9	14	18	22
2	34	292	54915	2041	7	2	2	2	3	7	7	7	7	7
3	12	22	1117	561	26	2	2	2	2	5	9	14	19	23
3	34	488	209085	11234	23	2	7	14	23	23	23	23	23	23
Totals					135	12	18	28	40	53	65	80	94	106
95% conf. int. ± % (total)						103.5	49.2	27.6	13.9	6.7	4.6	3.2	2.4	1.8
95% conf. int. ± % (change 2)						146.4	69.6	39.0	19.7	9.5	6.5	4.5	3.4	2.5
95% conf. int. ± % (change 3)						42.3	24.6	16.2	12.3	6.5	4.4	3.0	2.2	1.6

Without SAV density strata.

Zone	Den. str.	SAV mean	Var. mean	SAV total	Quads	Number of Sample Quadrangles 9%	19%	29%	39%	49%	59%	70%	79%
1		119	98059	4167	35	3	7	11	15	19	23	27	35
2		58	20732	2981	51	2	5	7	10	12	15	18	23
3		278	168239	13642	49	7	14	21	28	35	42	49	49
Totals					135	12	26	39	53	66	80	94	107
95% conf. int. ± % (total)						115.1	66.0	49.2	37.8	29.8	23.0	16.6	11.0
95% conf. int. ± % (change 2)						162.8	93.3	69.6	53.5	42.1	32.5	23.5	15.6
95% conf. int. ± % (change 3)						63.6	35.8	27.8	21.8	18.2	15.2	12.4	9.2

Without zones.

Zone	Den. str.	SAV mean	Var. mean	SAV total	Quads	Number of Sample Quadrangles 11%	19%	30%	39%	49%	60%	70%	79%
	1	4	40	278	70	2	2	2	4	8	16	29	42
	2	35	1577	952	27	2	2	5	11	20	27	27	27
	3	250	29156	7013	28	5	12	23	28	28	28	28	28
	4	973	280806	9734	10	6	10	10	10	10	10	10	10
Totals				17978	135	15	26	40	53	66	81	94	107
95% conf. int. ± % (total)						30.6	15.1	7.8	3.6	2.2	1.0	0.6	0.4
95% conf. int. ± % (change 2)						43.3	21.4	11.0	5.1	3.1	1.4	0.8	0.6
95% conf. int. ± % (change 3)						28.2	16.0	8.6	4.2	2.6	1.4	1.0	0.6

A 48% sample of the quadrangles will estimate changes between two annual total SAV areas with a confidence interval width of ± 2.8% of the annual estimate with design 2 (rotating sampling units) and ± 2.6% with design 3 (fixed sampling units). Using two instead of four SAV density strata increases the widths to ± 6.5% and ± 4.4% for designs 2 and 3 respectively (48% sample). Not using SAV density strata increases the widths to ± 42.1% and ± 18.2% (49% sample). The widths are about the same (± 3.1% and ± 2.6% with a 19% sample) if zone estimates are not required.

CONCLUSIONS

* Sampling can substantially reduce the cost of SAV surveys for estimating annual total SAV areas and changes in total SAV area between years.
* Quadrangles are too large as sampling units to estimate SAV areas for sections of the Bay and reduce the precision when zone estimates are required. Quarter or ninth quadrangles appear to be better sized sampling units because they allow greater opportunity for stratification and larger sample sizes.
* Precision is greatly increased by stratification on SAV density.
* Designs 2 and 3 provide equally precise estimates of the annual total SAV area.
* Design 3 (fixed sampling units) provides more precise estimates of the change in total SAV area between years than design 2 (rotating sampling units) but requires a periodic complete census to provide maps for reviewing wetland permit applications and for following changes on specific areas. Design 2 does not require a periodic complete census, but the composite map contains parts mapped in different years.
* A ratio estimate should probably be used to estimate total SAV area and changes in total SAV area.

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REFERENCES

Cochran, W.G. 1977. *Sampling techniques.* Wiley, New York, 428pp.

Orth, R., J. Simons, J. Capelli, V. Carter, L. Hindman, S. Hodges, K. Moore, and N. Rybicki. 1986? *Distribution of submerged aquatic vegetation in the Chesapeake Bay and tributaries - 1985*, EPA. Final Report. 296 pp.

Snedecor, G.W. and W.G. Cochran. 1980. *Statistical methods.* Iowa State Univ. Press, Ames, 507 pp.

Steel, R.G.D. and J.H. Torrie. 1980. *Principles and procedures of statistics.* McGraw-Hill, New York, 633 pp.

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Abstract only

The Status of Yellow Perch Populations in Five Chesapeake Bay Tributary Streams

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Yellow perch commercial fishery landings have declined significantly in many Chesapeake Bay tributaries since the early 1970's. The proposed partial closings of the yellow perch commercial and recreational fishery by the Maryland Department of Natural Resources reflect the concern by State officials that yellow perch populations may be declining in many areas of the Chesapeake Bay. Although commercial statistics and anectodal information from recreational fishermen suggest declining yellow perch populations, a more accurate assessment of yellow perch abundance is needed to determine the status of this important anadromous fish species in Chesapeake Bay waters.

As part of a study to examine effects of the addition of calcium carbonate on freshwater tributary stream chemistry and biota, yellow perch spawning runs, ichthyoplankton abundances, and juvenile abundances were examined in five Chesapeake Bay tributaries during 1986 and 1987. Results indicate that the abundances of spawning adults in four historically important spawning streams near Annapolis are very low, with fewer than 50 spawning individuals observed in Bacon Ridge Branch, North River, and Magothy Run, and fewer than 200 spawning adults in Severn Run. The majority of the spawning adults observed in these streams are large, relative old (6-8 year) individuals nearing the end of their lifespan. In contrast, results from Mattawoman Creek (a tributary to the Potomac River) show that the yellow perch spawning run is relatively strong, with more than 10,000 individuals collected during the 1987 spring run. The majority of spawning yellow perch in Mattawoman Creek were estimated to be 2 to 5 years old, with few individuals older than 6 years of age collected.

As expected from the results of the spawning survey, ichthyoplankton abundances in Bacon Ridge Branch and North River were lower than in Mattawoman Creek. Yellow perch larvae were present in ichthyoplankton collections for these three streams throughout most of the 1987 study period (March through June). The occurence of yolk-sac larvae long after the observed spawning period in Bacon Ridge Branch and North River suggest the possibility of resident yellow perch populations in these streams.

As with the spawning survey, greater numbers of juvenile yellow perch were observed in the Mattawoman Creek estuary collections (126 individuals) than in the other estuarine seine collections. No yellow perch were collected in the 1986 or 1987 South River, the 1986 Magothy River, or the 1986 Severn River seine collections, and only 5 individuals were observed during the 1987 Severn River juvenile survey. Results of the 1986-1987 seining studies are compared with results of seining surveys conducted at the same sampling locations in the mid-1970's.

These results suggest much lower current yellow perch populations in the Annapolis region study streams (Bacon Ridge Branch, North River, Magothy Run, and Severn Run) than had been estimated in the 1960's and 1970's. In addition, the current status of yellow perch populations may be different in different streams or regions of the Chesapeake Bay.

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Abstract only

Bay Anchovy Ecology in Mid-Chesapeake Bay

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Bay anchovy Anchoa mitchilli is the most abundant and ubiquitous fish in the Chesapeake Bay. It is believed to be a major link in pelagic food chains through its role in converting planktonic biomass into available forage for piscivorous fishes. Its population ecology, including abundance, distribution, age, growth, reproduction, and trophic significance, is being investigated in mid-Chesapeake Bay. A trawl sampling program was conducted on a transect from inside the Patuxent River mouth to four kilometers offshore, from March 1986 to November 1987. Mean among month Catch per Unit Effort (CPUE) (numbers per 10 minutes of trawling) varied ten-fold in each year. CPUE was highest in September when catches were dominated by 0+ recruits. July to November CPUE was more than 5 times higher in 1986 than in 1987. Ten-fold annual variability in abundances have been observed in the past 30 years, based on Maryland Department of Natural Resources index surveys in Chesapeake tributaries. Annuli in otoliths indicate that there are four age groups. Maximum age is 3+ when anchovies reach 85 mm fork-length and 5 g wet-weight. Most growth-in-length is completed by age 1+, but large weight increases occur in older fish. Size-at-age is variable and in part attributable to a protracted spawning season and prolonged recruitment period.. Based on a gonosomatic index, the reproductive season extends from May through August. Apparently all mature females spawn each night between 9 p.m. and 1 a.m. Batch fecundities range from 514 to 2,026 eggs and are directly related to female weight. Spawned eggs were more abundant in 1987 on a transect off the Choptank River than off the Rappahannock River, but larval anchovies were more common on the Rappahannock transect, indicating a differential mortality between the two areas. Eggs and larvae were relatively common below the pycnocline on the Rappahannock transect but rare on the Choptank transect, suggesting that low oxygen limits the depth distribution of anchovy eggs and larvae. Future studies will include estimates of mortality rates of early life stages and examination of trophic relationships, energetics, and oxygen tolerances of all life stages to understand better the role of bay anchovy in pelagic, estuarine communities.

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Abstract only

Hatchability and Viability of Striped Bass Eggs: Effects of Tributary Source and Female Size

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Variability in hatching success, survival of larvae to three days posthatch (i.e. viability), and the quality of eggs and larvae were examined in relation to spawning tributary in which females were captured, and size of spawning female. Fertilized eggs from 26 females collected during 1986 in the Nanticoke River, Patuxent River, and C&D Canal were tested. Most variables that were examined differed significantly among female spawners but were not related to spawning tributaries from which females originated. Mean hatchability of 30-36h post-fertilization eggs was 21% lower for spawns of ≤ 10 lb females than for spawns of > 10 lb females. Larval survival to 3 days posthatch was essentially independent of the probability of hatching. Egg dry weights, egg yolk and oil globule volumes, and weights per egg of protein and lipid all varied by 1.5- to 3-fold among spawns from individual females. The smallest eggs and least amounts of constituents were from ≤ 10 lb females. Eggs and 5-day posthatch larvae from ≤ 10 lb females weighed, on average, only 68% and 67% as much, respectively, as those from larger females. Results of the analysis imply that young females from the most recently-matured year-classes (i.e. 1981-82) did not produce eggs of high quality in 1986, compared to those produced by older, larger females. The ultimate probability of survival to recruitment of small larvae that hatched from eggs of small females cannot be determined from our analysis, but size seemingly would increase vulnerability to both starvation and predation-related mortality. It is probable, although not certain, that 1981-82 year-class females will produce larger eggs of higher quality when they grow to larger size. Implications for striped bass management and hatchery production may be important. Aside from low fecundity and egg yield, small females (i.e. < 10 lb) from Maryland tributaries produce relatively poor quality eggs. Our data quantify the relationships, document variability in egg constituents, and allow estimation of hatchability and survival rates, as well as egg characteristics that will be useful in future hatchery applications. Because hatching varies in relation to adult female size and age, results also can aid development of age or size-specific egg production models for Chesapeake Bay striped bass.

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Current Research to Improve the Understanding of Habitat Use, Distribution, and Population Status of Canvasbacks on the Chesapeake Bay

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Abstract: The U.S. Fish and Wildlife Service is currently conducting a 5-year research project to study canvasbacks using the Chesapeake Bay during migration and winter. The objectives of the study are to estimate daily survival rate, identify causes of mortality, evaluate habitat use, determine distribution and abundance of key aquatic foods, and estimate energy and nutritional value of natural foods. Radio-telemetry is being used to evaluate survival, movement patterns, and habitat use of juvenile female canvasbacks wintering on the Chesapeake Bay. Aquatic food availability and distribution will be determined using currently available data bases and field sampling of nutritional aspects of the most important aquatic foods will be evaluated during feeding and reproduction of captive canvasbacks at Patuxent Wildlife Research Center.

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Abstract only

Patterns of Post Larval Availability and Settlement in the Blue Crab: Effects of Time, Space, and Habitat

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One focus of a long term program to elucidate blue crab (Callinectes sapidus) recruitment dynamics in Chesapeake Bay is the resolution of post larval availability and its relationship to settlement patterns in time and space. Postlarvae and early juveniles were collected on three dates from three habitats (plankton, artificial settlement substrates and submersed vegetated bottom) within each of two sites at two geographic locations in the York River. Relationships of postlarval and early juvenile abundance varied among time, location and habitat. These complex patterns suggest the importance of fine-scale processes in settlement dynamics.

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Abstract only

Regulatory Mechanisms of Postlarval Blue Crab Recruitment: Settlement, Metamorphosis, and Developmental State

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As part of a long-term program on blue crab, Callinectes sapidus, recruitment dynamics in Chesapeake Bay, we quantified the developmental state (molt stage: proximity to metamorphosis) of recruiting postlarvae in relation to time, habitat and geographic location. Developmental state was assessed by molt staging the uropods of freshly-collected postlarvae, and verifying the staging procedure with laboratory cultures. Geographic locations included two sites each at the mouth of the York River and 10 km upriver; habitats included plankton, artificial settlement substrate, and submersed vegetated bottom. Temporal sampling comprised daily and monthly variation. Various main and interaction effects of time, habitat and geographic location were significant, and yielded a complex model of the interrelationships between postlarval developmental state, the scale and nature of spatio-temporal factors, and regulatory mechanisms of recruitment in the blue crab.

Understanding the Estuary: Advances in Chesapeake Bay Research. Proceedings of a Conference. 29-31 March 1988. Baltimore, Maryland. Chesapeake Research Consortium Publication 129. CBP/TRS 24/88.

Abstract only

Variation in Postlarval Blue Crab Settlement on Artificial Substrates in the York River, Virginia

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Annual and temporal variation in blue crab settlement on artificial substrates was examined during the period of primary postlarval ingress into the York River (Aug. - Dec.) over a three year period (1985-1987). Daily records indicated 1-3 day settlement events correlated with lunar phase; settlement was strongly associated with full moon and less so with new moon. Significant differences existed in the magnitude of settlement between years. Juvenile populations of blue crabs which settled or recruited into a seagrass bed approximately 12 km nearer the mouth of the York River exhibited similar annual variations in abundance. Settlement dynamics must be examined on various scales ranging from days to years in order to understand the regulation and patterns of recruitment in the blue crab.

CONCURRENT SESSIONS AND POSTER SESSION:

NUTRIENTS

Chairs:

George Simmons
Virginia Polytechnic Institute and
State University

Carl Cerco
U.S. Army Corps of Engineers

Understanding the Estuary: Advances in Chesapeake Bay Research. Proceedings of a Conference. 29-31 March 1988. Baltimore, Maryland. Chesapeake Research Consortium Publication 129. CBP/TRS 24/88.

Radionuclide Concentrations in Susquehanna River and Chesapeake Bay Sediments -- Implications for Transport and Distribution of Particle-Reactive Pollutants

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The Peach Bottom Atomic Power Station (PBAPS) has contributed measurable quantities of radioactivity to the lower Susquehanna River and Chesapeake Bay. Since 1980, we have monitored, in spring and fall, concentrations of plant-related radionuclides in sediments. Mass balance estimates derived from grab samples indicate that less than 20% of particle-reactive radiozinc, radiocobalt, and radiocesium is present in surface sediments (<10 cm) of the Conowingo Reservoir, an impoundment of the lower Susquehanna. The remaining release inventory is assumed to be transported to the Chesapeake Bay. Significant seasonal variations in radionuclide trapping efficiency by the reservoir are not apparent. Core samples confirm that some, but not all of this surface sediment radionuclide inventory, remains within the reservoir -- trapped in discrete locations by subsequent sediment accumulation. The detection of PBAPS-related radionuclides in sediments of the upper Chesapeake Bay (above Baltimore, MD) confirms transport of these radionuclides from the Susquehanna. Radionuclide concentrations in sediments were generally undetectable south of the Sassafras River. This is attributed to dispersion of radionuclide-labeled particles, dilution by unlabeled particles from other upper-bay tributaries, and possibly, desorption of particle-bound radioactivity.

INTRODUCTION

The Susquehanna River is the principal source of fresh water and fluvial sediments in the Upper Chesapeake Bay (Schubel 1972). Because many pollutants, including trace metals, radionuclides and trace organics

have a high affinity for association with particles suspended in the water column, information regarding the transport and environmental distribution of particulates is important in determining the fate of Susquehanna-derived pollutants in the Upper Chesapeake Bay.

Since 1980, the Power Plant Research Program has conducted environmental monitoring in the Susquehanna River and Upper Chesapeake Bay to assess the radioecological impact of radioactivity released by the Peach Bottom Atomic Power Plant. Located approximately 5 km north of the Pennsylvania-Maryland border, the plant has discharged low levels of radiocesium (Cs-134 and Cs-137), radiozinc (Zn-65) and radiocobalt (Co-60) to the river since initial operation in 1975. Low levels of these radionuclides have consistently been detected in biota and sediments of the lower Susquehanna and Upper Chesapeake Bay (McLean and Domotor 1988). Because these radionuclides are particle-reactive, they may be scavenged by particulate matter suspended in the water column, ultimately to be deposited on the river bottom, or transported downriver as particulates or resuspended sediment. Because sediments serve as ultimate sinks for these radioactive metals, we have extensively monitored radionuclide concentrations in the sediments of the Conowingo Pond, the Susquehanna Flats, and the Upper Chesapeake Bay. Results of this sediment monitoring program have provided information useful in describing the transport and fate of other particle-reactive pollutants introduced into the Susquehanna River.

METHODS

Surface sediments (<10 cm) were collected twice annually from 1981 through 1986 -- once in the spring (Apr/May) and once in the fall (Sep/Oct) -- when Susquehanna River flow is at its maximum and minimum, respectively. Collection sites in the Susquehanna River and Upper Chesapeake Bay are shown in Figure 1. Sampling locations in the Conowingo Pond are shown in Figure 2. Radionuclide concentrations (pCi/kg) in Conowingo Pond sediments were converted to mass units (curies) using sample volumes and sediment densities. Radionuclide mass in the upper 10 cm of sediment was estimated for 22 areas (cells) within the Conowingo Pond sampling grid by extrapolation of each sample to the sediment volume (to 10 cm) of the respective cell. Radionuclide inventories in surface sediments of the Conowingo Pond are summations of the radionuclide masses within each cell. Deep cores (ca. 200 cm) were taken at selected locations. Radionuclide mass balance budgets were estimated using monthly radionuclide release quantities reported by the power plant operator. This source term was adjusted for radioactive decay.

RESULTS AND DISCUSSION

As indicated in Figure 3 and Table 1, less than 12% of the decay-adjusted Co-60, 21% of the Zn-65, and 8% of the Cs-134 released by Peach Bottom was found on any sampling date within the upper 10 cm of Conowingo Pond sediments. Significant differences in seasonal (spring/fall) radionuclide inventory are not apparent (Table 1). This implies, that approximately 80% of the particle-reactive radionuclides released to the Susquehanna by Peach Bottom is either, or both: 1) transported in soluble or particulate-associated form beyond the Conowingo Dam, or

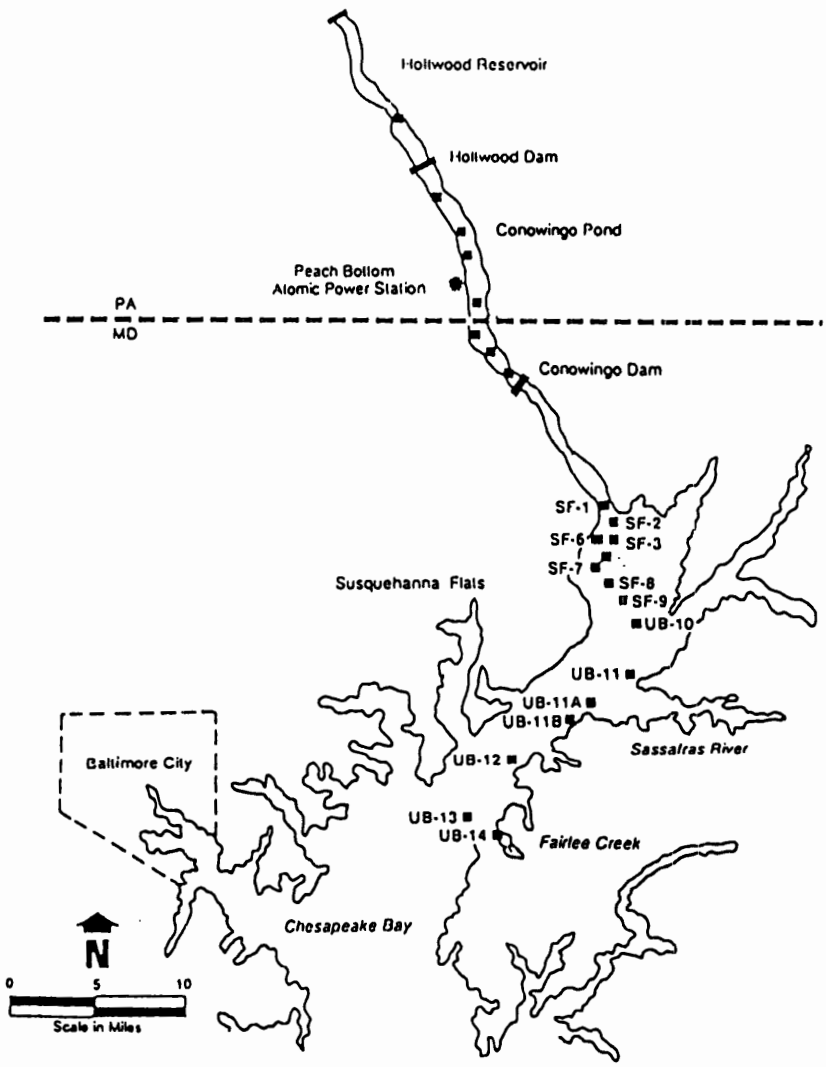

Figure 1. Sediment collection locations in the Susquehanna River and Upper Chesapeake Bay.

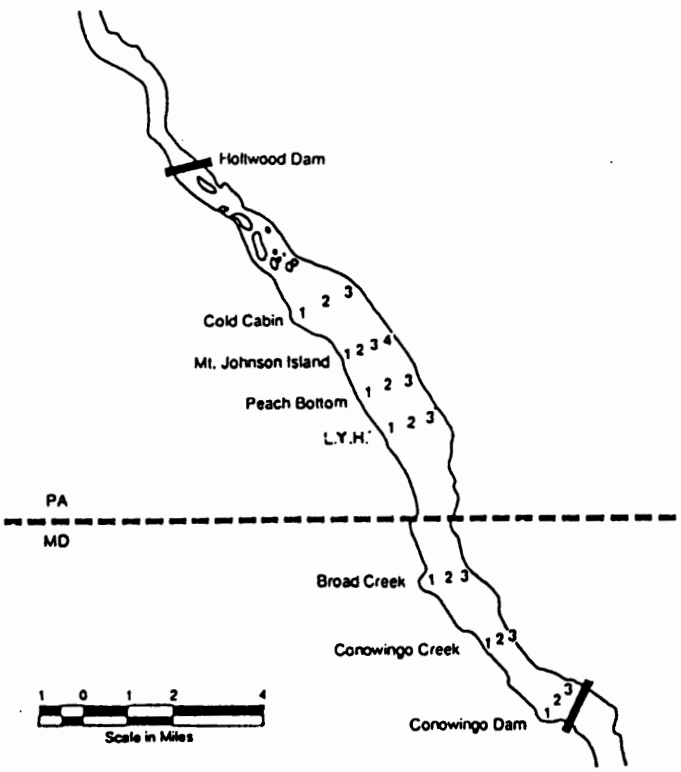

Figure 2. Sediment collection locations in the Conowingo Pond.

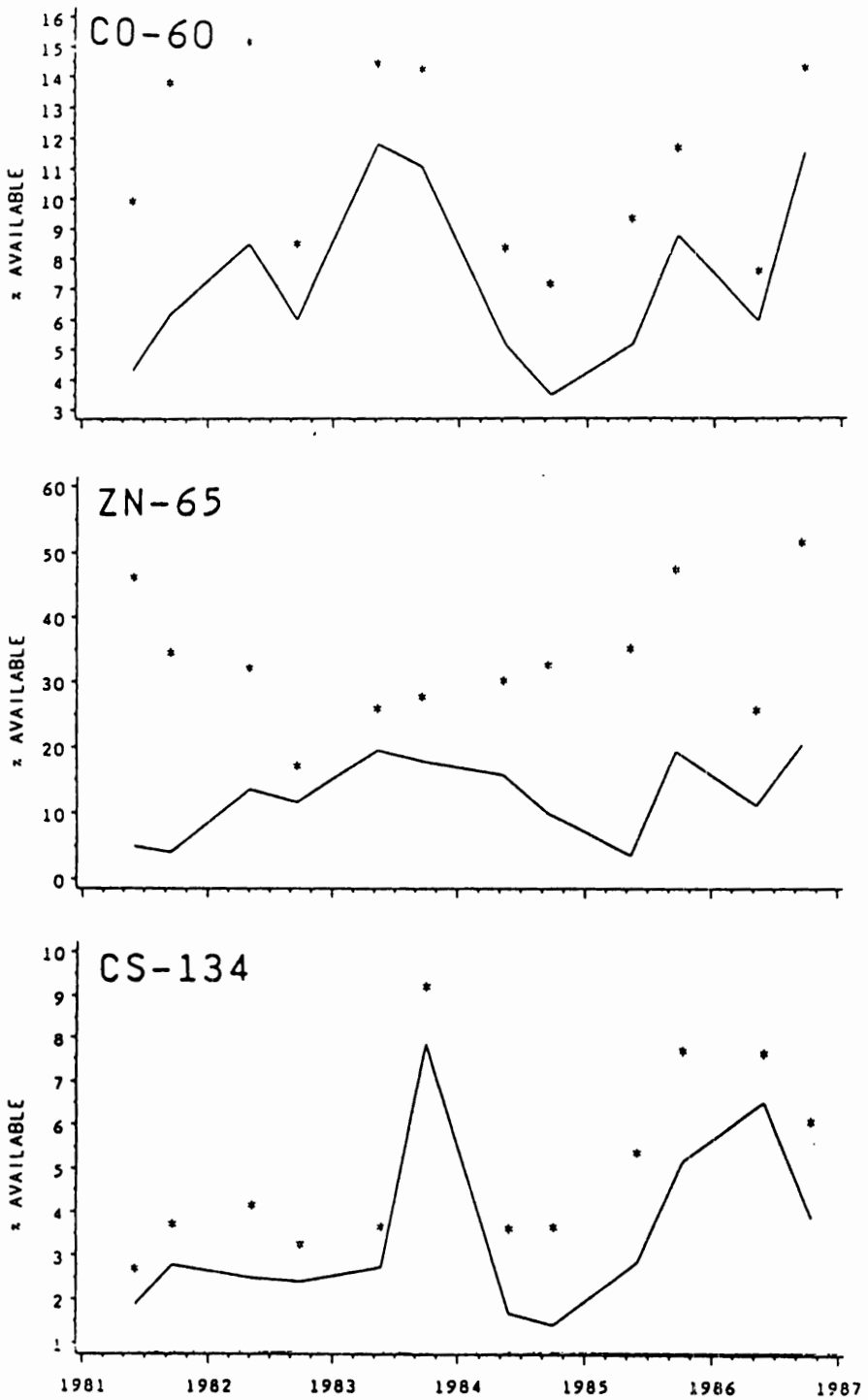

Figure 3. Percentage of decay-adjusted radionuclide release quantities present in surface (<10 cm) sediments of the Conowingo Pond, spring and fall, 1981-1986. Solid line describes mean estimates. Asterisks are upper limit estimates derived from 2σ counting uncertainties.

Table 1. Percentage of decay-adjusted radionuclide release quantities present in surface (<10 cm) sediments of the Conowingo Pond, spring and fall, 1981-1986.

	RADIONUCLIDE			
	Co-60	Zn-65	Cs-134	Cs-137
1981				
Spring	4.3	5.0	1.8	8.8
Fall	6.2	4.8	2.8	10.5
1982				
Spring	8.5	14.0	2.5	9.0
Fall	6.0	12.0	2.5	7.7
1983				
Spring	11.8	20.0	2.6	9.5
Fall	11.1	18.0	7.8	10.3
1984				
Spring	5.1	16.5	1.6	6.4
Fall	3.5	11.0	1.5	6.5
1985				
Spring	5.1	5.0	2.9	6.3
Fall	8.8	20.0	5.1	6.7
1986				
Spring	6.3	13.0	6.5	5.3
Fall	11.6	21.0	3.9	4.7
Mean, +/- 1 S.E.				
Spring	6.9 +/-2.8	12.3 +/-6.1	3.0 +/-1.8	7.6 +/-1.8
Fall	7.9 +/-3.2	14.5 +/-6.3	3.9 +/-2.3	7.7 +/-2.3

remains trapped within the reservoir and is buried below the 10 cm sample depth by subsequent sediment accumulation. Our data suggest that, with the exception of selected locations, appreciable burial of these radionuclides within Conowingo Pond does not occur. Data presented in figures 4 and 5 indicate that these radionuclides are consistently available in sediments collected from the immediate downstream vicinity of the Peach Bottom discharge on the western shore of the Conowingo Pond. The two nearest sampling locations (LYH-1 and BC-1) account for about 70% of the Conowingo Pond inventory. These figures present mean percentages of the available radionuclide mass for each sampling cell within the Conowingo Pond. The percentage of the total mass found within each cell during each sampling period (not presented) indicates that there is not a significant variation in percentage over time. This fact argues against appreciable radionuclide burial, as subsequent samplings would reflect higher percentages in later collections -- assuming sediment accumulation does not exceed 10 cm in the 5-6 month period between spring and fall samplings.

Because surface grabs indicated that LYH-1 and BC-1 were areas of greatest radionuclide concentration within the Conowingo Pond, deep core samples were collected at these locations. Cores were also collected from two other locations on the eastern shore downstream of the Peach Bottom discharge (BC-3, CON CK-3). These locations were regarded as representative of locations downstream of BC (i.e., CON CK and DAM transects), given the general similarity in surface sediment inventories at these locations (see Figures 4 and 5).

Table 2 compares, for the same collection period, estimates of the percentage of decay-adjusted radionuclides within cells, derived from surface grabs and deep cores. Core samples were sufficiently deep as to account for all buried Peach Bottom-derived radioactivity (the deepest penetration for Cs- 134 was about 60 cm). It is apparent that some burial of radionuclides occurs at the LYH-1 and BC-1 locations, but little occurs at the other two sites. These data indicate that mass balance estimates derived from surface grabs alone underestimate the percentage of radioactivity retained within Conowingo Pond. The underestimation appears, however, to apply only to the BC-1 and LYH-1 locations as comparisons of deep core and surface estimates are not significantly different at the other locations. Therefore on these sampling dates, the buried increment at BC-1 and LYH-1 combined would result in an increase in mass balance for Conowingo Pond of 16% for Co-60, 5% for Cs-134, and 10% for Zn-65. Given the maximum percentages presented in Table 1, adjusted mean mass balances for Conowingo Pond are 28% for Co-60, 13% for Cs-134, and 31% for Zn-65. It was not possible to obtain cores for all locations, however, we feel the data collected provide a reasonable estimate of the degree of retention of these radionuclides within Conowingo Pond.

Although there is no significant net difference between spring and fall mass balances for the Conowingo Pond as a whole (Table 1), there appear to be seasonal differences in the distribution of radioactivity among sampling grid cells (Figures 4 and 5). During both spring and fall, western shore locations account for higher percentages of total available mass than mid-river or eastern shore locations. Collections made

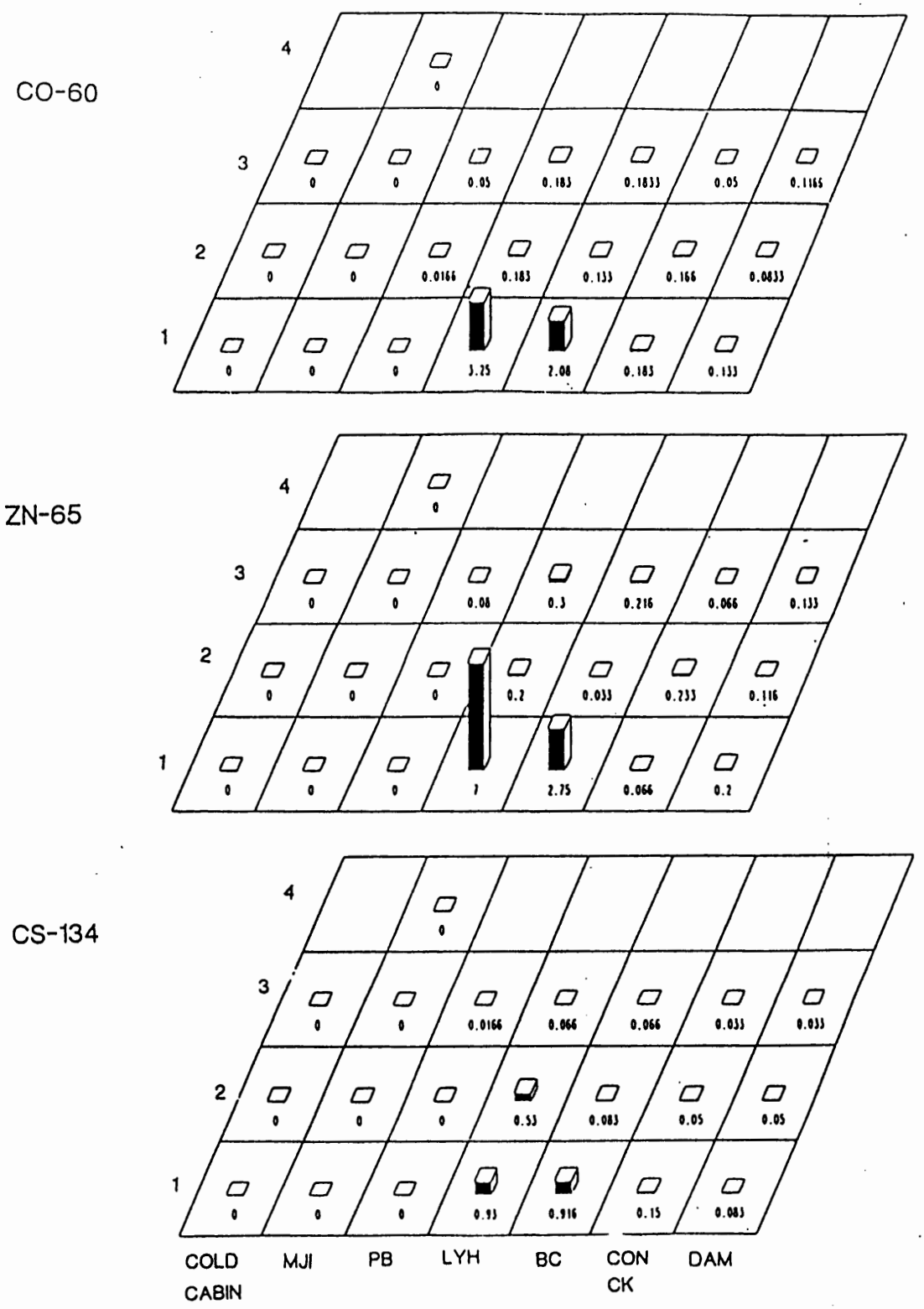

Figure 4. Mean percent of decay-adjusted radionculide release quantities in each Conowingo Pond sampling location, spring collections, 1981-1986.

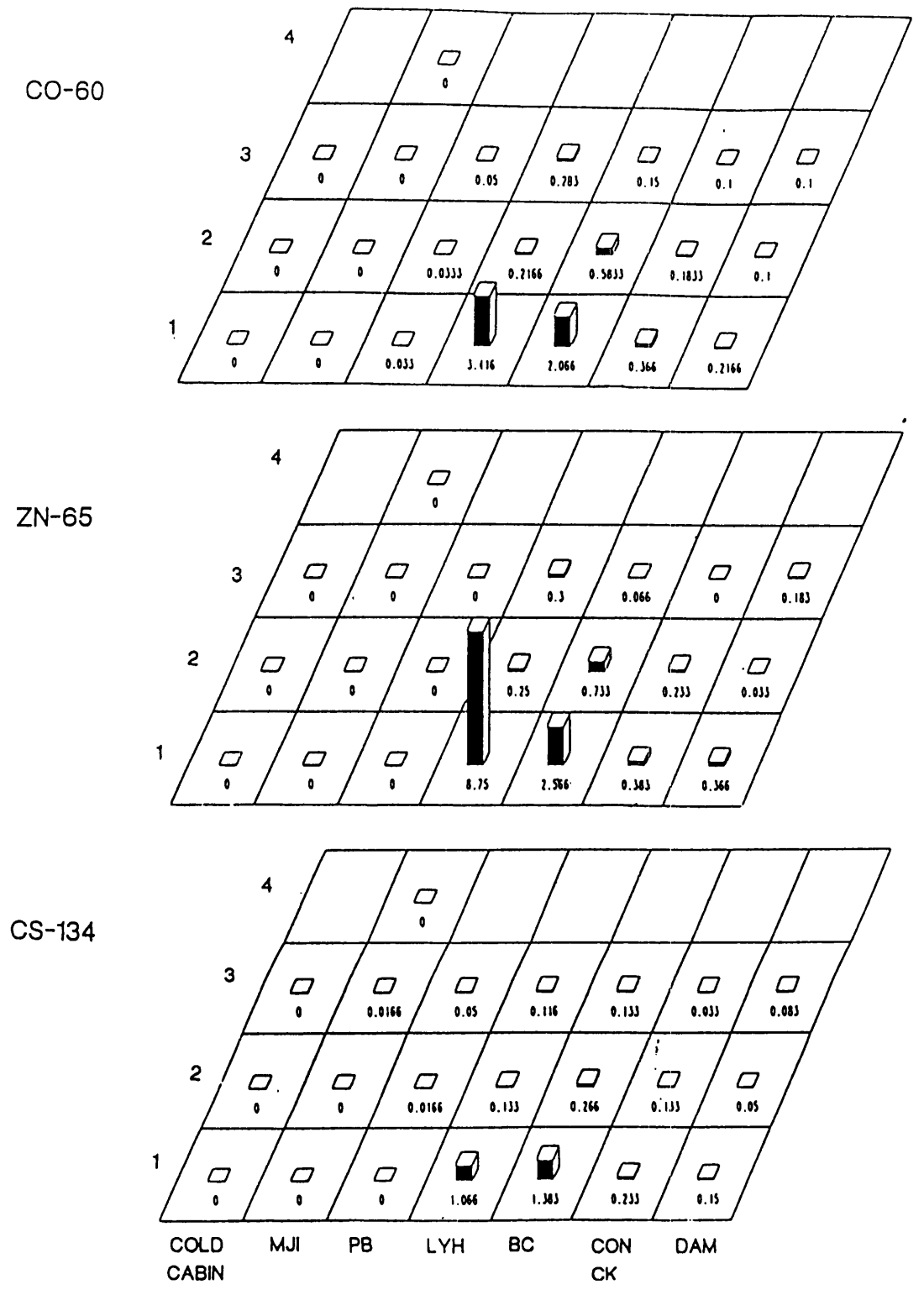

Figure 5. Mean percent of decay-adjusted radionuclide release quantities in each Conowingo Pond sampling location, fall collections, 1981-1986.

Table 2. Percentage of decay-adjusted radionuclide quantities at selected sampling locations for deep core (ca. 200 cm) and surface grabs (10 cm).

Date	Location	Station	Nuclide	% of Available (Core)	% of Available (Surface)	Delta % (Core-Surface)
MAY85	LYH	1	CO-60	15.10	3.06	12.04
MAY85	LYH	1	CS-134	3.88	0.80	3.08
MAY85	LYH	1	CS-137	8.41	0.91	7.50
MAY85	LYH	1	ZN-65	13.23	3.34	9.89
OCT85	BC	1	CO-60	5.58	1.38	4.20
OCT85	BC	1	CS-134	3.26	1.41	1.85
OCT85	BC	1	CS-137	37.04	1.17	35.87
OCT85	BC	1	ZN-65	2.99	4.53	-1.54
OCT85	BC	3	CO-60	0.15	0.34	-0.19
OCT85	BC	3	CS-134	0.51	0.56	-0.05
OCT85	BC	3	CS-137	13.85	0.61	13.24
OCT85	BC	3	ZN-65	0.21	0.00	0.21
OCT85	CON CK	3	CO-60	0.01	0.03	-0.02
OCT85	CON CK	3	CS-134	0.03	0.06	-0.03
OCT85	CON CK	3	CS-137	4.84	0.09	4.75
OCT85	CON CK	3	ZN-65	0.12	0.00	0.12

in the fall, however, exhibit a more equitable distribution of radionuclide mass throughout the reservoir. This suggests that the relatively greater Susquehanna River flow during the spring may scour and remove radionuclide-labeled sediments from channel and eastern shore stations within the Conowingo Pond, and that during the fall, lower river flows allow for greater cross-sectional migration of particulates and sediments within the reservoir.

There is little question that the Conowingo Pond serves as an efficient trap and deposition area for particulates suspended in the Susquehanna River. Radionuclide profiles in our core data indicate that average sediment accumulation rates range from about 3 cm/yr to greater than 6 cm/yr at the sampled locations (McLean et al., unpublished). Gross et al. (1978) have estimated that one-half to two-thirds of the particulate load of the lower Susquehanna is trapped behind the Conowingo Dam and two upriver dams (Holtwood and Safe Harbor). Olsen et al. (1981) have also shown accumulation rates to be high based on radiocesium (Cs-137) profiles in a Conowingo Pond sediment core. However, although Cs-137 (a weapon test fallout product) was observed at great core depths (>80 cm), PBAPS-derived Cs-134 was confined to the upper 10 cm. This information supports our data, further suggesting that, although Conowingo Pond serves as a sediment trap -- and does trap Peach Bottom radionuclides -- there is a turnover within the reservoir of surface sediments. Our data indicate that radionuclide-labeled particulates deposited in the reservoir -- with the exception of discrete areas in the downriver vicinity of the PBAPS discharge -- are not appreciably buried by subsequent sediment accumulation. Instead, they are continuously or periodically eroded, resuspended, and transported beyond the Conowingo Dam. The concurrent dilution and replacement of PBAPS-labeled sediments by particulates transported from upriver sources reasonably accounts for the relatively high rates of sediment accumulation observed by us and others in Conowingo Pond.

Radionuclides discharged by Peach Bottom, which are transported beyond the Conowingo Dam, are found in sediments on the Susquehanna Flats and in the Upper Chesapeake Bay (Table 3). Because sediment collection locations were selected primarily to define the extent of down-bay detectability of PBAPS radionuclides, they are not considered to represent surface sediments beyond the immediate sampling vicinity. Therefore, mass balance estimates of PBAPS-derived radioactivity in sediments were not calculated for this area. Outside the Conowingo Pond region, Peach Bottom radioactivity in sediments is most frequently detected and found at highest concentrations on the Susquehanna Flats. Radionuclide concentrations diminish with distance down-bay and become generally undetectable south of the Sassafras River. These diminishing concentrations are a function of particle dilution from other Upper Bay tributaries, and likely as well, desorption of particle bound radioactivity with increasing salinity (Olsen et al., in prep.).

The information derived from mass balance estimates of radionuclides discharged into the Conowingo Pond suggests, that given average annual river flows similar to those experienced during our study, less than 20% of particle-bound pollutants introduced into the Susquehanna River are deposited within the reservoir. This percentage is probably conservative (i.e., maximal) as it involves a radionuclide source situated on

Table 3. Mean, maximum, and minimum concentrations (piCu/kg) and number (N) of detectable concentrations (of 12 collections) for radionuclides in sediments collected from Susquehanna Flats and Upper Chesapeake Bay locations, 1981-1986.

Location	Co-60				Zn-65				Cs-134			
	N	Mean	Min	Max	N	Mean	Min	Max	N	Mean	Min	Max
SF-1	7	19	5	46	4	27	4	81	11	34	6	93
SF-2	2	10	8	12	2	23	11	35	11	18	3	37
SF-3	2	4	3	4	3	83	13	192	8	14	3	52
SF-6	2	4	2	6	0	0	0	0	6	14	7	27
SF-7	1	4	4	4	0	0	0	0	7	12	3	24
SF-8	5	7	3	10	2	18	4	42	9	23	9	37
SF-9	8	8	2	15	1	15	15	15	12	40	13	85
UB-10	2	5	4	6	1	12	12	12	9	29	10	61
UB-11	4	16	2	34	0	0	0	0	8	25	11	39
UB-11A	4	18	11	30	1	16	16	16	6	12	5	33
UB-11B	3	23	13	31	0	0	0	0	4	31	14	59
UB-12	6	37	21	45	2	26	7	46	7	22	6	43
UB-13	0	0	0	0	0	0	0	0	0	0	0	0
UB-14	0	0	0	0	0	0	0	0	1	4	4	4

the reservoir which discharges almost daily; a fact which would likely enhance local deposition. Most of the pollutant-labeled particulates which are deposited within the reservoir, are likely resuspended and transported beyond the Conowingo Dam. A small fraction of deposited pollutants may ultimately become buried within the sediment record of the reservoir.

REFERENCES

Gross, M.G., Karweit, M., Cronin, W.B., and Schubel, J.R. 1978. Suspended sediment discharge of the Susquehanna River to Northern Chesapeake Bay, 1966-1976. Estuaries, 1:106-110.

McLean, R.I., and Domotor, S.L. 1988. Environmental radionuclide concentrations in the vicinity of the Peach Bottom Atomic Power Plant, 1981-1984. Maryland Power Plant Research Program PPRP-R-9. Annapolis, Maryland.

Olsen, C.R., Larsen, I.L., Cutshall, N.H., Donoghue, J.F., Bricker, O. P., and Simpson, H.J. 1981. Reactor-released radionuclides in Susquehanna River sediments. Nature, 294:5838; 242-244.

Schubel, J.R. 1972. Distribution and transportation of suspended sediments in Upper Chesapeake Bay. In: The Environmental Framework of Coastal Plain Estuaries, B.W. Nelson, Editor. Geol. Am. Memoir No. 133: 151-167.

Understanding the Estuary: Advances in Chesapeake Bay Research. Proceedings of a Conference. 29-31 March 1988. Baltimore, Maryland. Chesapeake Research Consortium Publication 129. CBP/TRS 24/88.

Quantifying Pollutant Sources to Rock Creek Estuary: The Patapsco River, Sediment Remineralization, and Non-point Source Runoff

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INTRODUCTION

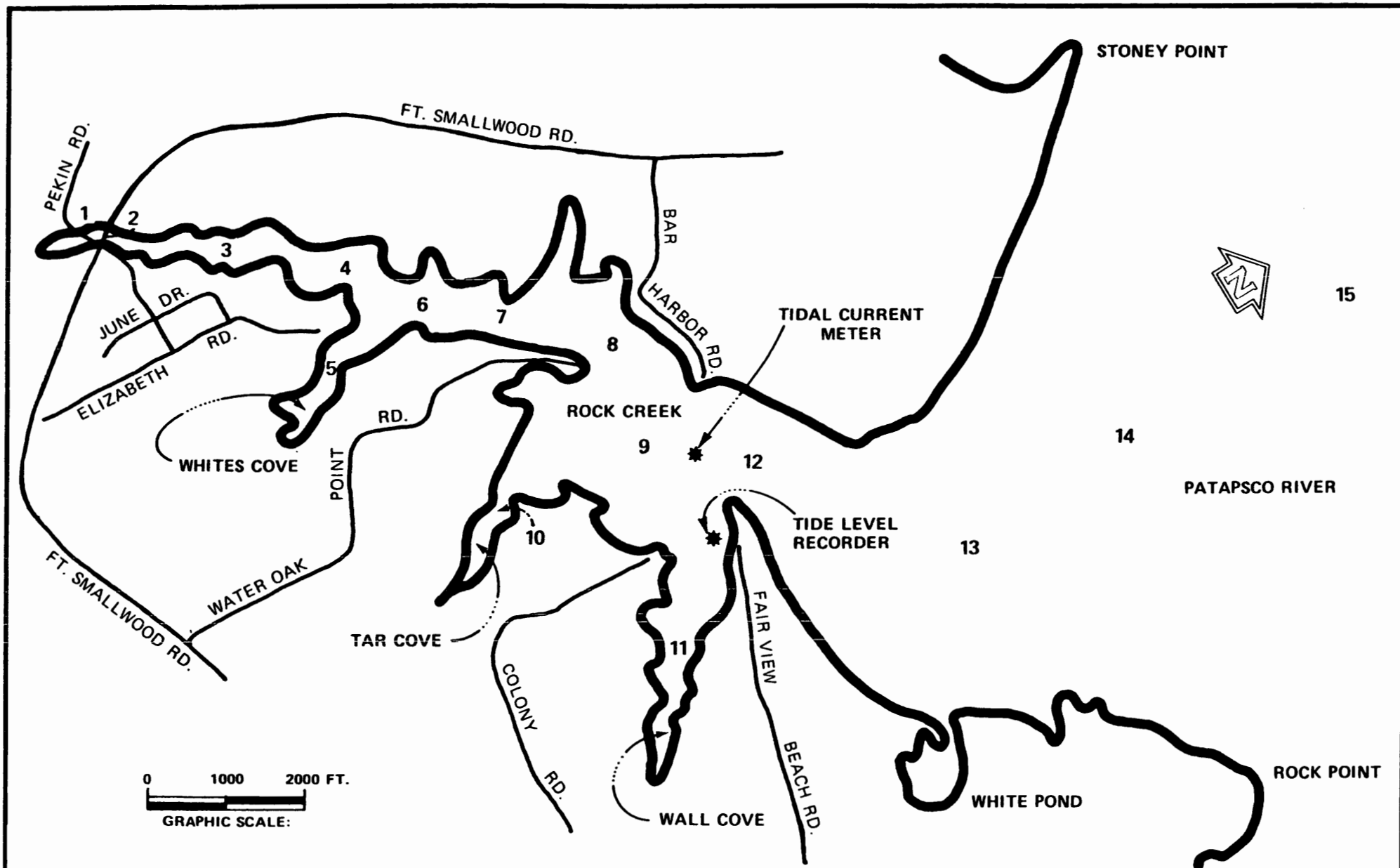
Rock Creek is a 353 hectare tidal creek in northern Anne Arundel County, Maryland that is tributary to the Patapsco River. The mean depth of the estuary is 3.0 meters. The length of the estuary is 5 kilometers, and the mean low tide volume is 9.98 million cubic meters. The watershed drainage area is 1022 hectares, of which 51.6% is urbanized, 5.4% is agricultural land, and 43% is forested. Rock Creek has serious water quality problems including wide fluctuations in dissolved oxygen, fish kills, and noticeable odors from hydrogen sulfide emissions, and dense algae blooms.

Water quality investigations were conducted in 1987 and 1988 by Anne Arundel County and the Maryland Department of the Environment (MDE). The purpose of the studies was to identify the major causes of the observed water quality problems and to formulate feasible management strategies to improve water quality. This paper will describe the method of determining the major causes of the water quality problems and quantify their relative impact on the Creek.

WATER QUALITY STUDIES

Field studies were conducted in 1987 to provide data for a nutrient mass balance for the estuary. Anne Arundel County funded studies included estuary water quality sampling on eight dates at five stations, measurements of organic sediment thickness in headwater areas, sediment chemistry, stormwater runoff monitoring, hydrodynamic investigations utilizing current meters, and in-situ measurements of sediment oxygen demand and nutrient remineralization. MDE field studies included estuarine water quality sampling on 15 dates at 16 stations, sediment thickness measurements, sedimentation rate studies, benthic community surveys, and hydrodynamic investigations utilizing dye studies. Figure 1 presents the estuarine water quality monitoring stations and the tidal current meter location. Further details regarding the methods utilized in the water quality studies and the results are provided in Dames & Moore (1988) and MDE (1987). Highlights of the study results are provided below.

Several key study results indicate the severity of the water quality problems in Rock Creek estuary. Dissolved oxygen (DO) concentrations throughout the study area were measured in the early morning and late afternoon by MDE. A sample plot of this data is



NOTE: Note that station 16, in the Patapsco River at the Brewerton Channel is not shown due to scale.

FIGURE 1

ROCK CREEK MONITORING STATIONS

Dames & Moore

presented in Figure 2. Morning surface DO concentrations within Rock Creek on June 30 averaged 4.16 mg/l (s = 2.2) while afternoon values averaged 11.32 mg/l (s = 3.03). Morning and afternoon DO concentrations and sectional estuary volumes were used to compute average daytime production rates. The average change in DO mass from the morning to the afternoon for Rock Creek inside Fairview Point (stations 1 to 11 shown in Figure 1) was 10,140 kilograms (Kg) or 0.35 mg/l/hr.

Chlorophyll a concentrations were also highly variable. Chlorophyll a data for August 25 are presented in Figure 3. On the plot the values for stations 2 and 3 are truncated. The laboratory results indicated concentrations in excess of 800 ug/l for these stations and no quality assurance problems were noted for the samples. The values were truncated in order to provide sufficient resolution on the plot.

Phosphorus and nitrogen concentrations are also interesting as shown in Figure 4. These concentrations are volume weighted average concentrations for Rock Creek inside Fairview Point. Orthophosphate concentrations are usually close to detection limits, while total phosphorus concentrations are often in excess of 0.2 mg/l. Dissolved nitrogen forms are also close to detection limits in June and July while total nitrogen concentrations during the summer exceed 2 mg/l, with concentrations in excess of 3 mg/l common. Peak total concentrations were observed in late July and August. Hydrogen sulfide odors were severe on June 22 and August 3. The importance of the relationship between odors and nutrient concentrations will be discussed later in this paper.

Sediment oxygen demand (SOD) and nutrient remineralization was measured near station 3 and near station 9 (see Figure 1). Duplicate measurements were made at station 3A on three days and at station 9A on one day. The duration of the experiments ranged from 4 to 6 hours. The results are presented in Table 1. Methane production rates were consistently high with somewhat lower rates when overlying waters were anaerobic. The average C:N:P molar ratio for releases of methane, ammonium, and orthophosphate for station 3A was 301:30:1. The methane production deviates most from the Redfield ratio of 106:16:1. Nitrogen remineralization is also elevated relative to phosphorus. The deposition of iron from the SOD chambers may explain this relative lack of phosphorus remineralization.

Sulfide production was expected during the experiments, however all samples analyzed were below the detection limit of 0.066 mg/l. The resulting sulfide release rate was therefore less than 6.73 mmoles/m2/day. This result in itself is significant. Bacterial production of methane was much higher than sulfate production, which suggests that the sediments were depleted of available sulfate and that methane production was the dominant bacterial reduction process. Low sulfate production rates have been reported when high methane production rates were measured (Martens and Klump, 1984).

DESCRIPTION OF THE SEASONAL MASS BALANCE

In order to evaluate the relative importance of the principal nutrient sources to Rock Creek, a seasonal mass balance was prepared for the period June 17 to August 25, 1987. This period was selected because tidal current meters were in place for the period June 19 to

FIGURE 2. ROCK CK. DISSOLVED OXYGEN

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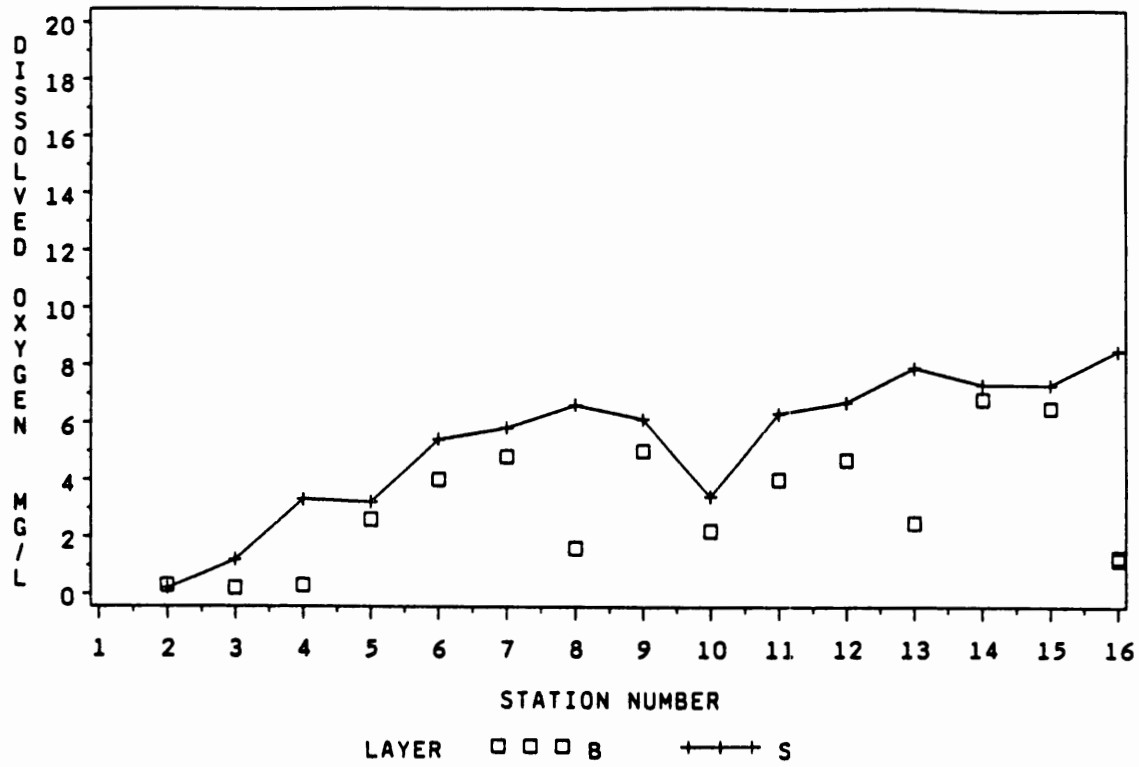

ROCK CK. DISSOLVED OXYGEN

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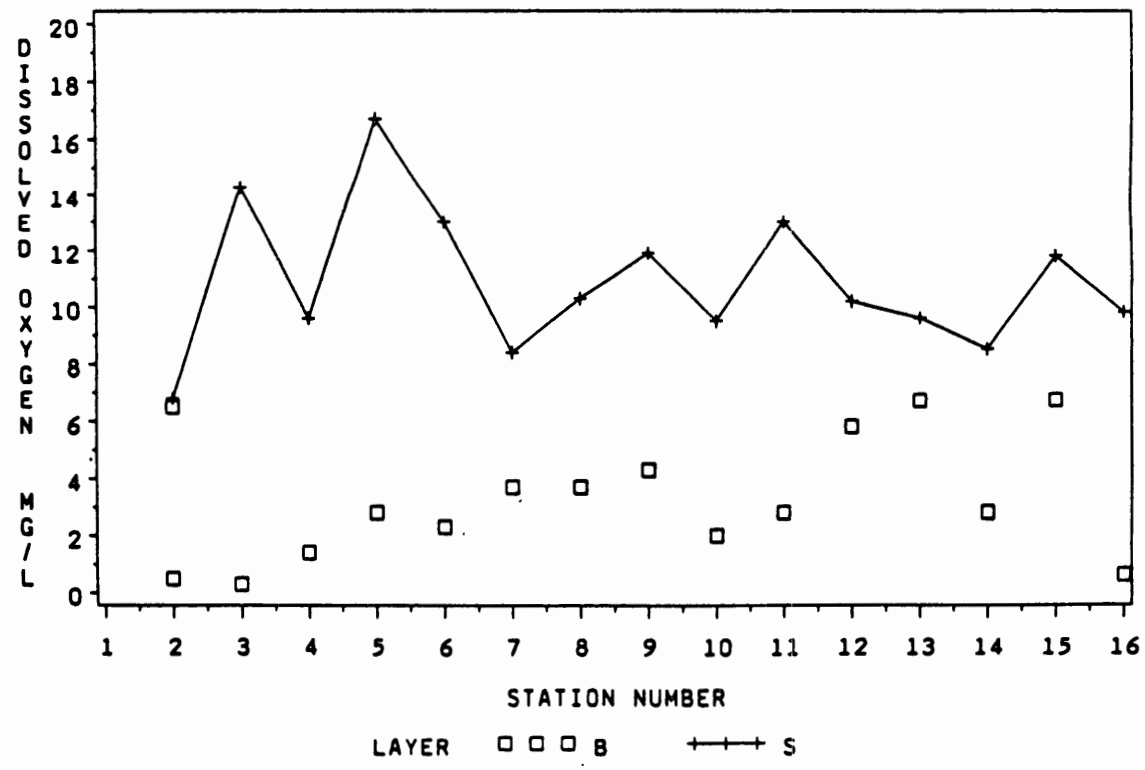


FIGURE 3. ROCK CK. CHLOROPHYLL "A"

DATE=25AUG87

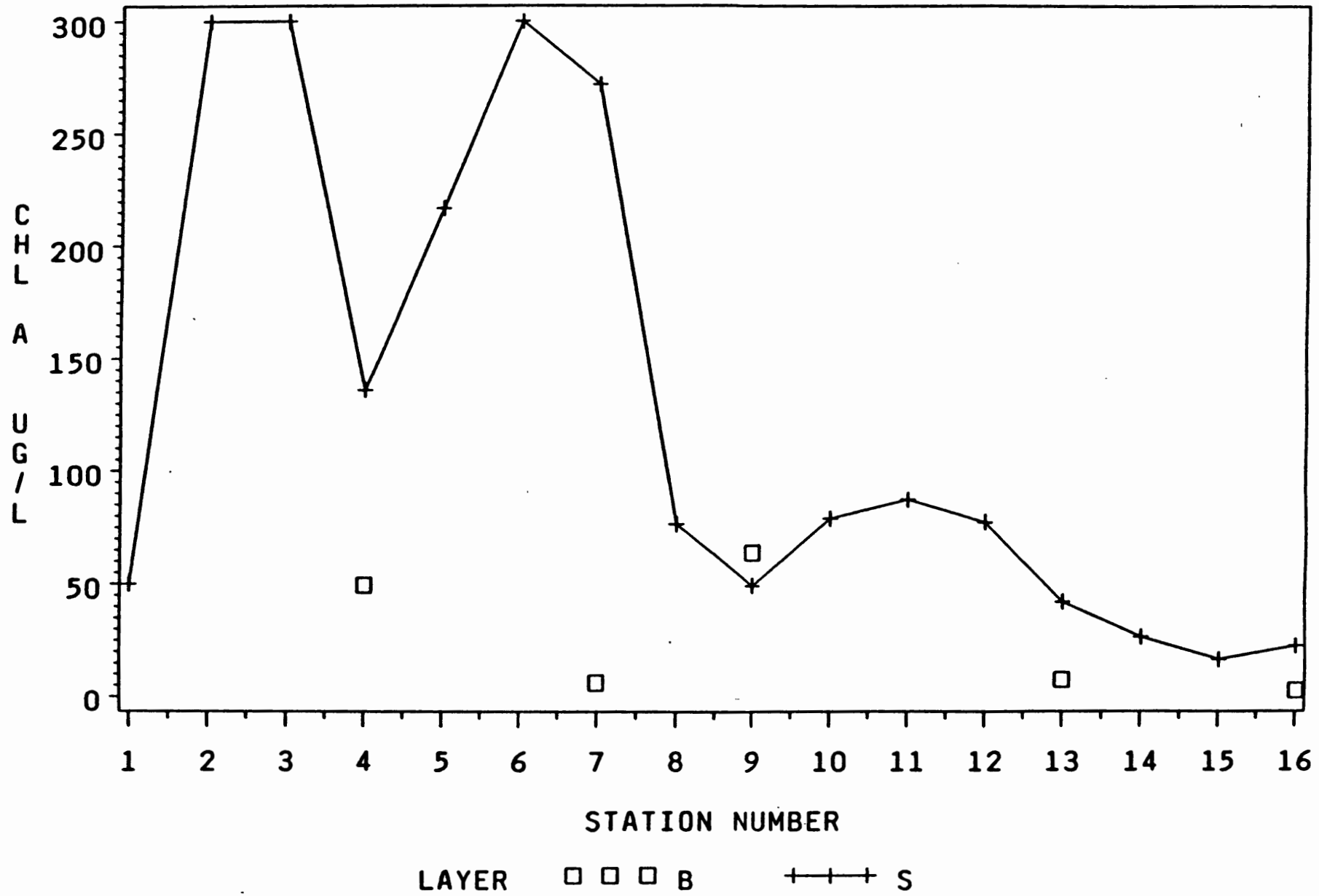
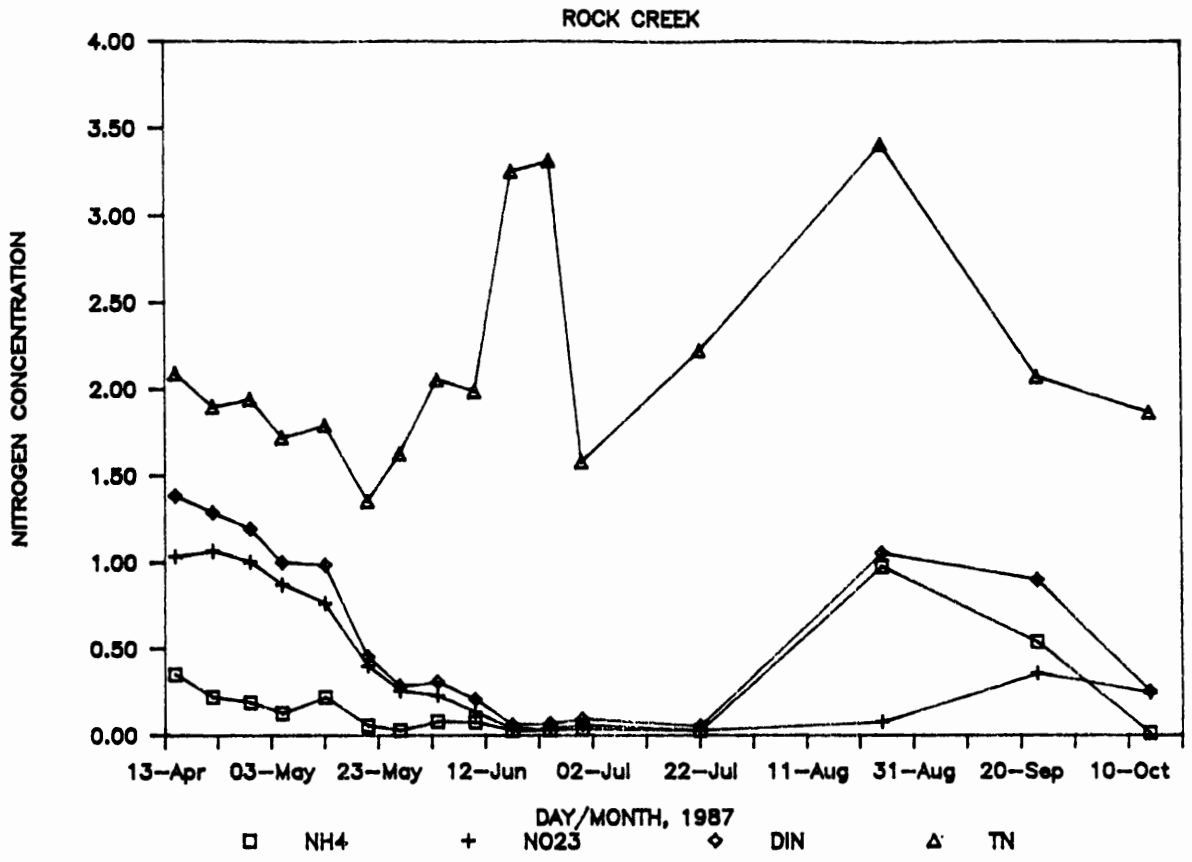


FIGURE 4. AVERAGE NITROGEN AND PHOSPHORUS CONCENTRATIONS IN ROCK CREEK



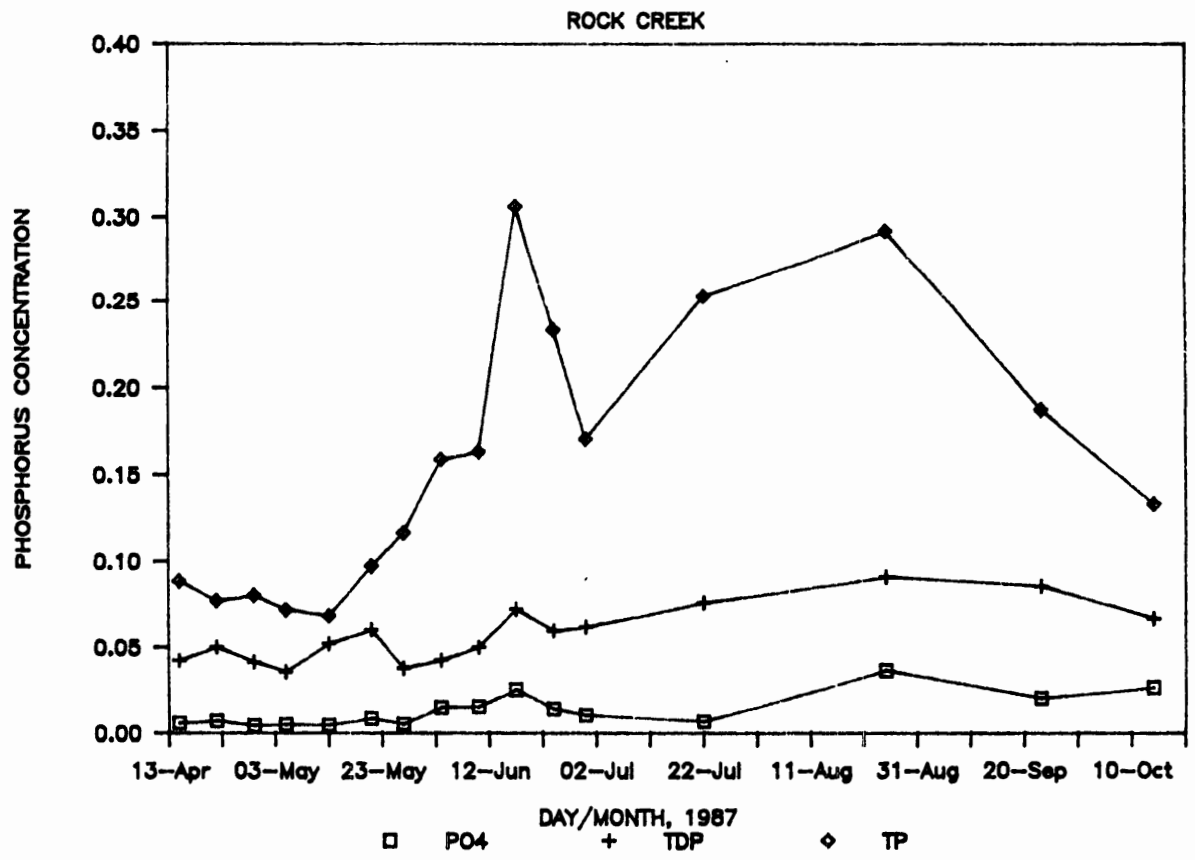


TABLE 1

Sediment Nutrient Remineralization Rates from Chamber Studies

Date	DO Range	Station	Chamber	PO_4-P (mmoles/m_2/da)	NH_3-N_2 (mmoles/m^2/da)	N:P	Methane as C (mmole/m^2/day)	C:P	Iron (mg/m^2/day)
9/3/87	4.3-2.0	3A	1	0.71	39.5	55.6	309	435	-35.9
9/3/87	7.7-4.6	3A	2	0.87	49.8	57.24	322	370	-29.9
9/8/87	2.8-0.7	3A	1	1.39	1.31	9.4	309	223	-66.6
9/8/87	2.5-0.8	3A	2	0.61	21.6	35.4	300	492	- 3.7
9/17/87	0.1-0.01	3A[a]	1	0.29	7.87	27.1	267	921	
9/17/87	0.1-0.01	3A	2	1.23	23.7	19.2	221	180	
Mean (std. dev)		3A		0.97 (.32)	29.5 (14.8)	30.4	292 (41)	301	
9/4/87		9A[b]	1	0.00	6.0		3.4		-452
9/4/87		9A	2	0.16	9.1	56.9	c		-501

[a]Chamber appeared to be improperly sealed.

[b]Excluding chamber 1, 9/17/87.

[c]Vial broken by vandalism.

August 21 and nine water quality surveys were conducted between June 17 and August 25. Mass balances for total phosphorus (TP) and total nitrogen (TN) were developed that explicitly considered nutrient inputs from Patapsco River inflows, sediment remineralization, stormwater runoff, septic systems, and atmospheric deposition. Daily inputs from these sources were summed, outputs due to settling algae and Patapsco River outflows were computed, and the net masses of TN and TP in Rock Creek inside Fairview Point were compared to nutrient masses computed from monitoring data.

Components of the mass balance are discussed in more detail below:

Patapsco River

Hydrodynamic forces cause a small but consistent bottom inflow (average velocity = 0.9 cm/sec) from the Patapsco River to Rock Creek. The confluence of Rock Creek and the Patapsco is constricted by Fairview Point and this constriction provided a convenient point to monitor the velocity of the inflows. The velocity, temperature, and salinity were measured every two minutes for approximately two months utilizing two Endeco 174SS meters placed 1.2 meters above the bottom and 1.2 meters below the surface. Total water depth was 5 meters. This constriction was used as the seaward boundary of Rock Creek. The width is 427 meters and the mean low tide cross section area is 1,211 m2. Based on salinity data collected from field surveys and from the current meters, it was assumed that the bottom cross sectional area was 594 m2. Average daily bottom tidal velocity was multiplied by the cross sectional area and observed bottom concentrations of TN and TP to compute the nutrient input from the Patapsco to Rock Creek.

The outflow from Rock Creek to the Patapsco was calculated as the sum of the hydraulic inputs minus evaporation. The nutrient output was calculated utilizing measured concentrations of nutrients in surface waters near Fairview Point (Station 13 shown in Figure 1).

In addition, the daily change in tides was considered in the mass balance. If the tide level at the end of the day was higher than at the beginning of the day, there was a net daily inflow from the surface layer. Similarly, if the tide level was lower at the end of the day, there was a net outflow of nutrients from the estuary. The surface area at different tide elevations was measured, and the tide inflow/outflow was computed as the average of the top and bottom areas multiplied by the change in tide height.

Sediment Remineralization

The phosphorus and nitrogen release rates were assumed to be constant throughout the study period. The selected phosphorus and nitrogen release rates were 35 and 490 mg/m2/day, respectively. These rates were selected based on remineralization rates measured in Rock Creek in September, 1987. Studies have shown that summer remineralization rates may be twice as high as fall rates (Boynton et al, 1986), therefore these measured rates of TN and TP remineralization may be considered as a lower bounds estimate. A reasonable upper bound for sediment remineralization could be as high as 70 and 750 mg/m2/day for phosphorus and nitrogen respectively. These upper and lower bounds for sediment remineralization were tested in the mass balance to determine the possible range of influence of sediment remineralization on the supply of nutrients fueling algal growth.

Stormwater Runoff

In order to estimate the runoff contribution to the nutrient loading of Rock Creek during the study period it was necessary to estimate the runoff volume and water quality. Stormwater runoff monitoring was conducted in the fall of 1987. Samples were collected from two stations in the watershed. The land use at one station was 56% residential, 27% forest, 3% commercial, and 14% construction, while the other watershed was 100% residential. Two storms were monitored in the mixed land use watershed while one storm was monitored in the residential watershed. The volume weighted average TP concentration for all monitored events was approximately 0.5 mg/l. The TN concentration was more variable, ranging from 1.6 to 14 mg/l. The median TP and TN concentrations (0.7 and 2.9 mg/l, respectively) for these monitored storms were used to represent stormwater quality in the mass balance analysis.

The summer runoff volumes into Rock Creek were estimated by comparing runoff volumes for monitored storms with runoff volumes for USGS gaging stations in nearby watersheds. The volumes of runoff for the two storms monitored in the mixed land use watershed were very similar to runoff volumes at the USGS gaging station at Bacon Ridge Branch at Chesterfield. Accordingly, runoff volumes (in centimeters) for the Bacon Ridge Branch gage for the summer of 1987 were used to estimate runoff to the Rock Creek watershed.

Septic Systems

The number of houses in the watershed was estimated utilizing Anne Arundel County topographic maps and 1985 aerial photographs. Estimates of per capita TP and TN loads were obtained from Metcalf and Eddy (1979) and it was assumed that there were 2.3 people per household. Further, it was assumed that all houses with in 152 meters of the estuary or tributary streams contributed nitrogen and phosphorus to the estuary. From these houses, it was assumed that all nitrogen and 50% of the phosphorus reached the estuary. The assumption regarding nitrogen is well supported in the literature (Canter and Knox, 1986; Sikora and Corey, 1976). Most references regarding phosphorus uptake in soils report higher removal rates than assumed in this mass balance (Sikora and Corey, 1976; Hansel and Machmeier, 1980). Accordingly, the assumption regarding phosphorus should be considered as an upper bounds estimate.

Atmospheric Deposition

Rainfall nutrient inputs were computed by utilizing rainfall data from the National Weather Service Baltimore Washington International Airport weather station and assumed TN and TP concentrations of 1.5 and 0.035 mg/l, respectively.

Settling

Settling rates of TN and TP for the mass balance were adjusted to provide a reasonable fit to the observed nutrient mass measured in Rock Creek. A realistic lower bound for the settling rate was selected partially based on settling trap data obtained from station R-64 in the Chesapeake Bay (Boynton et al, 1986)and partially based on deposition rates that would be necessary to support methane production rates observed in Rock Creek.

MASS BALANCE RESULTS

Starting with a known mass (based on measured concentrations from water quality surveys) of total nitrogen and phosphorus in the water column of Rock Creek on June 17, 1987, the various nutrient inputs and outputs discussed above were summed on a daily time step to obtain a value for the change in nutrient mass expected for the subsequent survey dates. With the settling loss term set to average TN and TP values of 26 and 1.2 mmole/m2/day, the predicted mass of nutrient in the water column is much higher than observed during the early summer(Figure 5). Since there was no reason to expect the settling rate in Rock Creek would be constant, settling was varied on a daily basis as a calibration parameter in order to match the predicted mass of nutrients in Rock Creek to the observed mass.

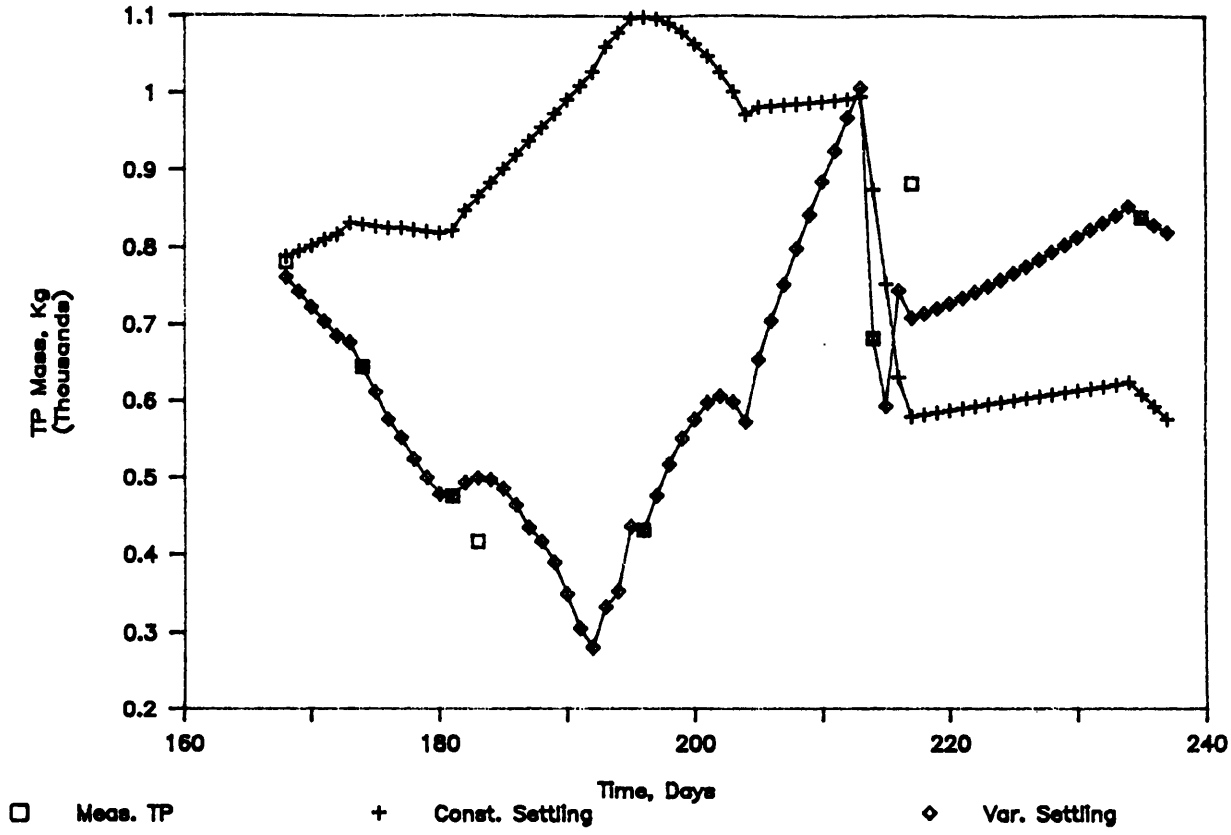
The following observations made in Rock Creek provide some justification for expecting very high sedimentation rates in the Creek.

1) There was high variability in phytoplankton populations during the summer of 1988. On monitoring dates June 17, June 24, June 30, July 7, July 22, August 3, August 5, and August 26, phytoplankton varied between dinoflaggelates, blue green algae, and diatoms. From June 24 to August 3, a different phylum was dominant on each monitoring date. Dinoflagellates again dominated in August. The rapid changes in phytoplankton dominance, and the disappearance of the previously dominant species during late June and July support the hypothesis of higher sedimentation during this period.

2) Measured sediment production of methane was very high. The carbon deposition rate necessary to provide organic carbon for the methane production would be in the range of 600 mmoles/m2/day. Assuming the Redfield C:N:P ratio of 106:16:1, the TN and TP deposition rate would be approximately 91 and 5.7 mmoles/m2/day (1365 and 175 mg/m2/day), respectively. The maximum TN and TP settling rates necessary in the mass balance to provide a close match between the observed and predicted nutrient masses were 75 and 2.2 mmoles/m2/day, respectively. Therefore, the methane production rates from Rock Creek sediments are consistent with the maximum release rates necessary to calibrate the mass balance.

3) The maximum settling rates required in order to obtain a mass balance with variable settling are much higher than maximum settling obtained at station R-64 of the Chesapeake Bay by Boynton et al (1986). The maximum TN and TP settling rates observed at R-64 were 17 and 0.81 mmole/m2/day. The mass balance derived settling rate for TP in Rock Creek is not significantly higher than the R-64 rate, however the TN settling rate in Rock Creek is much higher than the R-64 rate. This is

FIGURE 5. Mass Balance Results for Phosphorus



not unexpected since the average ammonium remineralization rate of 30 mmole/m2/day measured from Rock Creek sediments was more than double the highest rates (14 mmole/m2/day) reported by Boynton et al. (1986) for the Chesapeake Bay and other estuarine systems.

The resulting phosphorus and nitrogen mass balances are summarized in Table 2. It is evident from this analysis that the major sources of both nutrients are the Patapsco River and sediment remineralization. All other sources are quite minor relative to these two sources. As discussed earlier, the septic system loads and the stormwater phosphorus loads represent upper bounds estimates. The volume of runoff could be doubled and the phosphorus load would still remain a small source. The nitrogen concentration and the volume of runoff could be increased with negligible effects upon the mass balance.

Since there was some uncertainty concerning the exact magnitude of the nutrient exchange with the Patapsco, a range of exchange rates was tested for a reasonable upper and lower bound. The cross sectional area of the inflowing bottom layer of the Patapsco River was decreased to test the sensitivity of the mass balance to the Patapsco River influence. This change reduced the Patapsco River nutrient load from 58% to 48% of the total load.

The other major nutrient source is sediment remineralization. As mentioned above, there is reason to expect that summer rates, under higher temperatures and presumably higher sedimentation, could be as much as two times the rates measured in September. In order to evaluate this question, sediment remineralization rates were increased, and the impact on the mass balance was evaluated. This resulted in an increase from 41% to 60% for TP and 48% to 59% for TN for the portion of the nutrient inputs attributable to sediment remineralization.

The magnitude of the nutrient inputs to Rock Creek are high. The overall TP load during the summer period was estimated to be 6058 Kg or 5.6 g/m2. Jaworski (1981) has concluded that eutrophic conditions can be prevented in an estuary if the annual phosphorus load can be maintained below 1.0 g/m2/yr. A similar situation exists for nitrogen. The summer load to Rock Creek was estimated to be 74264 Kg or 68.7 g/m2. The acceptable nitrogen load according to Jaworski (1981) is 5.4 g/m2/yr. The phosphorus and nitrogen loads are 6 times and 13 times their respective acceptable loads.

SUMMARY

The mass balance approach described in this paper was a useful tool for determining the magnitude and relative importance of nutrient inputs to Rock Creek estuary from the Patapsco River, sediment nutrient remineralization, non-point source pollution, and other sources. The most important sources were the Patapsco River and sediment nutrient remineralization. A number of questions were raised from this investigation which can only be resolved by further study of this highly eutrophic system.

1) The settling rates of organic material in Rock Creek are essentially unknown. The mass balance analysis suggests that settling rates are extremely high. Settling rate measurements in Rock Creek are necessary before these values can be verified.

2) The observed methane release rates are extremely high compared to any other measurements reported in the literature. More

measurements are needed to determine if these rates are generally representative of methane release in Rock Creek.

3) Sediment nutrient concentrations are not unusually high in Rock Creek. This suggests that benthic-pelagic coupling must have a very rapid turnover rate in order to support the observed nutrient remineralization rates. Additional nutrient remineralization measurements would be helpful in verifying this assumption.

TABLE 2

Summary of Seasonal Mass Balance for Rock Creek
June 17 to August 25, 1987

Source	Phosphorus Kg	Phosphorus %	Nitrogen Kg	Nitrogen %
Patapsco River	3,511	57.9	41,886	56.4
Daily tide flux	-324	-5.3	-5,954	-8.0
Sediment remineralization	2,572	42.4	36,004	48.5
Stormwater runoff	108	1.8	414	0.6
Base flow	5	0.1	250	0.3
Septic systems	180	3.0	1,353	1.8
Rainfall	6	0.1	311	0.4
Total	6,058	100	74,264	100

Dames & Moore, 1988. The Rock Creek Estuary Study, Revised Draft, April 20, 1988, Bethesda, MD.

Boynton, W.R., et al., 1986. Ecosystem Processes Component Level 1 Data Report No. 3, July 1985-May 1986, CBL No. 86-56.

Canter, L.W., and R.C. Knox, 1986. Septic Tank System Effects on Ground Water Quality, Lewis Publishers, Inc.

Hansel, M.J., and R.E. Machmeier, 1980. "On-Site Wastewater Treatment on Problem Soils," Journal of Water Pollution Control Federation, Vol. 52, No. 3.

Jaworski, N.A., 1981. "Sources of Nutrients and the Scale of Eutrophication Problems in Estuaries," Estuaries and Nutrients, Human Press.

Metcalf and Eddy, 1973. Wastewater Engineering, 2nd edition, McGraw-Hill.

Sikora, L.J., and R.B. Corey, 1976. "Fate of Nitrogen and Phosophorus in Soils Under Septic Tank Waste Disposal Fields" Transactions of American Society of Agricultural Engineers, Vol. 19, No. 5.

Understanding the Estuary: Advances in Chesapeake Bay Research. Proceedings of a Conference. 29-31 March 1988. Baltimore, Maryland. Chesapeake Research Consortium Publication 129. CBP/TRS 24/88.

Nutrient Regeneration Rates in Chesapeake Bay Bottom Sediments Based on Bulk Properties

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INTRODUCTION

The bottom sediments in any estuarine system constitute the largest reservoir of trace elements and nutrients. It is therefore imperative that the interaction between the bottom sediments and the water column be understood. Currently fluxes from the sediment are measured by either pore water profiles, dome studies, or the incubation of sediments. Although these methods provide the best information on the processes occurring in the sediment, they are costly and time-consuming, thus are limited spatially and temporally. As a complement to these measurements, an upper limit average yearly rate of nutrient regeneration can be estimated. These estimates are independent of fluid analyses and are based solely on physical properties and composition of the sediments, coupled with sedimentation rates. Because these analyses are relatively simple and inexpensive wide spatial coverage can be obtained. Such a data set exists for the Chesapeake Bay.

The method is based on a steady-state approximation of the sedimentary processes occurring, and can be written as follows:

$$\frac{dC}{dt} = 0 = F_{in} - (F_{out} + F_{buried}) \qquad (1)$$

where: C - the concentration of either Carbon, Nitrogen, or Phosphorus species

F_{in} - the flux of nutrient, as a component of organic matter, into the sediment

F_{out} - the flux of nutrient out of the sediment, equal to the rate of regeneration

F_{buried} - the flux of organic material lost from the system due to burial of non-reactive organic matter

Equation (1) can be rewritten as:

$$F_{in} = F_{out} + F_{buried} \quad (1a)$$

or,

$$= (f_r + f_n) F_{in} \quad (2)$$

where: f_r and f_n are the reactive and non-reactive fraction of the input organic carbon. The reactive component input equals the rate of nutrient regeneration (F_{out}) and the non-reactive component equals the rate of burial of non-reactive organic matter (F_{buried}).

If it is assumed that the sediment at a site in the Bay is characteristic of the material input to the sedimentary reservoir at that site, then the rate of nutrient regeneration can be written as:

$$F_{out} = \omega \rho_s f_r C_o \quad (3)$$

where: ω - the sedimentation rate (cm/yr)
ρ_s - the density of solids in the sediments (g/cm^3)
f_r - the reactive fraction of the organic carbon
C_o - the initially deposited reactive carbon

Evaluation of this equation, using existing data, requires several conditions to be satisfied. These are:

1. The sediment type and porosity are constant at a given location, within the shallow sediment reservoir where the reactive carbon is depleted;
2. The average sedimentation rate has been constant, at any site, within the shallow sediment reservoir;
3. The amount of initially deposited carbon can be determined;
4. The properties of the organic matter are relatively uniform throughout the main stem of the Bay, i.e. the reactive and non-reactive components of C_o are the same everywhere,
5. Redfield's ratio (Redfield et al., 1966) can be applied to determine Nitrogen and Phosphorus fluxes, and;
6. Nitrification/denitrification and Phosphorus mineralization are not accounted for - thus the model yields the total potential flux of Nitrogen and Phosphorus.

These conditions above will be elaborated upon in the following text.

SEDIMENT DISTRIBUTION

Data on the bulk properties of the bottom sediments in the mainstem Chesapeake Bay were derived from the work of Byrne and others (1982) and Kerhin and others (1988). Sample stations were located on a one kilometer square grid in Maryland waters and a 1.4 kilometer offset grid in Virginia, a total of 5924 samples. Figure 1 shows the sample locations and segments used in this paper's analysis. At each sampling location the top 5 centimeters of sediment were collected with a grab sampler and analyzed for grain size, bulk density and percent water. Approximately one third of these samples were analyzed for carbon and sulfur content. Details of the analytic methodology may be found in the aforementioned reports. A sediment textural distribution map of the mainstem Bay, based upon these data and utilizing the ternary diagram and classification scheme of Shepard (1954), is presented in Figure 2.

As shown in Figure 2, sands, represented by the lower left corner of the ternary diagram, cover a much larger portion of the mainstem Bay than has been generally recognized. Over half of the bottom is composed

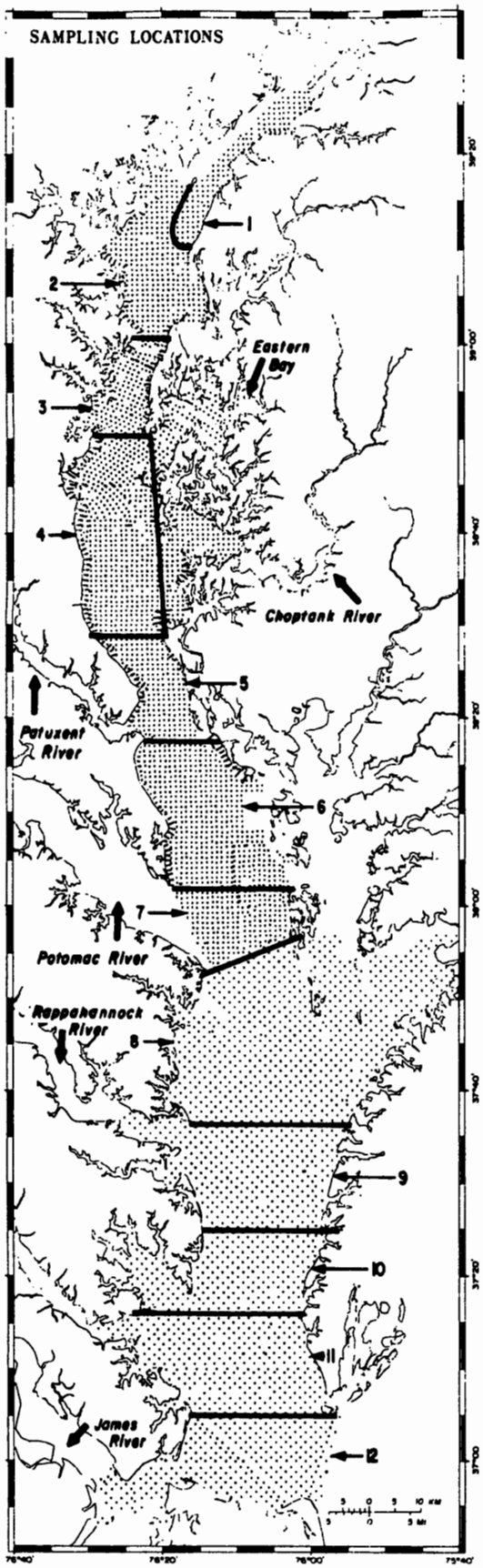
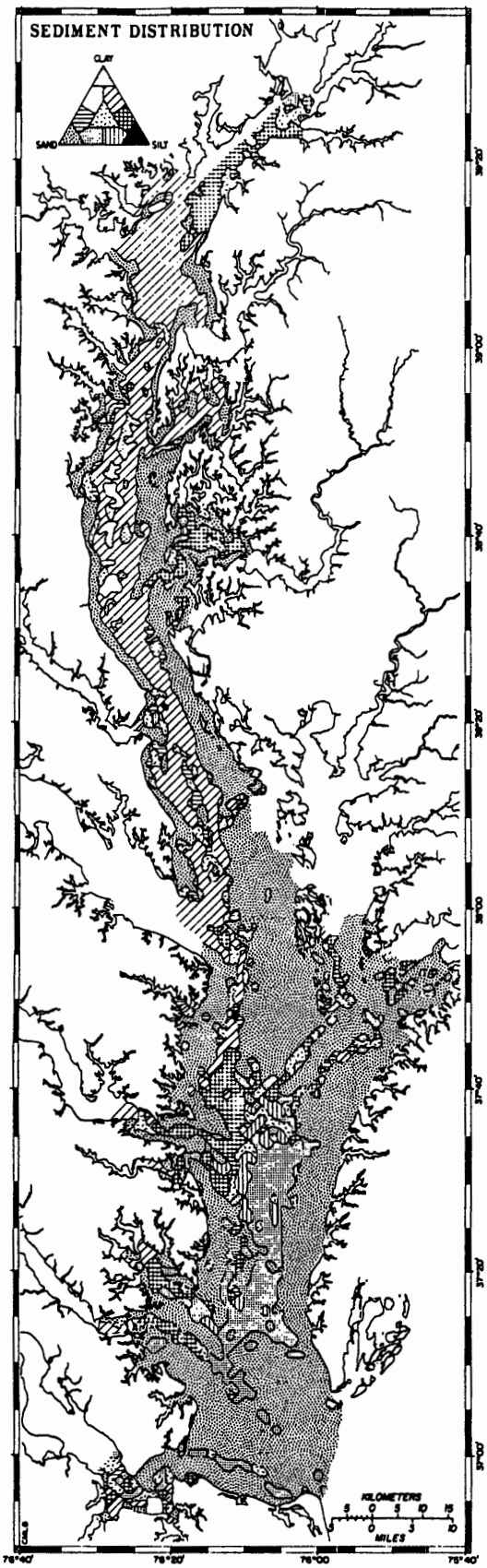


Figure 1: Location of surficial sediment samples used for the determination of bulk properties, adapted from Byrne and others (1982) and Kerhin and others (1983). Also shown are the physiographic segments within which sedimentation rates were determined.



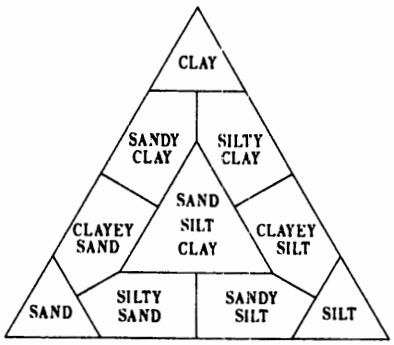


Figure 2: Textures of surficial sediments based upon the classification scheme of Shepard (1954). Adapted from Byrne and others (1982) and Kerhin and others (1983).

of sediments in which sand sized particles constitute more than 75% of the grains by weight (Table III). The area represented by the lower left three fields of the ternary diagram; the sand, silty sands, and clayey sands in which sand sized particles comprise more than half of the grains; covers 66% of the mainstem Bay. The importance of this large proportion of sand will be dealt with in the discussion section.

The bottom sediments in the deeper basins and axial channels largely consist of finer grained silts and clays (Figure 2). Throughout the Maryland portion of the Bay (segments 1-7) silty clays predominate with slightly coarser clayey silts occurring proximal to the Susquehanna River mouth, in segment 1. The finest grained clays are present only in the upper central portion of the Bay (segment 4). The nearly continuous field of silty clays extends to the south of the confluence with the Potomac River into Virginia where they grade into coarser clayey silts northeast of the Rappahannock River (segment 8). South of the Rappahannock River sands and silty sands dominate the bottom sediments even in the deepest waters (segments 9-12).

Within the dominant silty clay field extending between the Bay Bridge and Rappahannock River (segments 3-8) a wide variety of sediment types occurs in the deeper portions of the Bay (Figure 2). These include nearly all of the ten classes represented in the ternary diagram and have little if any relationship with water depth. Of particular note are the isolated pockets of sands and clayey sands which are located between the Patuxent and Potomac Rivers (segment 6). These pockets occur in over 12 meters of water depth and are separated from nearshore sources. The method of estimating potential nutrient fluxes describe herein incorporates these variations in textural distribution.

From the sediment data the density of solids at each sampling station was derived from the percent water using the formulas similar to those of Bennett and Lambert (1971). In addition the weight percent clay at each station was used to derive the originally deposited carbon content, and the weight percent of sand was used to modulate the sedimentation rates (see the following sections).

SEDIMENTATION RATES

Sedimentation rates for the mainstem of the Chesapeake Bay were determined by the method of bathymetric comparisons, with the initial data derived from Byrne and others (1982) and Kerhin and others (1988). Using the original NOAA survey sheets the water depths within cells 6 seconds on a side (≈150 by 200 meters) were averaged for the earliest (circa 1850) and the most recent (circa 1950) surveys for which there was an adequate density of data points. The results denote the changes in the height of the water column over the time interval spanning the two surveys. The data were rectified to the same mean low water datum by applying correction factors for eustatic sea-level rise (1 mm/yr) and the estimate of recent crustal warping for the Bay region. The latter ranged from over 2.6 mm/yr subsidence in the north to 1.0 mm/yr near the Bay mouth (Holdahl and Morrison, 1974).

The bathymetric comparison technique was utilized to estimate sedimentation rates because of its ability to provide measurements for the sandy sediments which have been shown to cover a large proportion of the Bay floor. Due to the nature of the data available from Byrne and

others (1982) and Kerhin and others (1988) the Bay was divided into 12 segments based upon basin geomorphology (Figure 1), with the averaged sedimentation rate determined within each segment. In Maryland the average rate of accumulation per year was calculated for both the muddy and sandy sediments within each segment. In Virginia only an average sedimentation rate could be calculated for each segment due to the characteristics of the original data set. It was felt that the bathymetrically determined rates provided a better estimate of the sedimentation rate than radiometrically determined rates due to the large areal expanse of sandy sediments, especially in the Virginia mainstem Bay.

The sedimentation rates for each of the geomorphic segments are shown in Figure 3 with rates for both muddy and sandy sediments determined in the Maryland segments and a single rate in the Virginia segments. The highest rate of nearly 0.8 cm/yr occurs in muddy sediments of the northern Bay adjacent to the Susquehanna River mouth. The rates decline southward from this high value to a minimum (approximately zero) in the upper middle bay (segment 4). Continuing south, rates rise again for both the muddy and sandy sediments to maxima of between 0.5 and 0 .6 cm/yr in segments 7, 8, and 10. It is interesting to note that the shape of the curve is similar to that reported from radionuclide dating (Officer et al., 1984) with the minimum bathymetrically determined rate occurring approximately 60 km further upbay than the radiometrically determined minimum. The rate for each segment was input to the model equation for each of the bottom sampling stations. In Maryland waters where two end member rates were determined, sands and muds (as shown in Figure 2), the rate utilized in the model equation at each of the stations was calculated using a linear approximation based on the percent sand present at each location.

INITIALLY DEPOSITED CARBON (C_o)

One of the most difficult parameters of prime importance to determine in estimating nutrient regeneration from bulk sediment data is the concentration of initially deposited carbon (C_o). In the literature to date there has been no independent determination of this parameter. Generally when examining sediment cores uniform deposition is assumed and C_o is assigned the value of the carbon content at the sediment-water interface; carbon values down-hole are subtracted from this assigned C_o to provide the amount of carbon oxidized by bacterial action, thus the amount of nutrient regeneration (Berner, 1980; Martens et al., 1978). This method cannot be readily applied to the estuarine sediments of the Bay because virtually all of the carbon data is from surficial samples (i.e. the interval 0-5 cm; no C_o can be assigned at the sediment-water interface), secondly much of the deposition in the Bay is episodic, not uniform, and finally, there is a significant terrigenous (coal) component of the carbon content in the northernmost portion of the Bay. An alternate approach to determine C_o uses the strong relationship of carbon to the clay-sized particle content of the sediment (Hennessee et al., 1986; Hobbs, 1983) coupled with a modification of Berner's (1972) and Sweeney's (1972) approach to describing the relationship of Sulfur to Carbon in marine sediments.

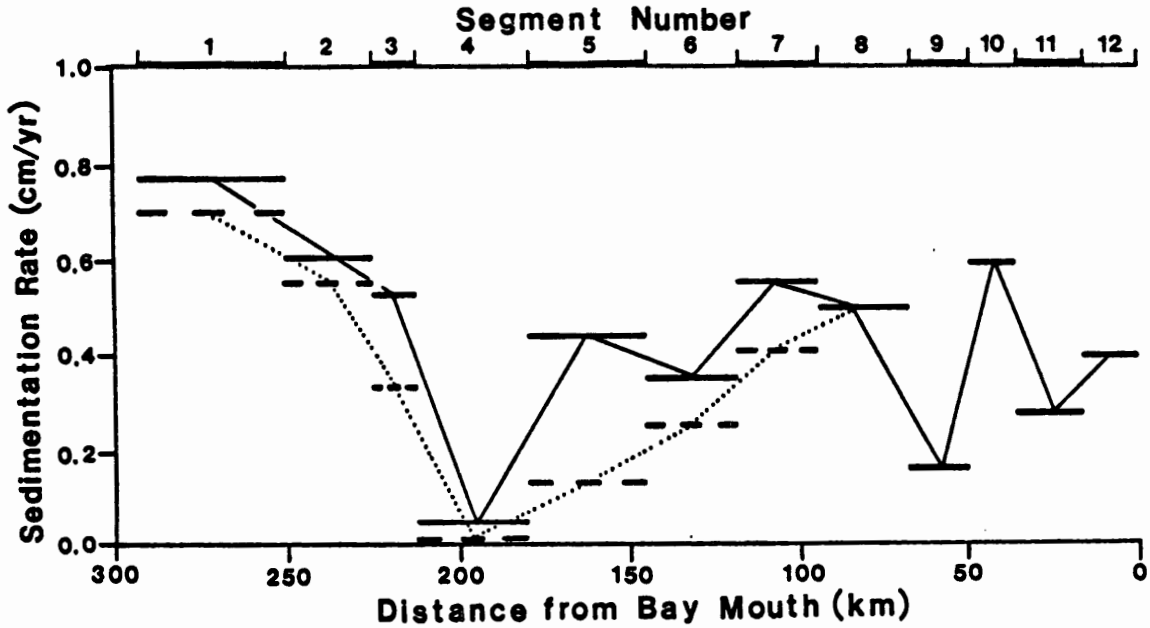


Figure 3: Average sedimentation rates (heavy lines) within each of the geomorphic segments delineated in Figure 1, connected by light lines to show trends. In segments 1-7 (Maryland) rates for muddy sediments are shown by solid lines and for sandy sediments by dashed lines. Only a single overall rate could be calculated in segments 8-12 (Virginia). All rates in cm/yr.

Sulfur in the sediments of the Bay is found in the form of metal sulfides formed during microbial sulfate reduction (Hennessee et al., 1986). This sulfur is deposited in the sediment at the expense of the oxidation of organic carbon. A skeletal diagenetic reaction can be written as follows:

$$4SO_4^{2-} + 9CH_2O + 4Fe(OH)_3 \rightarrow 4FeS + 9HCO_3^- + 10H_2O \qquad (4)$$

Based on this reaction the amount of sulfur found in the sediment at any given time is:

$$S(t) = \alpha(C_o - C(t)) \qquad (5)$$

where: S(t) - the bulk sulfur concentration measured at time t (g S/g dry wt sed; all percentages are based on dry sediment weight)
C_o - the initially deposited carbon
C(t) - the organic carbon content at time t
α - a constant containing the stoichiometric reaction factor and the conversion of carbon mass to sulfur mass

This is the approach taken by Berner (1972) and Sweeney (1972). Two major difficulties arise when trying to apply this equation. First, C_o must be independently defined. The second, and more important reason, is that not all of the sulfur generated by sulfate reduction is bound in the sediment. A significant fraction of the sulfide is recycled in the sediment and not mineralized. Jorgensen (1977) showed that only 10% of the sulfide generated was preserved in the sediments in a Norwegian fjord. Since Jorgensen's work the recycling of sulfur has been well documented (see Jorgensen, 1983).

Consequently, in order for Equation (5) to be validly applied to sedimentary systems recycling must be taken into account. If at any location in a sedimentary basin it is assumed the sediments are in steady state, then the fraction of generated sulfide preserved is a constant, designated as p (range 0-1). Equation (5) then becomes:

$$S(t) = \alpha\, p\, (C_o - C(t)) \qquad (6)$$

or,

$$C_o = C(t) + S(t)/\alpha p \qquad (6a)$$

Within the Chesapeake Bay, south of the Maryland Bay Bridge (latitude ≈39°05'), there is a strong correlation between the percent of the clay-sized grain fraction and organic carbon; see Table I. It follows that since C(t) is strongly related to percent clay, that C_o is also strongly related to percent clay. C_o can be calculated using the measured values of C(t) and S(t), the stoichiometric/conversion factor α, and an average p of 0.52 (Hill, 1987; Hill, in prep). The results of the linear regression of C_o as a function of percent clay are shown in Table 2 for comparison with the C(t) versus percent clay fit. The goodness of fit, in both Maryland and Virginia, is greater for C_o than C(t), and is quite good, thus allowing the use of percent clay to calculate C_o in the mainstem of the Bay south of the Maryland Bay Bridge. However there is some question as to the applicability of C_o, determined by the clay content, in the region effected by terrigenous carbon input (segments 1 & 2).

In order to establish whether the percent clay to C_o relationship is valid for segments 1 and 2, the amount of terrestrial carbon, principally coal, can be calculated as follows:

$$C_{ter(o)} = (C(t) + S(t)/\alpha p) - C_o\,(clay) \qquad (7)$$

where: $C_o(clay)$ is C_o calculated by the relation in Table 2
p is approximately equal to one (Hill, 1987; Hill, in prep.)

This equation in theory subtracts out the planktonic component of the organic carbon ($C_o(clay)$), leaving only the terrestrial or coal component. Figure 4 plots $C_{ter}(o)$ as a function of percent clay for segments 1 & 2; where $C_{ter}(o)=0$ (shown by the solid line) the organic carbon in the sample is solely planktonic. There are a number of samples that contain solely planktonic carbon, however most of the samples show a linear decrease in carbon content with increasing clay content; the lowest values, at high clay contents, are approximately equal to $C_{nr}(o)=0$. This is the expected behavior if the non-reactive carbon was coal dust which would be associated with the coarser size fractions. Consequently, percent clay will be used throughout the mainstem of the Bay to calculate C_o.

REACTIVE CARBON COMPONENT OF C_o

The next component of Equation (3) to be evaluated is the fraction of C_o which is reactive. This can be determined using Berner's (1980) multi-G model, applied to core data. Berner's model is based on the idea that there are several component fractions of organic matter in sedimentary environments. Each of the components decay at a different rate, but each follow a first-order decay mode. This can be written as:

$$C(x) = \sum C_i e^{-k_i(x/\omega)} \qquad (8)$$

where: $C(x)$ - the total measured organic content at depth x
C_i - the initial concentration of the i^{th} component
k_i - the first-order rate constant of the i^{th} component
x - depth into the sediment
ω - is the sedimentation rate

This equation can be rewritten to deal with fractional components (f_i) of C_o:

$$C(x) = C_o \Sigma (C_i/C_o) e^{-k_i(x/\omega)} \qquad (9)$$

or
$$= C_o \sum f_i e^{-k(x/\omega)} \qquad (9a)$$

However, this is not directly applicable to the Bay because C_o and ω vary with depth due to changes in sedimentation. Changes in C_o with depth can be determined by clay content, however little data is available on detailed down-hole sedimentation rates (Brush and Davis, 1984). The best data that exists use radiometric sedimentation rates which assume constant sedimentation. Equation (9) becomes:

$$f_T(x) = \frac{C(x)}{C_o(x)} = \sum f_i e^{-k_i(x/\omega)} \qquad (10)$$

where: $f_T(x)$ - the total fractional amount of $C_o(x)$ at depth x, which ranges in value from 1 @ x = 0 to f_{nr} @ x = ∞.
$C_o(x)$ - $C_o(clay)$ at depth x
ω - the sedimentation rate as determined by Pb^{210} techniques.

Table 1. Linear regression fits of C(t) and C_o as a function of percent clay.

$$C(t) = m(\%clay) + b$$

Segments	N	m	b	R^2	p
3-7 (MD)					
C(t)	548	0.0361	0.262	81%	-
C_o	548	0.0601	0.375	85%	0.52
8-12 (VA)					
C(t)	956	0.0402	0.229	70%	-
C_o	956	0.0662	0.320	76%	0.59

Table 2. Fractional components (f_i) of C_o and first order rate constants calculated in the regression fit of Equation (11); along with the sedimentation rates used, and the goodness of fit.

Station Number	f_1	f_2	f_{nr}	f_T	f_r (f_1+f_2)
55	0.33	0.36	0.32	1.01	0.69
62	0.22	0.41	0.39	1.02	0.63
63	0.48	0.28	0.26	1.02	0.76
64	0.21	0.28	0.51	1.00	0.49
83	0.55	0.20	0.24	0.99	0.70
85	0.36	0.29	0.39	1.04	0.65
					0.65±0.08

Station Number	$k_1(yr^{-1})$	$k_2(yr^{-1})$	$\omega(cm/yr)^*$	$R^2(\%)$
55	0.076	0.0071	0.87	82
62	0.170	0.0074	1.26	70
63	0.081	(-0.00024)	0.66	97
64	0.923	0.044	1.12	100
83	0.010	0.00096	0.19	94
85	0.103	0.00599	0.37	92

* Sedimentation rates from Helz et al., 1982.

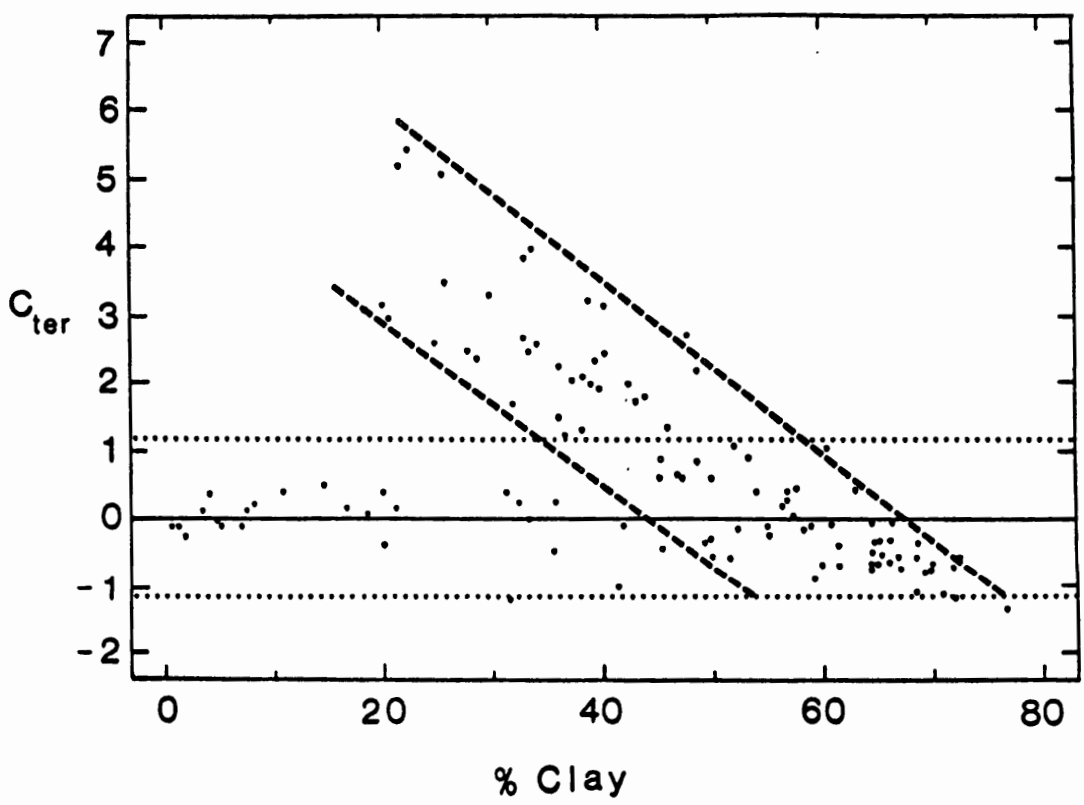


Figure 4: Plot of terrigenous carbon versus percent clay. The solid horizontal line represents planktonically derived carbon with the dotted lines showing the spread of data in segments 3-7. The linearly decreasing values of C_{ter} (encompassed by the heavy dashed lines) represent the coal component of organic carbon in segments 1 and 2.

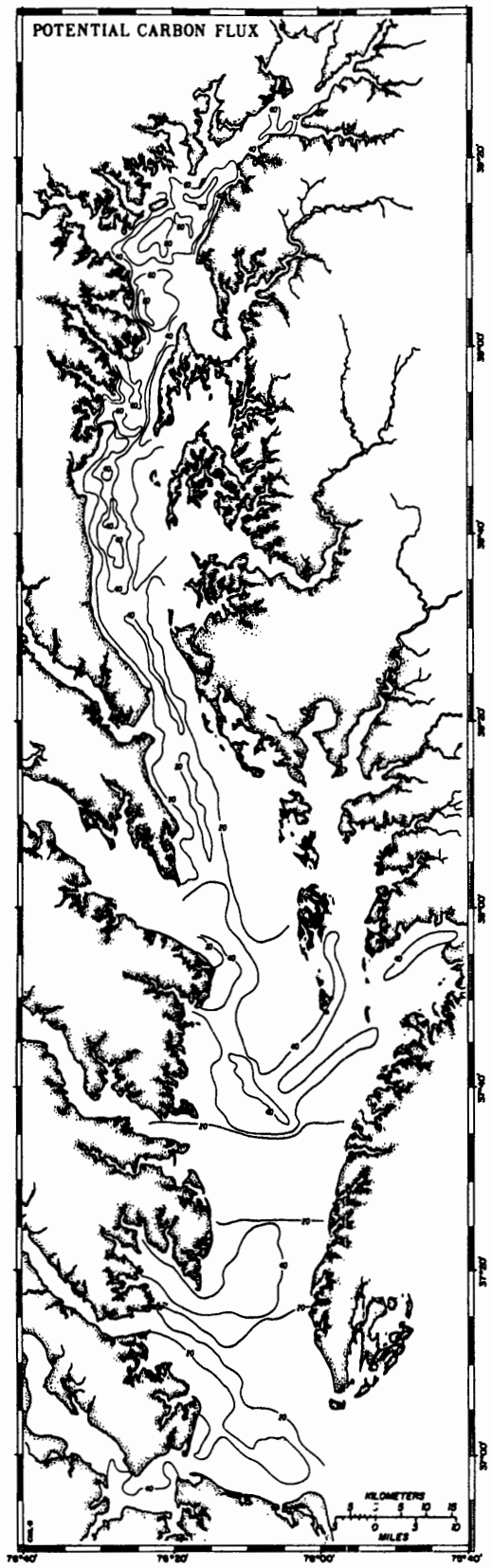


Figure 5: Potential flux of organic carbon out of the sediments (20 g/m^2-yr contour interval).

The parameter $f_T(x)$ is used in order to normalize for down-hole variability, thus allowing the computation for f_i, the various component fractions of the organic carbon.

Equation (10) has been applied to six cores taken throughout the mainstem of the Bay for which grain size, carbon, sulfur and Pb^{210} sedimentation rates were measured (MGS unpubl. data; Helz et al., 1982). A Marquardt-style regression (Draper and Smith, 1966) was used to fit the data to a function in the form:

$$f_T(x) = f_1 e^{-k_1(x/w)} + f_2 e^{-k_2(x/w)} + f_{nr} \qquad (11)$$

where: x, w, and $f_T(x)$ are measured parameters

The reactive carbon component, f_r, is the sum of f_1+f_2; the average value of f_r is 0.65±0.08. This value is within the range observed by others; Nixon (1981) noted in coastal marine waters 25-50% of the organic matter is fixed in the sediments (compared to 35% in this model). The variation in f_r is quite small (RSD = 12%) indicating that the planktonically derived organic carbon is relatively uniform throughout the Bay. This is to be expected due to the uniform behavior of C_o to clay throughout the Bay.

DISCUSSION/RESULTS

The results of the model are given in Figure 5 and Table 3. Figure 5 is a flux map of the mainstem of the Bay, F_{out}; in contours of 20 g of carbon/m^2-yr. The flux map mimics the sediment distribution (with the higher fluxes corresponding with the higher clay contents) but is strongly modulated by sedimentation rates. The highest fluxes occur where clay contents and sedimentation rates are both high. This is the case north of the Bay Bridge (segments 1 & 2) where the potential carbon flux locally exceeds 80 g/m^2-yr in the large field of silty clays. To the south of the bridge in segments 3 and 4 the potential flux of the muddy sediments decreases corresponding to the decrease in sedimentation rate even though the percent clay content remains high. In segment 6, the potential carbon flux in reduced to less than 20 g/m^2-yr in the isolated deep water coarser sediments. Conversely, the lowest potential carbon fluxes occur in the coarser grained sandy sediments located around the Bay's periphery and in the vicinity of the mouth, and in those areas where the sedimentation rate is lowest (segments 4 and 9).

Table 3 is a more quantitative statement of the model results. It shows the integrated flux within each sediment type in the Bay. The integrated flux was calculated by: obtaining the average flux in each sediment type in each Bay segment; this average flux was in turn multiplied by the areal extent, in square kilometers, of each sediment type in the segment, and; the results were totalled to give Table 4. The potential fluxes of Nitrogen and Phosphorus are calculated by using Redfield's ratio (Redfield et al., 1966) of planktonic matter composition.

There are several interesting features to note in this table. The first is that sandy sediments provide over 38% of the total flux. This is somewhat surprising because the sandy sediments are quite low in organic carbon. However, the low carbon content is offset by the large areal expanse of these sediments, >57% of the total bottom area, and the high solids density (≈1.3 g/cc for sandy sediments as opposed to ≈0.4

Table 3. Calculated potential flux of carbon, divided according to sediment type. For comparison to EPA estimates, Redfield's ratio is employed to calculate the Total Nitrogen (TN) and Phosphorus (TP) flux.

Sediment Type	Area (km^2)	% Area	Flux C (kg/yr) x 10^6	% Flux
SANDY CLAY	67.9	1	0.092	0
SAND	3715.0	57	74.956	38
SILT	1.9	0	0.009	0
CLAY	73.1	1	3.122	1
SAND-SILT-CLAY	327.6	5	12.941	7
CLAYEY SAND	67.9	1	1.084	1
CLAYEY SILT	496.6	8	22.679	12
SILTY CLAY	1207.3	18	65.592	33
SILTY SAND	546.8	8	14.003	7
SANDY SILT	87.0	1	2.031	1
	6526.6		196.509	

Total = 196.512 (C)
34.604 (N)*
4.784 (P)*

	EPA ANNUAL INPUT	
	Total	Benthic
(TN)	137.3	14.6
(TP)	13.7	3.4

* Based on Redfield's ratio of 106:16:1 (C:N:P)

g/cc for silty clays). As a result, sandy areas contribute significantly to the overall benthic flux and should not be discounted when detailed flux measurements are to be made in the Bay.

The other interesting feature to note arises from the comparison of the EPA flux estimates (EPA, 1982) to the flux estimates of the model. The model predicts input values significantly higher than the EPA estimates, even though the model only accounts for the main stem of the Bay, while the EPA estimates are for the Bay plus tributaries. It would be expected that the model estimates would be higher than measured fluxes because the mineralization of Phosphorus and denitrification processes can not be taken into account. If the EPA estimates of nutrient regeneration are comparable then 58% of the ammonium released by diagenetic processes is converted to N_2 by nitrification/denitrification processes, and 29% of the Phosphate liberated is mineralized in the sediments (most of the mineralization probably occurs in the northernmost portion of the Bay segments 1-3). However, since the model does not include the tributaries, which almost doubles the area involved, the model estimate would be substantially increased. Denitrification removes 5-25% of the nitrogen originally incorporated into the sediment (Nixon, 1981); up to approximately 50% of the regenerated Nitrogen. Consequently, either denitrification accounts for the removal of much greater than 50% of the Nitrogen in the sediments, or the EPA estimates of the total annual benthic input is quite low based on this model.

In summary, this model provides an independent means of determining upper limit flux estimates using relatively simple, inexpensive techniques. Better estimates would require more data from cores, with accompanying sedimentation rate data.

Bennett, R.H.; Lambert, D.N.; Rapid and reliable technique for determining unit weight and porosity of deep sea sediments; Marine Geology, v. 11, pp. 201-207; 1971.

Berner, R.A.; Sulfate reduction, pyrite formation, and the oceanic sulfur budget; (in) Nobel Symposium 20; The Changing Chemistry of the Oceans (eds. Dyrrsen, D. and Jagner, D.), pp. 347-361; 1972.

Berner, R.A.; A rate model for organic matter decomposition during bacterial sulfate reduction in marine sediments; Biogeochimie de la Matiere Organique A'L'Interface Eau-Sediment Marin, Colloque Internationaux du C.N.R.S., pp. 34-44; 1980.

Brush, G.S.; Davis, F.W.; Stratigraphic evidence of human disturbance in an estuary; Quaternary Research, 22 pp.; 1984.

Byrne, R.S.; Hobbs, C.H.; Carron, M.J.; Baseline sediment studies to determine distribution, physical properties, sedimentation budgets, and rates in the Virginia portion of the Chesapeake Bay; Virginia Institute of Marine Science, Final Report to the U.S. Environmental Protection Agency, 155 pp.; 1982.

Draper, N.R.; Smith, H.; Applied Regression Analysis; New York: Wiley; 1966.

Environmental Protection Agency; Chesapeake Bay Program Technical Studies: A Synthesis; U.S. Government Printing Office: 1982-509-660; 635 pp.; 1982.

Helz, G.R.; Sinex, S.A.; Setlock, G.H.; Cantillo, A.Y.; Chesapeake Bay sediment trace elements; Final Report to the U .S. E.P.A., 202 pp.; 1982).

Hennessee, E.L.; Blakeslee, P.J.; Hill, J.M.; Distributions of organic carbon and sulfur in surficial sediments of the Maryland portion of Chesapeake Bay; Jour. Sed. Pet., (in press); 1984.

Hill, J.M.; Sedimentary carbon and sulfur as indicators of biogeochemical processes; Geological Society of America annual meeting (Phoenix, Arizona); Abstracts with Program vol. 19, p. 703; 1987.

Hill, J.M.; The behavior of carbon and sulfur in the sediments of Chesapeake Bay; (in prep.).

Hobbs, C.H.; Organic carbon and sulfur in sediments of the Virginia Chesapeake Bay; Jour. Sed. Pet., v. 53, pp. 383-393; 1983.

Holdahl, S.R.; Morrison, N.L.; Regional investigation of vertical crustal movements in the U.S., using precise relevelings and mareographic data; Tectonophysics 23: 373-390; 1974.

Jorgensen, B.B.; The sulfur cycle of a coastal marine sediment (Limfjorden, Denmark); Limnology and Oceanography, v. 22, no. 5, pp. 814-832; 1977.

Jorgensen, B.B.; The microbial sulfur cycle; (in) Microbial Geochemistry; (ed.) Krumbein, W.E.; Blackwell Scientific Publications, pp. 91-124; 1983.

Kerhin, R.T.; Halka, J.P.; Wells, D.V.; Hennessee, E.L.; Blakeslee, P.J.; Zoltan, N.; Cuthbertson, R.H.; The surficial sediments of Chesapeake Bay, Maryland: Physical characteristics and sediment budget; Maryland Geological Survey, Report of Investigations No. 48, 160 pp.; 198?).

Martens, C.S.; Berner, R.A.; Rosenfeld; J.K.; Interstitial Water Chemistry of Anoxic Long Island Sound sediments. 2. Nutrient Regeneration and Phosphate Removal; Limn. Oceanogr., v. 23 (4), p. 604-617; 1978.

Nixon, S.W.; Remineralization and nutrient cycling in coastal marine ecosystems; (in) Neilson, B.J., Cronin, L.E., eds., Estuaries and Nutrients, The Humana Press; pp. 111-138; 1981.

Officer, C.B.; Lynch, D.R.; Setlock, G.H.; Helz, G.R.; Recent sedimentation rates in Chesapeake Bay; (in) Kennedy, V.S., ed., The Estuary as a Filter, Orlando, FL: Academic Press, p. 131-157; 1984.

Redfield, A.C.; Ketchum, B.H.; Richards, F.A.; The influence of organisms on the composition of seawater; (in) The Sea: Vol. II (ed. Hill); Wiley-Interscience, New York; 1966.

Shepard, F.P.; Nomenclature based on sand-silt-clay ratios; Jour. Sed. Pet., v. 24, pp. 151-158; 1954.

Sweeney, R.E.; Pyritization during diagenesis of marine sediments; (unpubl. Ph.D. dissertation): Los Angeles, Univ. of CA, 184 pp.; 1972.

Understanding the Estuary: Advances in Chesapeake Bay Research. Proceedings of a Conference. 29-31 March 1988. Baltimore, Maryland. Chesapeake Research Consortium Publication 129. CBP/TRS 24/88.

Delineation of Regional Sediment Resuspension Potential in Chesapeake Bay, with Implications for Bottom Sediment Management

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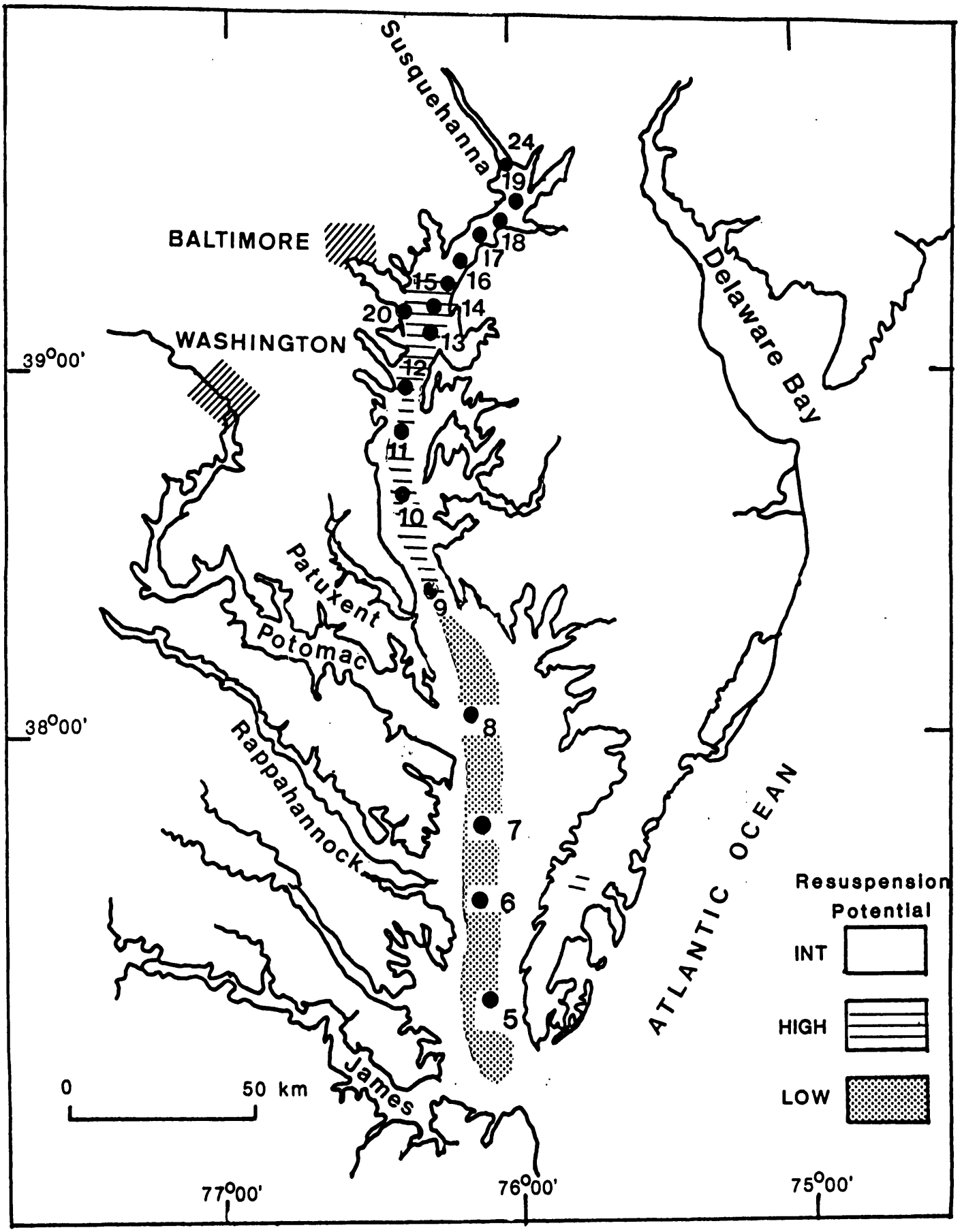
INTRODUCTION

Chesapeake Bay, the largest estuary on the east coast of North America, is a partially-mixed system in which seaward flowing, low-salinity water overlies landward flowing, high-salinity water with mixing occurring between the two water layers. Nearly 50% of its freshwater comes from the Susquehanna River which is also the primary source of its sediment. Sediment input depends upon Susquehanna flow rates, but in the decade 1966-1976, approximately 50 million metric tons were delivered to the upper Bay from the Susquehanna (Gross et al 1978). Estimates of sediment retention for the Bay vary between 90 to greater than 100% (includes sediment introduced from offshore) (Meade 1982; Biggs and Howell 1984; Schubel and Carter 1984). Sediment accumulation appears to be greatest in the northern part; lesser but significant accumulation occurs in the southern part; and minimal accumulation is found in the middle portion of the Bay (Officer et al 1984).

Geochemical studies have demonstrated the existence of near-bottom suspensions of fine-grained sediments, in excess of 60 cm thick, extending northward from the mouth of the Potomac (Station 8, Fig. 1) to Tolchester Beach (Station 15, Fig. 1) (Nichols et al 1981). Sampling has generally been limited to the Bay axis, consequently, the areal extent of these suspensions is unknown. They have been called "fluid mud", "sling mud", "fluff", "creme de vase". and "slib" when occurring in concentrations >10 g/l (Wells and Coleman 1981). No



1. Location of stations occupied for hydrographic and sediment sampling in Chesapeake Bay. Map also shows regions of high, intermediate, and low resuspension potential determined from this investigation.

universal definition is yet available for this phenomenon inasmuch as its recognition depends on the techniques used (Parker and Kirby 1982). In this paper, "fluid mud" is considered to be a suspension ranging between 10 to 480 g/l, corresponding to a density from 1.03 to 1.30 Mg/m^3. Kirby and Parker (1983) classified fluid muds as stationary and mobile suspensions which may show discontinuities or density stratifications, termed "lutoclines". Similar features have been observed in situ in suspensions from a dredged channel in the James River (Nichols et al 1978; Nichols 1985). These suspensions possess the rheological properties of a viscous fluid (Krone 1963; Migniot 1968; Pazwash and Robertson 1971; Bryant et al 1980; Faas 1981; Dyer 1986) and are believed to play an important, but as yet unknown role in estuarine transport processes (Officer 1981; Nichols 1985).

NATURE OF THE PROBLEM

River-borne sediment enters the estuary, mixes with sediment brought up from seaward reaches and, through processes not fully understood, settles through the water column to accumulate on the bottom (Schubel 1968, 1971, 1982; Zabawa 1978; Kranck 1975, 1986; Gibbs 1983; 1985). During passage from fresh into saline water, clay particles react with dissolved ions of various types, e.g., trace elements, nutrient ions, and toxic compounds, and, by settling, remove some of these materials from the water. This results in chemical enrichment of bottom sediments which when consolidated, serve as a quasi-permanent reservoir of these materials (Harris et al 1980; Gibbs 1982, 1986). Fluid mud suspensions tend to remain potentially mobile under dynamic tidal conditions and have been observed moving along the bottom as well-defined, cohesive sediment flows at velocities between 15-25 cm/s (Ingliss and Allen 1957; Kirby and Parker 1977).

The potential for an estuarine fluid mud to be resuspended is dependent upon many factors, e.g., morphologic, hydrographic, and rheologic. These latter properties include the "apparent" viscosity of the suspension, its rheological behavior throughout the range of shear rate and shear stress experienced during an accelerating tide, and its tendency to develop a yield stress during hindered settling. A simple measure of resuspension potential should be the difference between the yield stress of the suspension and the maximum tidal shear stress to which the suspension has been exposed during a tidal cycle. The purpose of this study is to determine the rheological behavior of fluid mud suspensions under simulated estuarine shear rate and shear stress conditons as a predictor of their potential for resuspension. Two aspects of the fluid muds seem critical to this problem: 1) their hindered settling behavior and density development during a slack water interval; and 2) the corresponding development of yield stress and viscous flow characteristics during the same interval. It is suggested that these properties interact in such a way as to either inhibit resuspension or determine when resuspension will occur in the shear stress field.

Background

In muddy estuaries, fine sediment particles settle out of the water column during slack water and may accumulate to form stationary suspensions which, if undisturbed, may consolidate into settled mud (Parker and Kirby 1982; Kirby and Parker 1983). Little is known about the development of time-dependent properties of short-term (1-4 hr) consolidation of fine sediment suspensions. Michaels and Bolger (1962) studied the settling rates and volumes of flocculated kaolin suspensions with respect to plastic and structural properties. Migniot (1968) showed that fine sediment suspensions from different rivers consolidated at different rates, dependent upon salinity and solids concentration. Owen (1970) demonstrated that a thin 1.1 cm mud layer, capable of withstanding erosion could be deposited from a dilute sediment suspension in four hours. Creutzberg and Postma (1979) showed that muds from the southern North Sea did not stabilize in consolidation periods shorter than 1.5 hours and would be resuspended by current velocities less than 12 cm/s. Hawley (1981) demonstrated that a thin mud layer, capable of resisting erosion at velocities as great as 14 cm/s could develop in 1.25 hours settling from a dilute (4 g/l) suspension in water having a salinity of 15 g/l.

Studies of the viscosity of naturally occurring clay/water suspensions were made by Krone (1963) who determined the viscosity of San Francisco Bay muds; Migniot (1968) who studied the relationships between viscosity, salinity, and solids concentration for muds from a variety of marine, estuarine, and fluvial environments; and, several studies by Dutch workers (NEDECO 1965, 1968; Allersma 1982) on the muds from Surinam and the Port of Bangkok, Thailand. Wells (1978) measured viscosities of muds from the central Surinam coast, and Faas (1981) observed different forms of viscous behavior which appeared to control resuspension patterns in the Rappahannock Estuary, and recorded similar behavior patterns from fluid muds on the NE Brazilian continental shelf (1985, 1986). Bryant et al (1980) found that muds from Rotterdam and Brisbane behave as Newtonian fluids in low concentrations (<10 %) and exhibit non-Newtonian behavior as concentrations increase. This same phenomena is observed in stationary and mobile suspensions from the Severn Estuary and inner Bristol Channel (Kirby and Parker 1983).

Methods

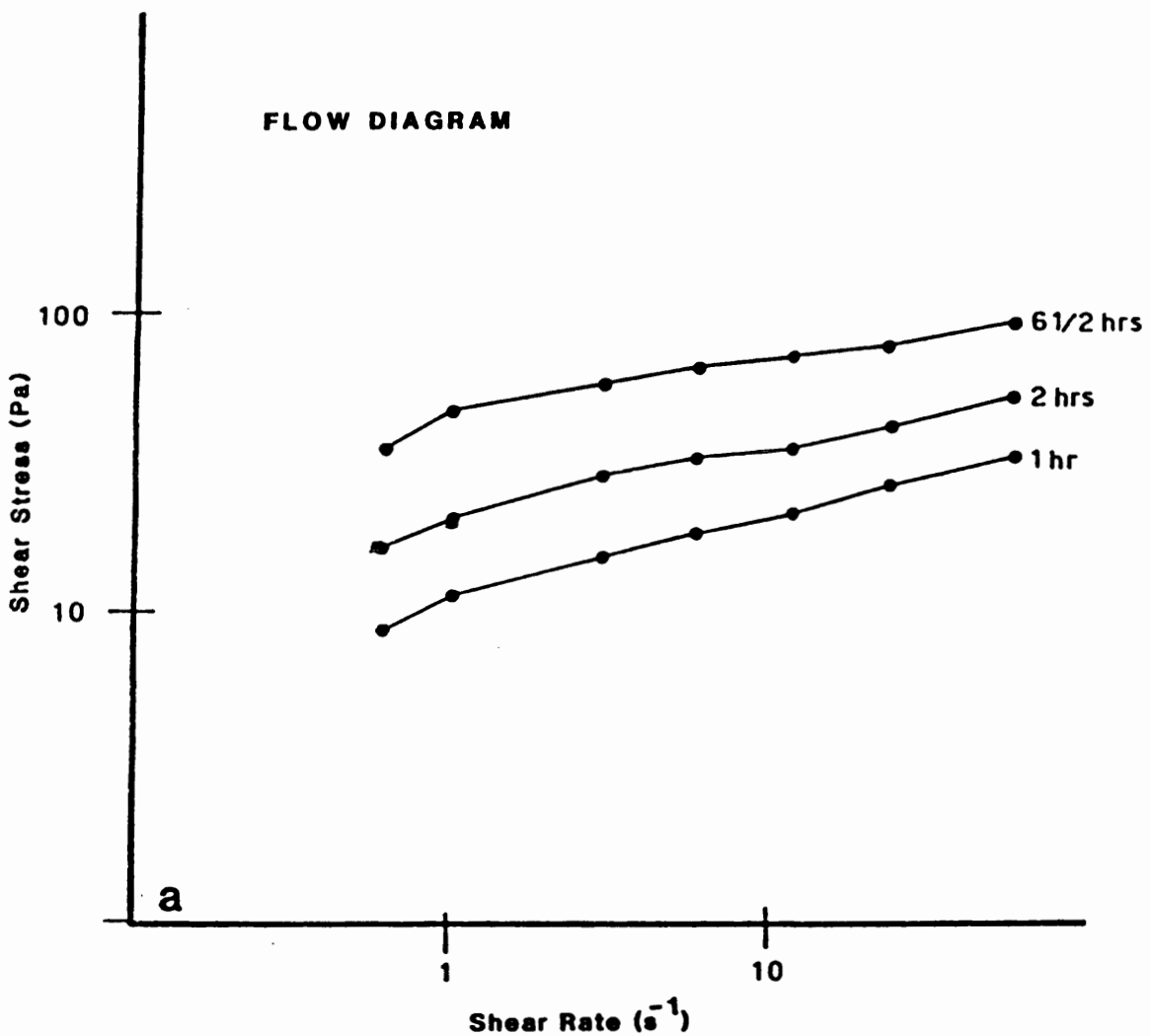
Samples were collected in a north-south transect along the Chesapeake Bay axis in October 1982, using the RV Ridgely Warfield. Bottom samples were obtained with a standard grab sampler and were kept refrigerated at 10 ^{o} C until analysis. The grab samples were sampled for natural water content onboard ship immediately upon retrieval. Grain size analysis was performed in the laboratory with the 152H Bouyoucos hydrometer (Bouyoucos 1962; Kaddah 1974). Organic matter content was determined by loss on ignition (Davies 1974).

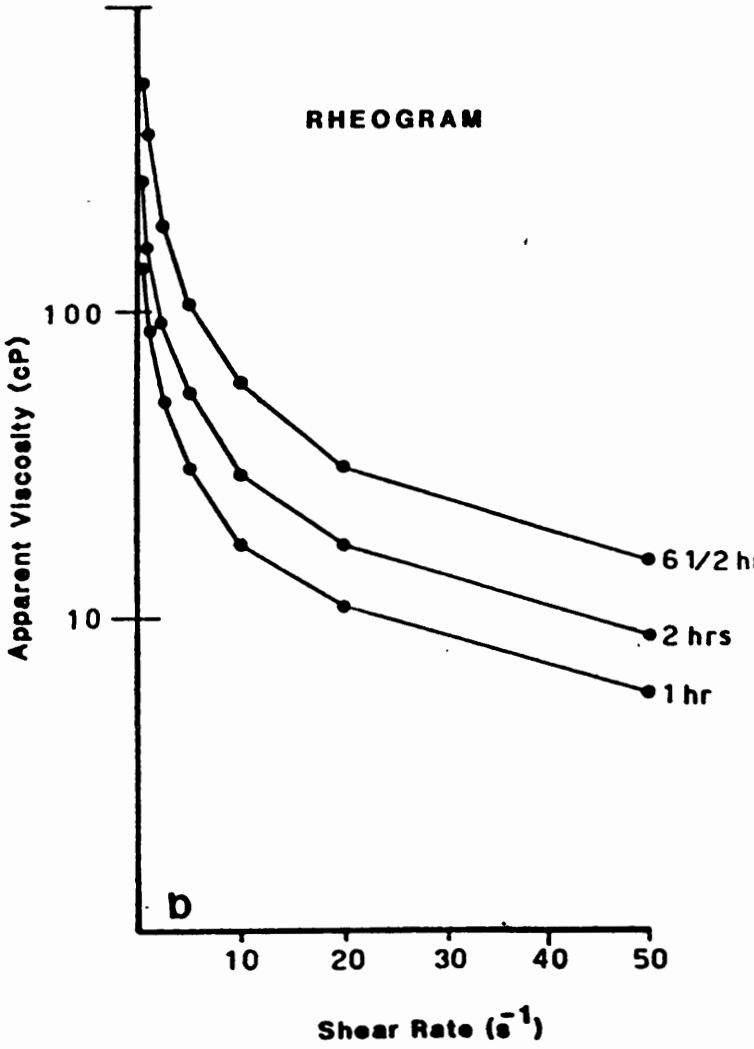
Although fluid mud was recovered at several localities, the amounts were too small to be useful for experimental work and suspensions for analysis were generated directly from the upper 5 cm of a bottom grab sample. In Chesapeake Bay and other similar muddy environments, fine-grained material enters the system from multiple sources, e.g., the watershed, the offshore, the atmosphere, from erosion of the tidal flats and estuary margin, and is constantly being regenerated from the bottom sediments through the activities of infaunal and epifaunal organisms (Rhoades 1963, 1974; Haven and Morales-Alamo 1968; Rhoades and Young 1971) and resuspended by tidal scouring (Schubel 1968, 1971; Nichols and Biggs 1985), and wind-waves (Anderson 1972). Consequently, it seems likely that suspensions generated from the bed will closely resemble the suspensions from which the bed accumulated.

Suspensions analyzed for density and viscosity were prepared from bottom samples by placing an amount of wet sample, equivalent to 50 g dry sediment, in a Waring blendor and adding room temperature water, with salinity adjusted to that existing at the time of sampling. After blending for 10 minutes, the sample was transferred to a 1-liter graduated cylinder, shaken thoroughly to ensure complete dispersion, and allowed to settle. Settling of the sharp interface that formed between the water and the sediment was measured for a two-hour interval. Each 50 g sample had the same initial density (1.031 Mg/m^3) and changes in density were calculated at 10, 30, 60 and 120 minutes as the interface settled. Grain density was assumed to be 2.65 Mg/m^3.

Viscosity analyses were performed on 20 ml subsamples of fluid mud taken from the settling tubes. Analysis was done with the **Brookfield** 8-speed **RVT** rotational viscometer, equipped with the **UL** adaptor for low viscosity fluids. Measurements consist of rotating a cylindrical spindle at eight different speeds in the suspension in a cup, the radius of which is only slightly larger than that of the spindle. The shear stress and shear rate at each speed can be calculated precisely (due to the mathematically correct dimensions of the cup and spindle). Inasmuch as behavior is usually non-Newtonian, viscosity is considered to be "apparent" as it varies with changes in shear rate and shear stress (Van Wazer et al 1963). Yield stress was determined by recording the highest value of shear stress initially achieved at the lowest shear rate. Each sample was analyzed through an accelerating and decelerating shear rate cycle to simulate the shear stresses of a tidal cycle. Shear rates ranged between 0.61 to 122.36 s^{-1}, corresponding to those customarily experienced in estuaries (Bryant et al 1980; Dyer 1986). Data were plotted as flow diagrams (log shear rate vs. log shear stress) and rheograms (shear rate vs. log "apparent" viscosity - Fig. 2a and 2b). After analysis, each sample was transferred to a pre-weighed alumunum moisture can and placed in a 105°C drying oven to determine the percentage of solid material.

STATION 13 - CHESAPEAKE BAY





2. a. Flow diagram for suspension from Station 13. Shear thinning (pseudoplastic) viscous behavior is characteristic as density changed from 1.055 Mg/m^3 to 1.080 Mg/m^3 - 1 to 6 1/2 hours settling.

b. Rheogram showing increase in "apparent" viscosity corresponding to density increases during settling interval.

DISCUSSION OF RESULTS

Hindered Settling and Density Development

Hindered settling is exhibited by flocculated suspensions in concentrations >10 g/l and is characterized by a sharp interface between the overlying clear water and the flocculated material which travels downward as interstitial water is expelled upward through the flocculent structure. The process has been suggested to be the mode of accumulation of fluid mud (Einstein and Krone 1962) and is described in detail by Been and Sills (1981) and Sills and Been (1984).

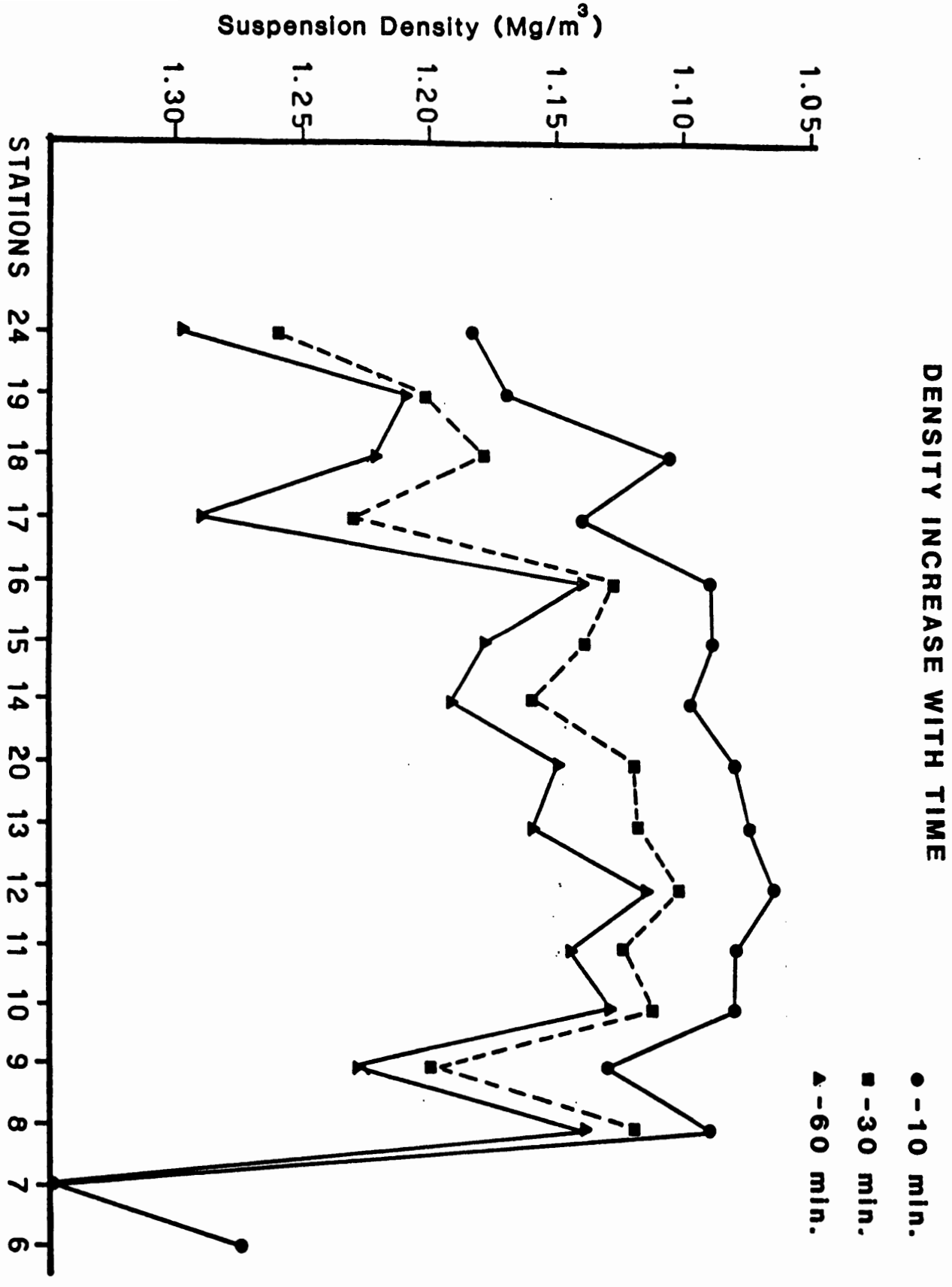
It is believed that hindered settling occurs in Chesapeake Bay during slack water by sediments which flocculate in the low salinity reaches of the upper Bay (Schubel and Kana 1972; Zabawa 1978; Gibbs 1985). It is understood that there will always be some residual circulation during most slack water intervals. However, the very presence of fluid muds indicates that the shear stresses generated by this residual circulation are too weak to keep the fine particles separated and in suspension. Creutzberg and Postma (1979) have demonstrated experiementally that there is a critical depositional velocity below which North Sea silts become deposited. This velocity is 12 cm/s, measured 15 cm above the bottom. It seems likely that a similar relationship exists for the sediments of Chesapeake Bay.

Figure 3 shows the changes in density with settling time of the suspensions from each sample site. Three regions, each demonstrating different modes of hindered settling, are shown. Rapid settling occurs at Station 7 and seaward. Intermediate settling is observed at stations landward of Station 16. Stations 17 and 24 settle to a slurry density of 1.30 Mg/m^3 in about 1-hour, hence fluid mud should not be found abundantly at these sites. Stations 18 and 19, while not achieving a density of 1.30 Mg/m^3, also settle to high density slurries. Slowest settling is seen between Stations 10 to 16, with settling increasing slightly toward Station 16. At Station 14, a density of 1.19 Mg/m^3 was achieved after 1-hour of settling.

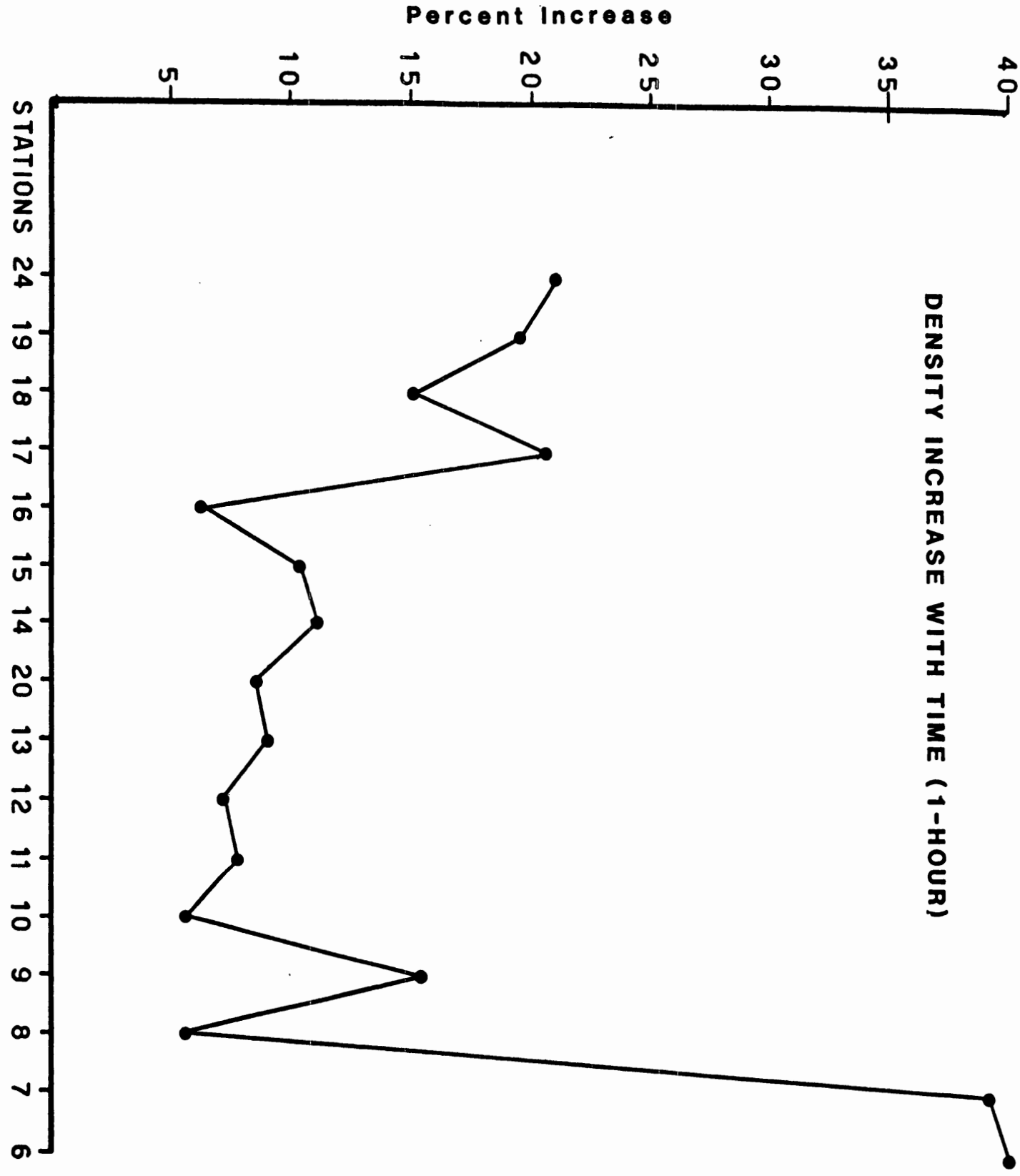
In a different view, Figure 4 shows the increase in density during 1-hour of settling of each suspension sample. Stations 10 through 16 show a maximum density increase of 12% as compared to 20-40% increase further up and down the estuary. It is clear that the least dense, most water-rich sediments will be found between Stations 10 to 16. Fluid mud should occur in significant quantities in this reach and has been documented by Nichols et al (1981).

Viscosity Characteristics

The suspensions exhibit non-Newtonian shear thinning (pseudoplastic) flow behavior, typical of dilute clay/water systems (van Olphen 1963). In physical terms, this means that the "apparent" viscosity of the suspensions decreased as the shear rate (defined as the velocity gradient, i.e., d_v/d_y. between the top and basal surfaces of the



3. Longitudinal profile showing density increase with settling time for all stations. Minimal increase occurs at Stations 10 to 16 with greater increase occurring landward of Station 16 and seaward of Station 10.



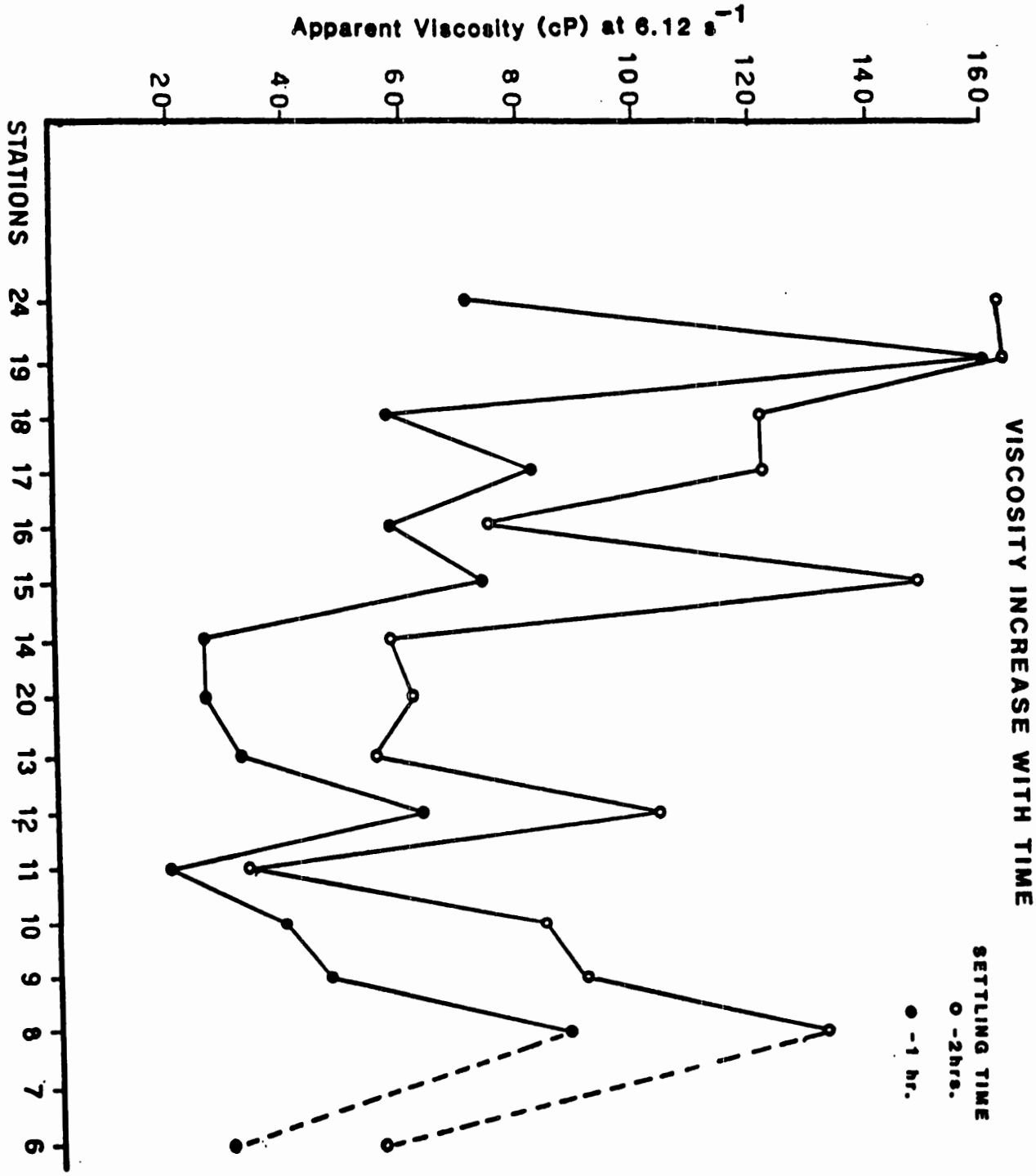
4. Percent increase of suspension density in 1-hour settling time for all stations. Lowest increase is observed to occur between Stations 10 to 16.

layers being sheared) increased. The reverse phenomena, shear thickening (dilatant) flow occurred in samples that had settled long enough to attain a 10% solids concentration (usually 5 hours). Flow diagrams and rheograms are shown in Fig. 2a and 2b for Station 13. The sample was analyzed after 1, 2, and 6.5 hours of settling. During this time, the density of the suspension increased from 1.04 Mg/m^3 (7.0% solids) to 1.054 Mg/m^3 (8.7% solids) to 1.069 Mg/m^3 (11.1% solids). A tendency toward shear thickening was observed at the lowest shear rate in the longest settled sample - however, shear thickening was not common in these suspensions. These behavioral patterns are important sedimentologically inasmuch as shear thinning results in a decrease in "apparent" viscosity which, under high shear rates, would encourage resuspension by reducing the thickness of the laminar layer of the benthic boundary layer. Shear thickening results in an increase in "apparent" viscosity which, if occurring at some critical higher shear rate, would increase the thickness of the laminar layer of the benthic boundary layer and inhibit resuspension (Faas 1985, 1986). A dramatic increase in the thickness of the laminar layer was observed in the drag reduction experiments of Gust (1976) and Gust and Walger (1976) when fine sediment was introduced into the flow field.

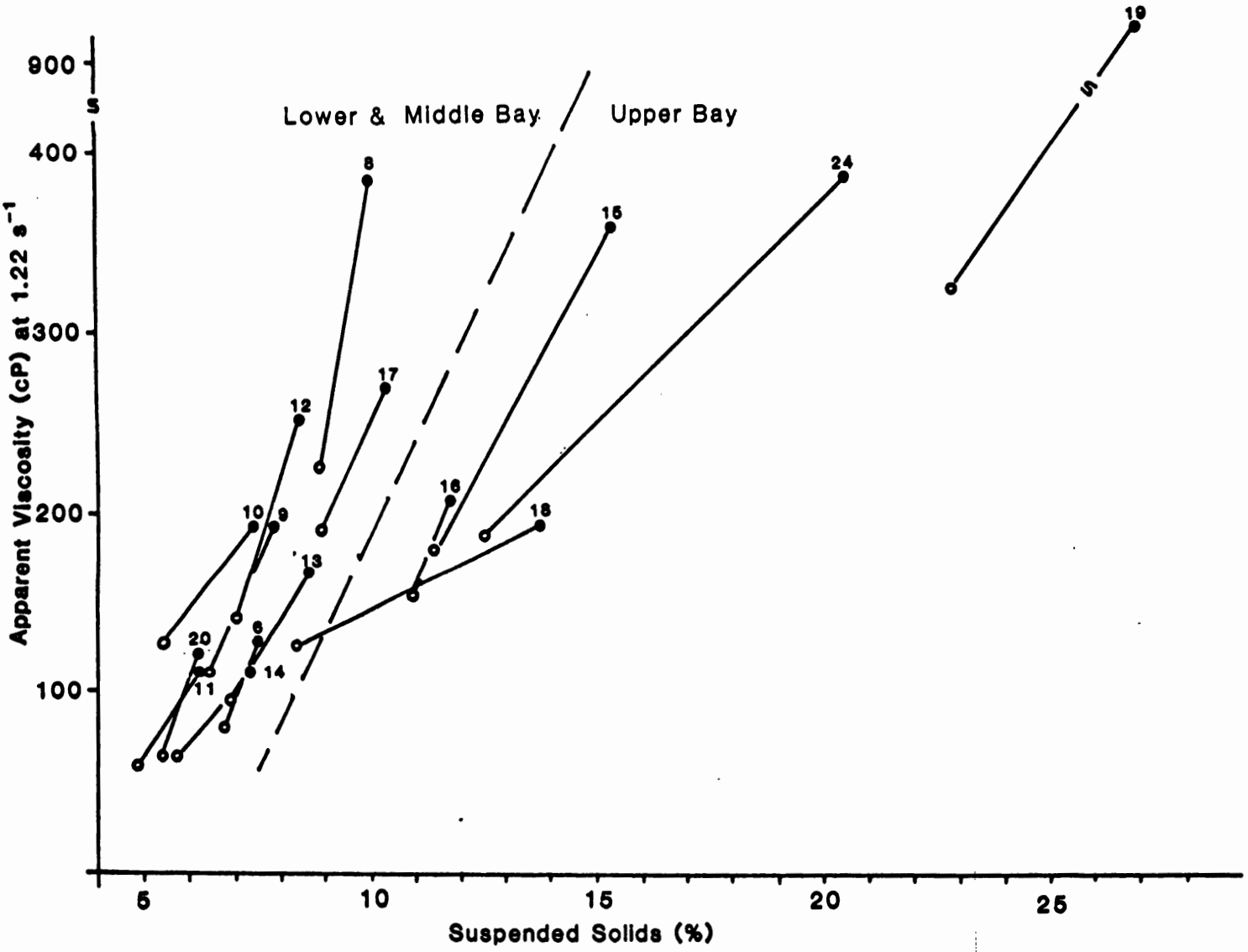
In order to visualize the pattern of viscous change throughout the Bay, "apparent" viscositieswere plotted for 1 and 2-hour settling times for samples from each station (Fig. 5). Suspensions possessing high "apparent" viscosities exist seaward of Station 10 and landward of Station 14. Low "apparent" viscosities occurred in suspensions between Stations 11 and 14, with an anomalous intermediate "apparent" viscosity suspension existing at Station 12.

Figure 6 shows the relationship between suspended solids and "apparent" viscosity for most samples. Each was measured at 1 and 2-hour settling intervals, and shows the associated density increase. Two distinct trends are noted. Samples from the upper Bay (Stations 15, 16, 18, 19, and 24) show a large increase of "apparent" viscosity with suspended solids concentration whereas the increase in middle Bay sediments is much less. Upper Bay samples are less viscous than those from the middle Bay since they require a greater solids concentration (2X) to achieve the same "apparent" viscosity. The distinction is quite abrupt and results from the fact that upper Bay sediments are generally coarser-grained than middle Bay sediments. However, since upper Bay suspensions achieve high densities more rapidly during the same settling interval than those from the middle Bay (Fig. 3), their "apparent" viscosities are also greater. This is best illustrated by the differences in "apparent" viscosity between the two groups after one hour of settling (Fig. 5).

One of the time-dependent properties often found in clay/water systems is yield stress. This phenomena results from an internal strengthening of the suspension and requires that a certain amount of shear stress be applied to the system before flow is initiated. Migniot (1968) found that yield stress increased rapidly in concen-



5. "Apparent" viscosity increase with settling time for all stations. Lowest viscosities are found at Stations 11, 13, 14, and 20. Greatest viscosity increases are found landward of Station 14.

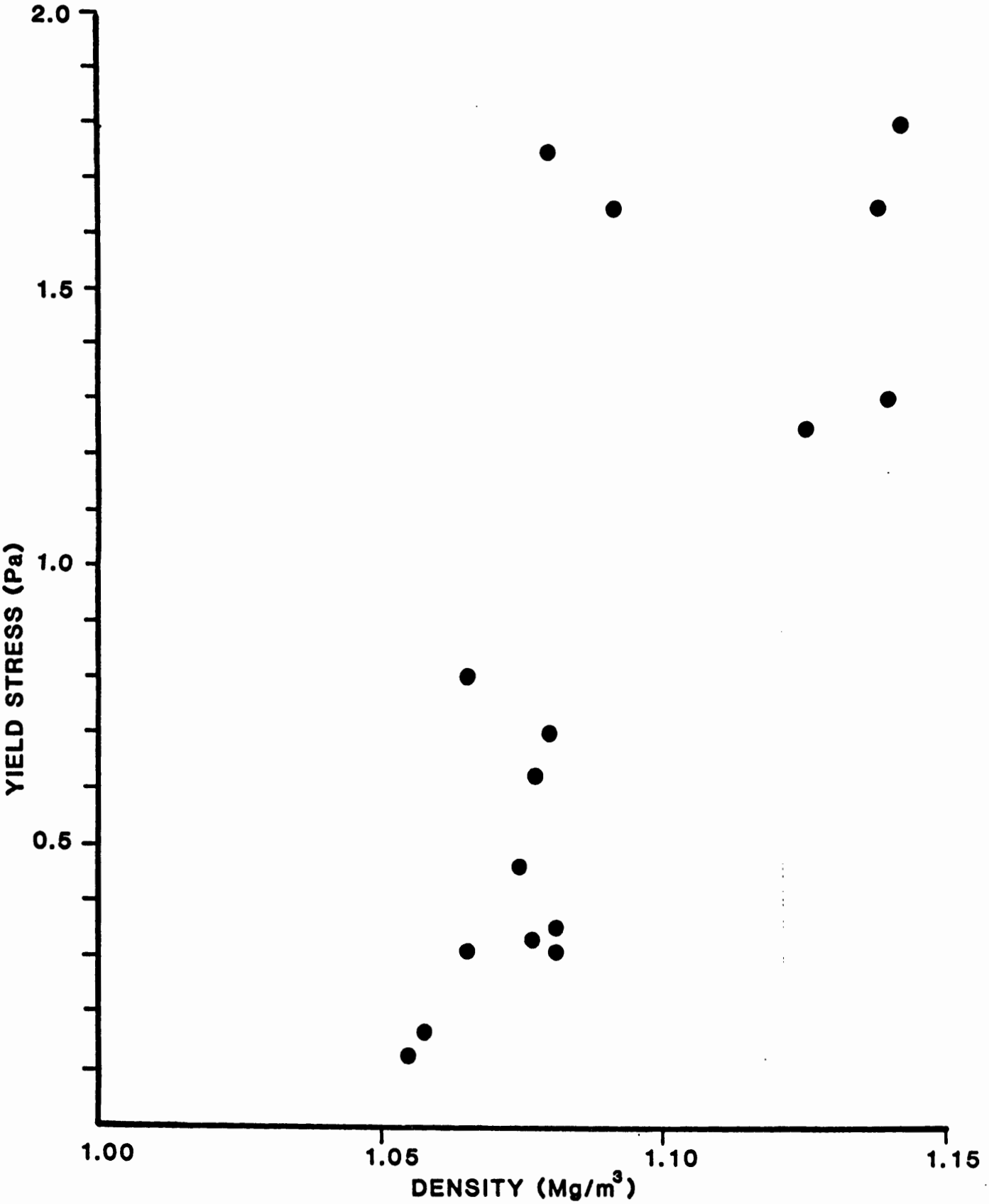


6. "Apparent" viscosity versus percent of suspended solids for all stations. Left side of dashed line shows "apparent" viscosity increase for one and two-hour settled samples from lower and middle Bay stations (6-14), measured at 1.22 s^{-1} shear rate. Low "apparent" viscosity (about 100 cPs) occurs in one hour of settling, generally doubling after two hours. Right side of dashed line shows the same data for upper Bay samples (15-24). "Apparent" viscosity after one hour is at least twice that of lower and middle Bay samples, and also doubles after two hours of settling. Station 17 behaves as an upper Bay sample at lower suspended solids concentrations as does Station 8 from the lower Bay. Station 6 (lower Bay) illustrates middle Bay behavior. Upper Bay suspensions require greater densities to achieve the same "apparent" viscosities as middle Bay suspensions. However, due to rapid settling and density increase, upper Bay suspensions become more viscous than middle Bay suspensions in equivalent settling times.

trated mud suspensions (>200 g/l), as did Malherbe et al (1986) in the muds from the harbor of Zeebrugge (Belgium). However, Bryant et al (1980) observed little indication of yield stress development in their study of the fluid mud of the Severn Estuary and Bristol Channel. Figure 7 shows that yield stresses as great as 2 Pa (1 Pa = 10 dynes/cm^2) may develop in Chesapeake Bay suspensions at densities to 1.15 Mg/m^3. This stress (2 Pa) corresponds to a critical shear velocity (U_*) of 2.5 cm/s (Migniot 1968). Calculations, following Terwindt and Breusers (1972, 1982) indicate this U_* can be achieved by a current of 67 cm/s, measured at a distance of 1 m above the fluid mud interface. Development of a greater yield stress in 60 minutes of hindered settling is unlikely to occur at Stations 8, and 10 to 16 (possibly excluding 14 and 15) inasmuch as densities through this reach increase very slowly. Pritchard (1971) indicates that the net non-tidal current (up-estuary in the lower layer) is one-fifth the magnitude of tidal currents (25 to 100 cm/s). Mean flood tidal velocity in the lower layer is about 40 cm/s and the mean net non-tidal bottom flow is approximately 10 cm/s up-estuary. Resuspension of these sediments could occur at current velocities in excess of 67 cm/s, measured 1 m above the fluid mud interface; however, since these stations are all in water depths greater than 12 m, such velocities seem unlikely.

Stations 17 to 24 reach suspension densities between 1.20 - 1.30 Mg/m^3 within 60 minutes and presumably develop correspondingly greater yield stress. However, they lie at depths less than 10 m and are likely affected by strong tidal currents. Schubel (1969) measured flood tidal velocities of ~75 cm/s and ebb tidal velocities of ~55 cm/s at 50 cm above the bottom in the upper Bay (corresponding to Station 18) with concentrations of suspended sediment as high as 279 mg/l.

Nichols (1986) relates resuspension to a simple relationship between energy capacity "e" and sediment concentration "c" and indicates resuspension will occur whenever "e" exceeds "c". This concept fails to take into account sediment responses associated with density-dependent phenomena such as shear thinning and shear thickening behavior and time-dependent phenomena such as yield stress. In addition, the geochemical setting may also be a determining factor. In oxidizing environments, when the sediment concentration approaches 200 g/l, shear thickening usually occurs at some critical shear rate and shear stress and resuspension may be restricted (Faas 1985, 1986). However, in anoxic environments, the rheological behavior of the sediments is quite different. Studies of acoustic properties of gassy sediments generally indicate a loss of a rigid framework resulting in absorption and attenuation of the acoustic energy (Wood and Weston 1964; Schubel and Schiemer 1973; Anderson 1974). Figure 5 indicates that low densities and low apparent viscosities occur in suspensions throughout this reach. In these sediments, shear thinning dominates, yield stress is minimal, and a greater potential for resuspension is predicted.



7. Yield Stress increases with density during two-hour settling interval of middle Bay samples (Stations 10 to 16).

CONCLUSIONS

Chesapeake Bay can be rather sharply divided into three regions of varying resuspension potential (Fig. 1). Lowest potential extends from the mouth of the Bay northward to Station 9 off the Patuxent River. This low potential is due primarily to high "apparent" viscosity and rapid density development due to lesser amounts of clay-sized material in bottom-generated suspensions. Greatest potential extends from Station 10 to Station 14 off the mouth of the Patapsco River in the middle Bay. Suspensions from this reach show the lowest "apparent" viscosity and lowest yield stress development of any suspensions within the Bay. Settling rate and density increase with time are very low which contributes to the low "apparent" viscosity. Intermediate resuspension potential exists in suspensions from Station 14 northward to Station 24 at the mouth of the Susquehanna. "Apparent" viscosity increases sharply, reaching its maximum at Station 19, then decreases slightly. The percent of density increase during one-hour of hindered settling is lowest at Station 16, then increases to about 20% for the remaining stations. The middle reach of Chesapeake Bay (Stations 10 to 14) contain dense suspensions of fluid mud (Nichols et al 1981). This reach experiences severe anoxia for several months each year (Officer et al 1984; Seliger et al 1985). Such conditions lead to a weakening of sediment bonds (Faas and Wartel 1977; Wartel et al 1985) and may be a factor contributing to the development of fluid mud in this reach.

Since this work dealt specifically with a laboratory characterization of the physical properties of suspensions generated from Chesapeake Bay bottom sediment, actual in situ resuspension behavior remains untested. The fact that laboratory experiments reveal distinct patterns of sediment behavior and a range of parametric variability which corresponds with independent field observations, e.g., distribution of fluid mud, sediment textural variation, and current circulation patterns, appears to substantiate the interpretations expressed and to provide an additional framework within which to analyze past and future data. However, it remains to be determined at which of the sampling stations nearbottom fluid mud suspensions actually exist, how much area they occupy, their persistence throughout the year, and their rheological behavior through a lunar tidal cycle.

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REFERENCES

ALLERSMA, E. 1982. Mud in estuaries and along coasts. Delft Hyd. Lab. Pub. 270: 663-685.

ANDERSON, A. L. 1974. Acoustics of gas-bearing sediments. App. Res. Lab. Rept. ARL-TR-74-18. The University of Texas at Austin, 162p.

ANDERSON, F. E. 1972. Resuspension of estuarine sediments by small amplitude waves. Jour. Sed. Pet. 42: 602-607.

BEEN, K., AND G. C. SILLS. 1981. Self weight consolidation of soft soils. Geotechnique 31: 519-535.

BIGGS, R. B., AND B. A. HOWELL. 1984. The estuary as a sediment trap: alternate approaches to estimating its filtering efficiency. In: Kennedy VS (ed) The estuary as a filter, Academic Press, New York, pp 107-129.

BOUYOUCOS, G. J. 1962. Hydrometer method improved for making particle size analysis of soils. Agron. Jour. 54: 464-465.

BRYANT, R., A. E. JAMES, AND D. J. A. WILLIAMS. 1980. Rheology of cohesive suspensions. In: Collins MB, Banner FT, Tyler PA, Wakefield SJ, James, AE (eds) Industrial embayments and their environmental problems. A case study of Swansea Bay. Pergamon Press, New York, pp 279-287.

CREUTZBERG, F., AND H. POSTMA. 1979. An experimental approach to the distribution of mud in the southern North Sea. Neth. Jour. Sea. Res. 13: 99-116.

DAVIES, B. E. 1974. Loss-on-ignition as an estimate of soil organic matter. Soil Sci. Amer. Proc. 38: 150.

DYER, K. R. 1986. Coastal and estuarine sediment dynamics. John Wiley and Son, Ltd. 342p.

EINSTEIN, H. A., AND R. B. KRONE. 1962. Experiments to determine modes of cohesive sediment transport in salt water. Jour. Geophys. Res. 67: 1451-1461.

FAAS, R. W. 1981. Rheological characteristics of Rappahannock estuary muds, southeastern Virginia, U.S.A. Spec. Publ. Int. Assoc. Sedimentol: 505-515.

_____. 1985. Time and density-dependent properties of fluid mud suspensions, NE Brazilian Continental Shelf. Geo-Mar. Lett. 4: 147-152

_____. 1986. Mass-physical and geotechnical properties of surficial sediments and dense nearbed sediment suspensions on the Amazon continental shelf. Cont. Shelf Res. 188: 189-208.

FAAS, R. W., AND S. WARTEL. 1977. The effect of gas bubble formation on the physical and engineering properties of recently deposited fine-grained sediments. Geol. en Mijnb. 56: 211-218.

GIBBS, R. J. 1982. The differentiation of toxic metals in estuaries. Thal. Jugo. 16: 181-190.

_____. 1983. Coagulation rates of clay minerals and natural sediment. Jour. Sed. Pet. 53: 1193-1203.

_____. 1985. Estuarine flocs: their size, settling velocity and density. Jour. Geophys. Res. 90, C2: 3249-3251.

_____. 1986. Segregation of metals by coagulation in estuaries. Mar. Chem. 18: 149-159.

GROSS, M. G., M. KORNWEIT, W. B. CRONIN, AND J. R. SCHUBEL. 1978. Suspended sediment discharge of the Susquehanna River to northern Chesapeake Bay, 1966 to 1976. Estuaries 1: 106-110.

GUST, G. 1976. Observations on turbulent-drag reduction in a dilute suspension of clay in sea-water. Jour. Fluid Mech. 75: 29-47.

GUST, G., AND E. WALGER. 1976. The influence of suspended cohesive sediments on boundary-layer structure and erosive activity of turbulent seawater flow. Mar. Geol. 22: 189-206.

HARRIS, R., M. NICHOLS, G. THOMPSON, J. BONACKI, AND G. VADAS. 1980. Heavy metal inventory of suspended sediment and fluid mud in Chesapeake Bay. Va. Inst. Mar. Sci. Spec. Sci. Rpt. 99, 133 p

HAVEN, D. S., AND R. MORALES-ALAMO. 1968. Occurrence and transport of faecal pellets in suspension in a tidal estuary. Sed. Geol. 2: 141-151.

HAWLEY, N. 1981. Flume experiments on the origin of flaser bedding. Sedimentology 28: 699-712.

INGLISS, C. C., AND F. H. ALLEN. 1957. The regimen of the Thames estuary as affected by currents, salinities, and river flow. Pro. Inst. Civ. Eng. 7: 827-868.

KADDAH, M. T. 1974. The hydrometer method for detailed particle-size analysis. 1. Graphical interpretation of hydrometer readings and test of method. Soil Sci. 118: 102-108.

KIRBY, R., AND W. R. PARKER. 1977. The physical characteristics and environmental significance of fine-grained sediment suspensions in estuaries. Estuaries, Geophysics and the Environment, Nat. Acad. Sci., Washington, D.C. pp 110-120.

_____. 1983. Distribution and behavior of fine sediment in the Severn Estuary and Inner Bristol Channel, U. K. Can. Jour. Fish. Aquatic Sci. 40: 83-95.

KRANCK, K. 1975. Sediment deposition from flocculated suspensions. Sedimentology 22: 111-123.

_____. 1981. Particulate matter, grain size characteristics, and flocculation in a partially-mixed estuary. Sedimentology 28: 107-114.

_____. 1986. Settling behavior of cohesive sediment. In: Mehta AJ (ed) Estuarine cohesive sediment dynamics. Springer-Verlag, New York, pp 151-169.

KRONE, R. B. 1963. A study of rheologic properties of estuarial sediments. Tech Bull 7, Comm on Tidal Hydr, U.S. Army Corps of Engr, WES, Vicksburg, MI.

MALHERBE, B., P. DEWOLF, AND B. PAQUOT. 1986. Nautical bottom research and survey for optimization of maintenance dredging in mud areas. Symp. on advances in underwater technology, ocean science and offshore engineering, 6, Brighton, U.K., pp 427-441.

MEADE, R. H. 1982. Sources, sinks, and storage of river sediments in the Atlantic drainage of the United States. Jour. Geol. 90: 235-252.

MICHAELS, A. W., AND J. C. BOLGER. 1962. Settling rates and sediment volumes of flocculated kaolin suspensions. Ind. & Engng. Chem. Fund. 1: 24-33.

MIGNIOT, C. 1968. A study of the physical properties of various forms of very fine sediments and their behaviour under hydrodynamic action. La Houille Blanche 7: 591-620.

NEDECO. (1965). A study of the siltation of the Bangkok Port Channel, Vol 2, The field investigation. Neth. Engrng. Consult., The Hague, 474 p.

_____. 1968. Surinam transportation study. Neth. Engrng. Consult., The Hague, 293 p.

NICHOLS, M. 1985. Fluid mud accumulation processes in an estuary. Geo-Mar. Lett. 4: 192-198.

_____. 1986. Effects of fine-sediment resuspension in estuaries. In: Mehta AJ (ed) Estuarine cohesive sediment dynamics. Springer-Verlag, New York, pp 5-42

NICHOLS, M., AND R. B. BIGGS. 1985. Estuaries. In: Davis R (ed) Coastal sedimentary environments, Springer-Verlag, New York, pp 77-186.

NICHOLS, M., G. S. THOMPSON, AND R. W. FAAS. 1978. A field study of fluid mud dredged material: its physical nature and dispersal. U. S. Army Corps of Engineers, WES Tech. Rept. D-78-40, 74 p.

NICHOLS, M., R. HARRIS, G. S. THOMPSON, AND B. W. NELSON. 1982. Metal inventory and fate of suspended sediment in Chesapeake Bay. Va. Inst. Mar. Sci. Spec. Sci. Rept. 106, 97 p.

OFFICER, C. B. 1981. Physical dynamics of estuarine suspended sediments. Mar. Geol. 40: 1-14.

OFFICER, C. B., D. R. LYNCH, G. M. SETLOCK, AND G. R. HELTZ. 1984 Recent sedimentation rates in Chesapeake Bay. In: Kennedy VS (ed) The estuary as a filter, Academic Press, New York, pp 131-157.

OFFICER, C. B., R. B. BIGGS, J. L. TAFT, L. E. CRONIN, M. A. TYLER, AND W. R. BOYNTON. 1984. Chesapeake Bay anoxia: origin, development, and significance. Science 223: 22-27.

OLPHEN, H., VAN. 1963. Introduction to clay-colloid chemistry. Interscience Publ., New York, 318 p.

OWEN, M. W. 1970. Properties of a consolidating mud. Hyd. Res. Sta. Rept. INT 83, 90 p.

PARKER, W. R., AND R. KIRBY. 1982. Time dependent properties of cohesive sediment relevant to sedimentation management - European experience. In: Kennedy VS (ed) Estuarine comparisons, Academic Press, New York, pp 573-589.

PAZWASH, H., AND J. M. ROBERTSON. 1971. Fluid-dynamic consideration of bottom materials. Jour. Hyd. Div. ASCE HY 9: 1317-1329.

PRITCHARD, D. W. 1971. Chemical and physical oceanography of Chesapeake Bay. The estuarine environment. Estuaries and estuarine sedimentation. AGI short course lecture notes, IVA -1 to 28.

RHOADES, D. C. 1963. Rates of sediment reworking by Yoldia limatula in Buzzards Bay, Massachusetts, and Long Island Sound. Jour. Sed. Pet. 33: 723-727.

_____. 1974. Organism-sediment relations on the muddy sea floor. Oceanogr. Mar. Biol. Ann. Rev. 12: 263-300.

RHOADES, D. C., AND D. K. YOUNG. 1971. Animal-sediment relations in Cape Cod, Massachusetts II. Reworking by Molpadia oolitica (Holothuroidea). Mar. Biol. 11: 255-261.

SCHUBEL, J. R. 1968. Suspended sediment of the Northern Chesapeake Bay. The Johns Hopkins Univ. Chesapeake Bay Inst. Tech. Rept. 35. Ref. 68-2, 264 p.

_____. 1969. Size distributions of the suspended particles of the Chesapeake Bay turbidity maximum. Neth. Jour. Sea. Res. 4: 283-309.

_____. 1971. Tidal variation of the size distribution of suspended sediment at a station in the Chesapeake Bay turbidity maximum. Neth. Jour. Sea. Res. 5: 252-266.

_____. 1982. An eclectic look at fine particles in the coastal ocean. In: Kimrey K, R. Burns R (eds) Proc. of a pollutant transfer by particulates workshop. Seattle, WA. pp 53-142.

SCHUBEL, J. R., AND T. W. KANA. 1972. Agglomeration of fine-grained suspended sediment in northern Chesapeake Bay. Powder Tech. 6: 9-16.

SCHUBEL, J. R., AND E. W. SCHIEMER. 1973. The cause of acoustically impenetrable or turbid character of Chesapeake Bay sediments. Mar. Geophys. Res. 2: 61-71.

SCHUBEL, J. R., AND W. H. CARTER. 1984. The estuary as a filter for fine-grained sediment. In: Kennedy VS (ed) The estuary as a filter, Academic Press, New York. pp 81-105.

SELIGER, H. H., J. A. BOGGS, AND W. H. BIGGLEY. 1985. Catastrophic anoxia in the Chesapeake Bay in 1984. Science 228: 70-73.

SILLS, G. C., AND K. BEEN. 1984. Escape of pore fluid from consolidating sediment. In: Parker WR, Kinsman DJJ (eds) Transfer processes in cohesive sediment systems, Plenum Press, New York. pp 109-125.

TERWINDT, J. H. J., AND H. N. C. BREUSERS. 1972. Experiments on the origin of flaser, lenticular and sand-clay alternating bedding. Sedimentology 19: 85-98.

_____. 1982. Flume experiments on the origin of flaser bedding. Sedimentology 29: 903-907.

VAN WAZER, J. R., J. W. LYONS, K. Y. KIM AND R. W. COLWELL. 1963. Viscosity and flow measurements. A laboratory handbook of rheology. Interscience Publ. New York, 301 p.

WARTEL, S. 1973. Variations in concentration of suspended matter in the Scheldt estuary. Bull. Inst. r. Sci. nat. Belg. 49: 1-11.

WARTEL, S., G. VERREET, AND W. VIAENE. 1985. Internal structures in mud sediments. (The occurrence of gas and its implications). In: Van Grieken R, Wollast R. (eds) Progress in Belgian oceanographic research, Proc. of Symp, Brussels. The University of Antwerp Press, pp 220-22.

WELLS, J. T. 1978. Shallow-water waves and fluid mud dynamics, coast of Surinam, South America. Coastal Studies Inst. Tech. Rept. 257, Louisiana State Univ, Baton Rouge, LA. 56 p.

WELLS, J. T., AND J. M. COLEMAN. 1981. Physical processes and fine-grained sediment dynamics, coast of Surinam, South America. Jour. Sed. Pet. 51: 1053-1068.

WOOD, A. B., AND D. E. WESTON. 1964. The propagation of sound in mud. Acustica 14: 156-162.

ZABAWA, C. F. 1978. Microstructure of agglomerated suspended sediments in Northern Chesapeake Bay Estuary. Science 202: 49-51.

Understanding the Estuary: Advances in Chesapeake Bay Research. Proceedings of a Conference. 29-31 March 1988. Baltimore, Maryland. Chesapeake Research Consortium Publication 129. CBP/TRS 24/88.

Comparison of Sediment Landscapes in Chesapeake Bay as Seen by Surface and Profile Imaging

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INTRODUCTION

The sediment-water interface is the boundary layer between the water column and sediments. It is involved in virtually all processes and cycles within aquatic and estuarine ecosystems. Interactions and reactions at the sediment-water interface are of particular importance in regulating processes involving nutrient regeneration-remineralization (Boynton and Kemp 1985), fate of toxicants (Olsen, Cutshall and Larsen 1982), development of hypoxia-anoxia (Garber 1987), sediment mixing (Schaffner et al. 1987a, b), and sediment transport (Wright et al. 1987). Much effort has and is being expended to provide details of these processes which will eventually be used in management plans for water quality, sediment quality, and fisheries resources.

Generally, field methods for investigating sediment-water interface processes or fluxes are time and labor intensive. Complementary methods are needed to support detailed studies and allow for better comprehension of these dynamic processes. Rhoads and Cande (1971) proposed the use of sediment profile cameras as a means of quickly collecting data on the character of the sediment-water interface. Rhoads and Germano (1986) outlined a scheme using sediment profile cameras to assess the character of the sediment-water interface relative to benthic community succession. Day, Schaffner, and Diaz (in press), in addition to using a sediment profile camera, also advocated the use of bottom surface cameras in conjunction with the profile camera to provide a more complete evaluation of the sediment-water interface.

Sediment profile and bottom surface cameras provide a unique in situ view of the sediment-water interface yielding both qualitative and quantitative data on its biological, chemical, and physical character. This in situ photographic approach and subsequent image analysis can quickly and cost effectively cover large areas of bottom defining biological, sediment fabric, and energy gradients or other spatial patterns. Natural or anthroprogenic events (i.e. storms, high flows, dredged material disposal) through time can also be easily followed and recovery rates measured.

In this paper we will demonstrate the utility of using a surface and profile imaging camera system to provide a broad characterization of the sediment-water interface from selected tributaries and mainstem of the Chesapeake Bay. Emphasis will be placed on defining the redox potential discontinuity and its depth in the sediment relative to biological and geochemical factors.

METHODS AND MATERIALS

A modified Benthos model 3731 sediment profile camera and Benthos model 371 standard camera and 372 standard flash were combined into a photographic system for evaluating sediment quality and benthic habitat complexity. The sediment profile camera provides images of the sediment column 15 cm wide and up to 20 cm deep. The profile camera does not provide comprehensive resolution of surface features, particularly if the prism penetration exceeds the optical axis of the camera lens. The standard camera is used to provide information on the surface by photographing an area approximately 20 x 30 cm in front of the profile camera. In combination this Surface and Profile Imaging (SPI) camera system provides a high resolution quick look into the character of the sediment water interface. The configuration of cameras in the SPI system can be seen in Figure 1.

Data from 359 SPI images collected in the Patuxent River, York River, and Lower Chesapeake Bay (Fig. 2) between April 1986 and February 1988 were used in this evaluation of sediment landscapes. Each image was analyzed using an International Imaging Systems I25 image processor interfaced to a Prime 9955 computer. Of the 14 major parameters measured from each image (Table 1) surface relief, depth of apparent RPD, void area, and sediment grain size were selected for evaluation.

Surface relief is maximum point of prism penetration minus the minimum point across the 15 cm width of the prism face plate. Apparent RPD depth is the area of the image visually discerned as being aerobic divided by the width of the analyzed image. We use the term apparent in describing this parameter because no actual measure is made of the redox potential. An assumption is made that, given the complexities of iron and sulfate reduction-oxidation chemistry, the reddish-brown color tones in sediments are indications of sediments that if not aerobic are not intensely reducing. This is in accordance with the classical concept of RPD depth which associates it with sediment color (Fenchel 1969). The area of an image occupied by voids and the type of voids are good indications of subsurface biological and physical processes. Void area is expressed as a percent of the total analyzed image area. All images are then standardized to a constant 15 cm prism penetration to avoid over or under weighting images that were less than or greater than 15 cm. Sediment grain size was estimated by comparing each image to sediments of known grain size. Sediment types followed the Wentworth classification as described in Folk (1974) and represent modal class for each image.

The entire data set was stratified a posteriori by sediment type (as described above), salinity at each location (from Stroup and Lynn 1963), and depth (recorded at time of collection) (Table 2). Broadscale patterns and trends were then evaluated using SPSSX (SPSS 1986).

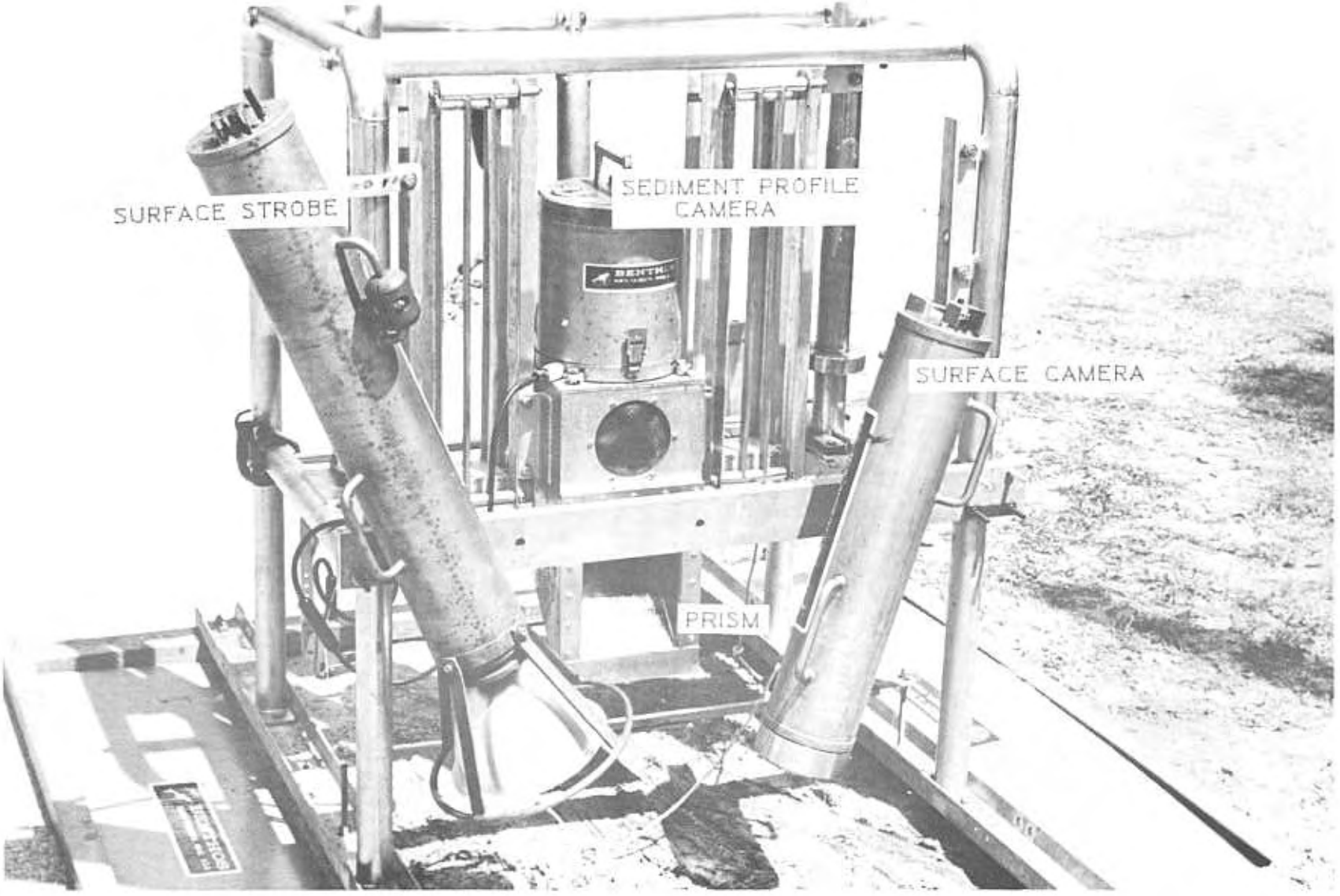


Figure 1. Surface and Profile Imaging (SPI) camera system.

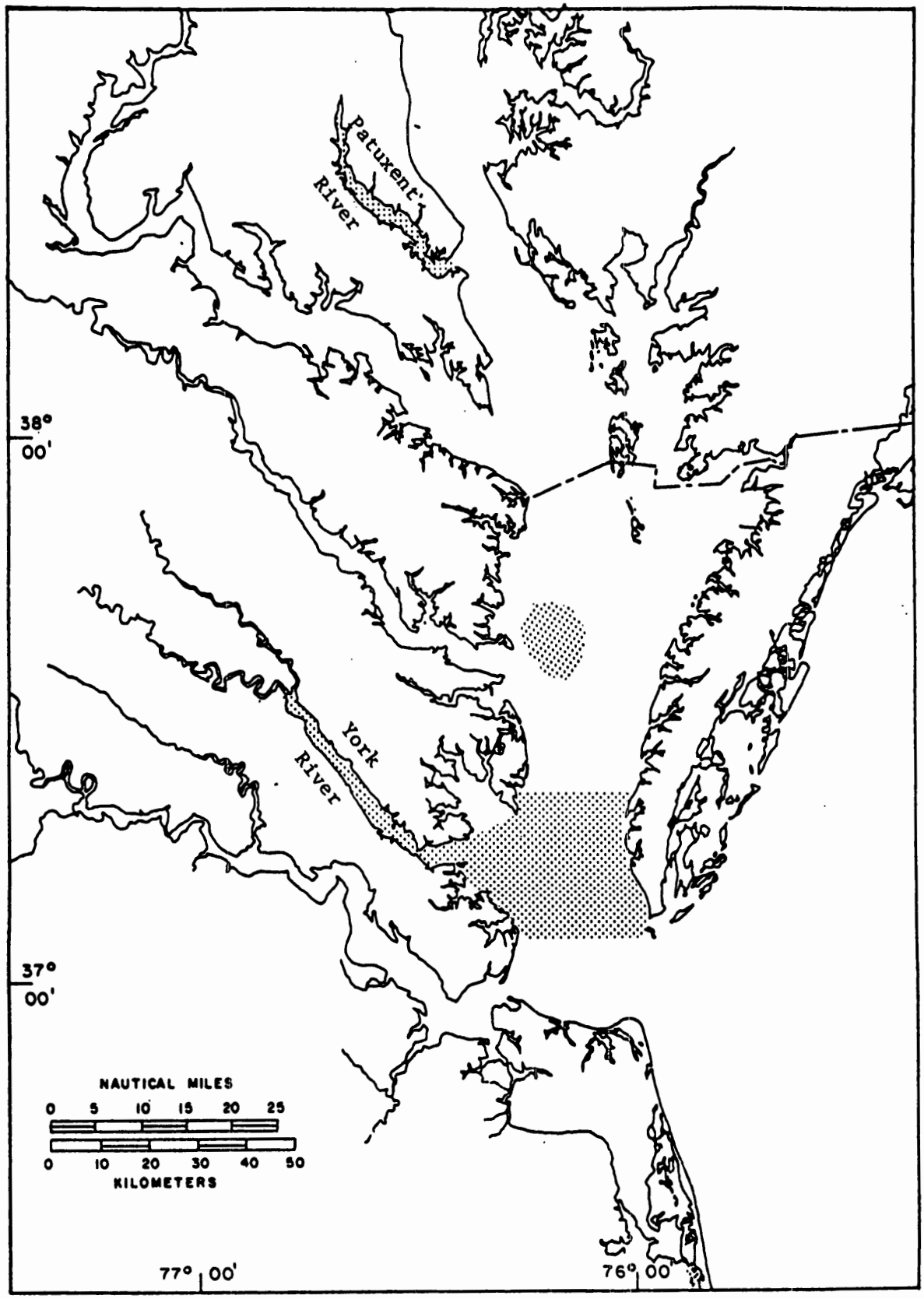


Figure 2. Location of areas around the Chesapeake Bay from which SPI data were collected.

Table 1. Image analysis measurements from sediment profile camera photographs.

Measurement	Method	Usefulness
a - Depth of Penetration	Average of maximum and minimum distance from sediment surface to bottom of prism window.	Penetration depth is a good indicator of sediment compaction.
b - Surface Relief	Maximum minus nimimum depth of penetration.	If the camera is level, this is a good measure of small scale bed roughness, on the order of 15mm (prism window width).
c - Digitized Image Statistics 1. Pixel densities for total image 2. Pixel densities for areas of interest	Actual range of densities the digitizing camera detects from the sediment profile image.	For cross comparisons of images, it is necessary to have measurements relying upon image pixel density done on a similar intensity range.
d - Depth of apparent RPD Layer	Area of apparently oxic layer (g) divided by width image. Maximum and minimum distance from sediment surface to top of RPD layer are also measured.	Gives a good indication of DO conditions in the bottom waters and the degree of biogenic activity in muddy sediments. In sands will be related to porosity and turbulence.
e - Color Contrast of apparent RPD	Contrast between oxic and anoxic layers is determined from light intensity level density slicing of digitized and specially enhanced image.	Establishes boundary of RPD. Depending upon whether the RPD is straight or convoluted will be of use in understanding the biologic and physical process.
f - Area of Anoxic Sediment	Select desired pixel density for boundary between oxic and anoxic, count anoxic pixels, and convert to area.	When calculated to a constant depth of penetration and combined with oxic layer area a good understanding of RPD dynamics can be obtained.
g - Area of Oxic Sediment	As in f, except use oxic pixel count.	When calculated to a constant depth of penetration and combined with anoxic layer area a good understanding of RPD dynamics can be obtained.
h - Voids	Number counted, depth from surface of each measured, area of each delineated.	Presence of oxic voids is a good indicator deep living fauna and high biogenic activity.
i - Other Inclusions (Methane Bubbles, Mud Clasts, Shells)	Number counted, depth from surface of each measured, area delineated.	Often other inclusions such as methane or mud clasts are indicative of certain processes and are helpful in understanding recent events.
j - Burrows	Number counted, area delineated.	Burrow presence is a good indication of deep living fauna and high biogenic activity.
k - Surface Features 1. Tubes 2. Epifauna 3. Pelletized Layer 4. Shell 5. Mud Clasts	 Counted and speciated. Counted and speciated. Thickness and area delineated. Qualitative estimate of coverage. Qualitative estimate of coverage.	Presence of these features is indicative of recent biological and physical processes.
l - Sediment Grain Size	Determined from comparison of image to images of known grain size.	Provides modal estimate of grain size and sediment layering.
m - Dredged Material or other Layers	Measure thickness above original sediment surface and area delineated.	Location of dredged material and measuring its thickness provide quantitative measure for relating impacts to the benthos of any disposal project.

Table 2. A. posteriori strata definition by sediment type, salinity, and depth.

Sediment strata (Wentworth Size Classes)

Clayey Mud
Silty Mud
Silt
Silty Sand
Fine Sand
Fine-Medium Sand
Medium Sand

Salinity range (ppt)

0 to 5
5 to 15
15 to 20
20 to 25
>25

Depth interval (feet)

<15
15-30
30-45
45-60
>60

RESULTS

The a posteriori stratification of image data by sediment type, salinity, and depth showed that most of the variation in surface relief, apparent RPD depth, and percentage of void area could be explained by sediment type alone. For example, the pattern of the apparent RPD depth was similar with regard to sediment type by salinity range (Fig. 3). Therefore, the data were restratified and reanalyzed by only sediment type.

Surface relief

Surface relief tended to increase with increasing grain size (Fig. 4). From clayey mud to silty sand the increase in surface relief was due to biogenic activities of the benthic fauna. In sands the surface relief was due to current generated bed forms. The magnitude of surface relief in fine sediments averaged 0.7 cm in clayey mud to 1.1 cm in silty-sand. This corresponds to surface slopes of 2.7^{o} and 4.2^{o}, respectively. Bed forms in sands averaged 1.4 to 1.7 cm in height, or 5.3^{o} to 6.5^{o} in slope.

Apparent RPD depth

The depth of the apparent RPD, as measured by brown and reddish-brown color tones of the sediment, tended to increase with increasing grain size (Fig. 5). The higher mean value for RPD in clayey mud over silty mud was due to several highly reworked low salinity stations. Median values for the apparent RPD were the same for both of these sediment types (0.5 cm). The increase in RPD depth in silt and silty sand was due to biogenic reworking of sediments by infauna. In sand sediments porosity was the major determinant of RPD depth.

The thin apparent RPD depths in clayey and silty mud sediments were clearly defined from the grey color tones of the subsurface sediments. Apparent RPD layers less than 1 cm thick in muddy sediments, while not smooth, were more uniform than deeper RPD layers. The complexity in the form of the RPD was highest in silt and silty sand sediments from biogenic activities of infauna. In sands the apparent RPD was simplest in form being close to a uniform surface between aerobic and anaerobic sediments.

Percent void area

The average and median percentage of void area, standardized to 15 cm of prism penetration, was low. Void area in fine and predominantly fine grained sediments averaged 1.3 to 2.1% with median values being much less at 0.0 to 0.8% for the same sediments (Fig. 6). In sands voids were not major subsurface features. At times voids do occur in sands, but they tend to be small. In fine sediments about 15% of the images have voids that were much larger than average, being up to 22% of the sediment area. The majority of these large voids appeared to be active biogenic structures from subsurface deposit feeding. Except in clayey muds many of the largest voids resulted from physical cracking of the sediment caused by the camera prism.

DISCUSSION

Sediment landscapes in the Chesapeake Bay exhibit broadscale patterns related mainly to sediment grain size and secondarily to salinity, which are a primary determinant of the character of infaunal

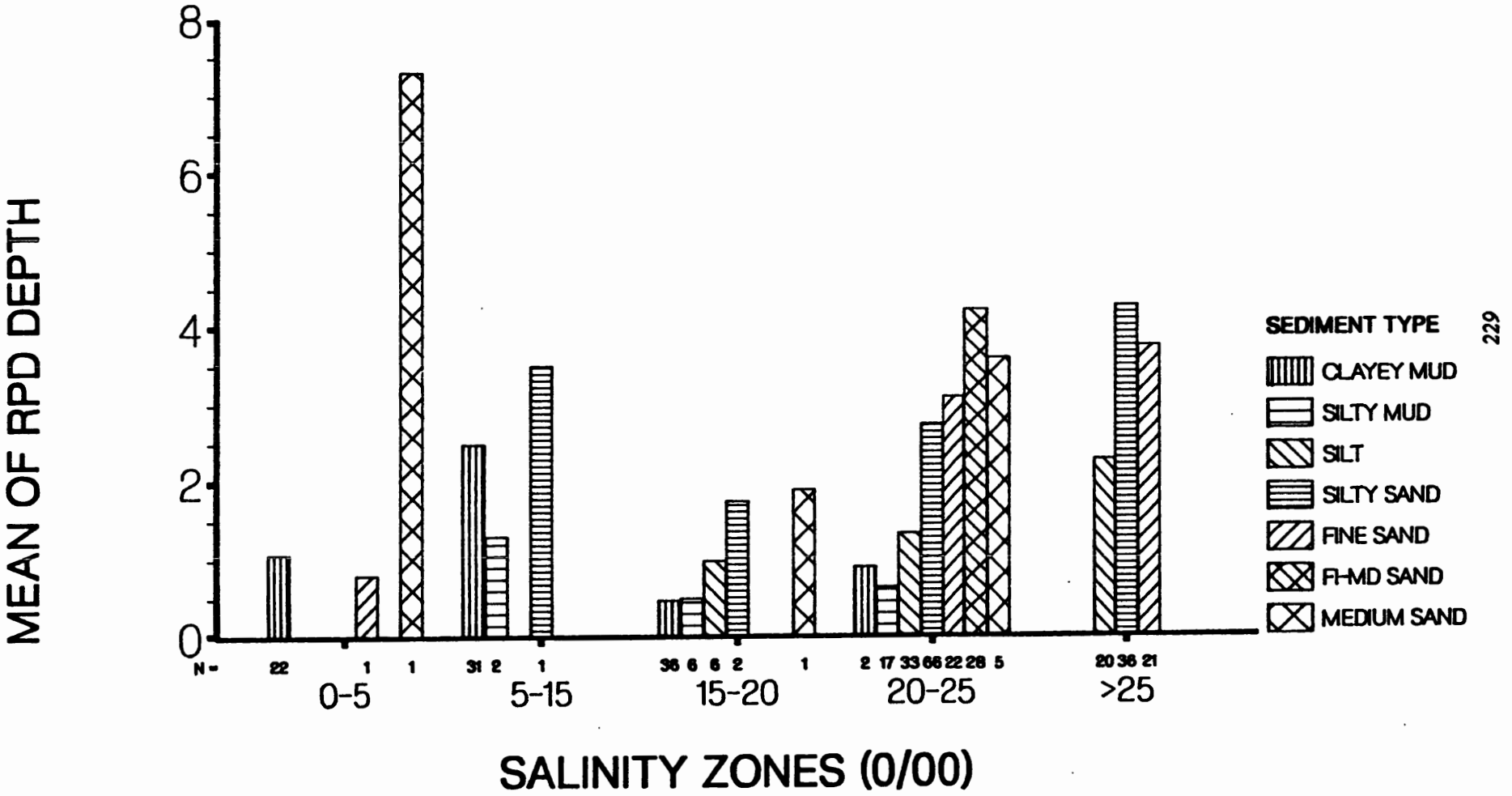


Figure 3. Depth of the apparent RPD, from profile camera images, by salinity zone and sediment type.

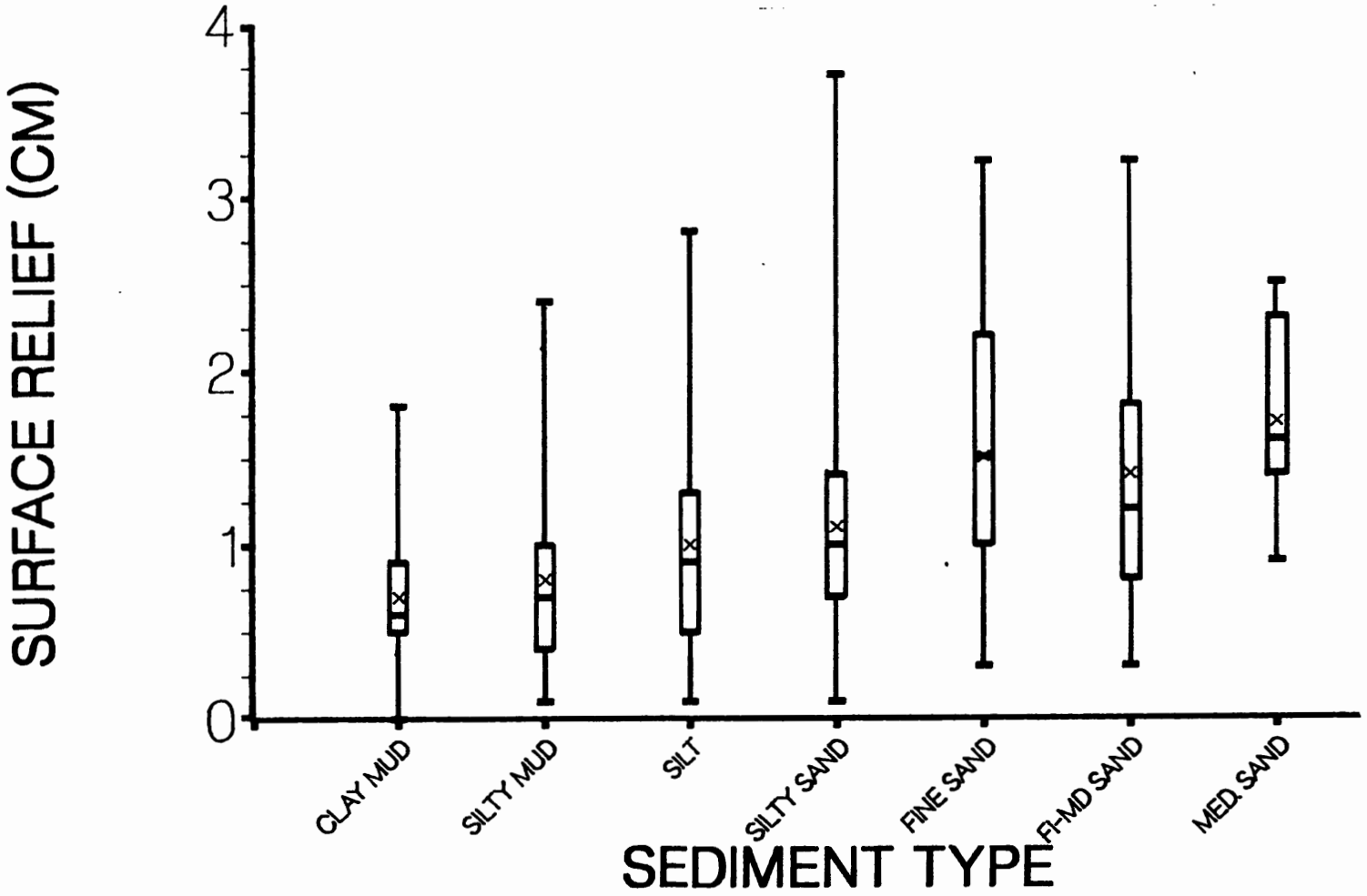


Figure 4. Surface relief, from profile camera images, by sediment type. Bar is median, x is mean, box is interquartile range, and end bars are total range.

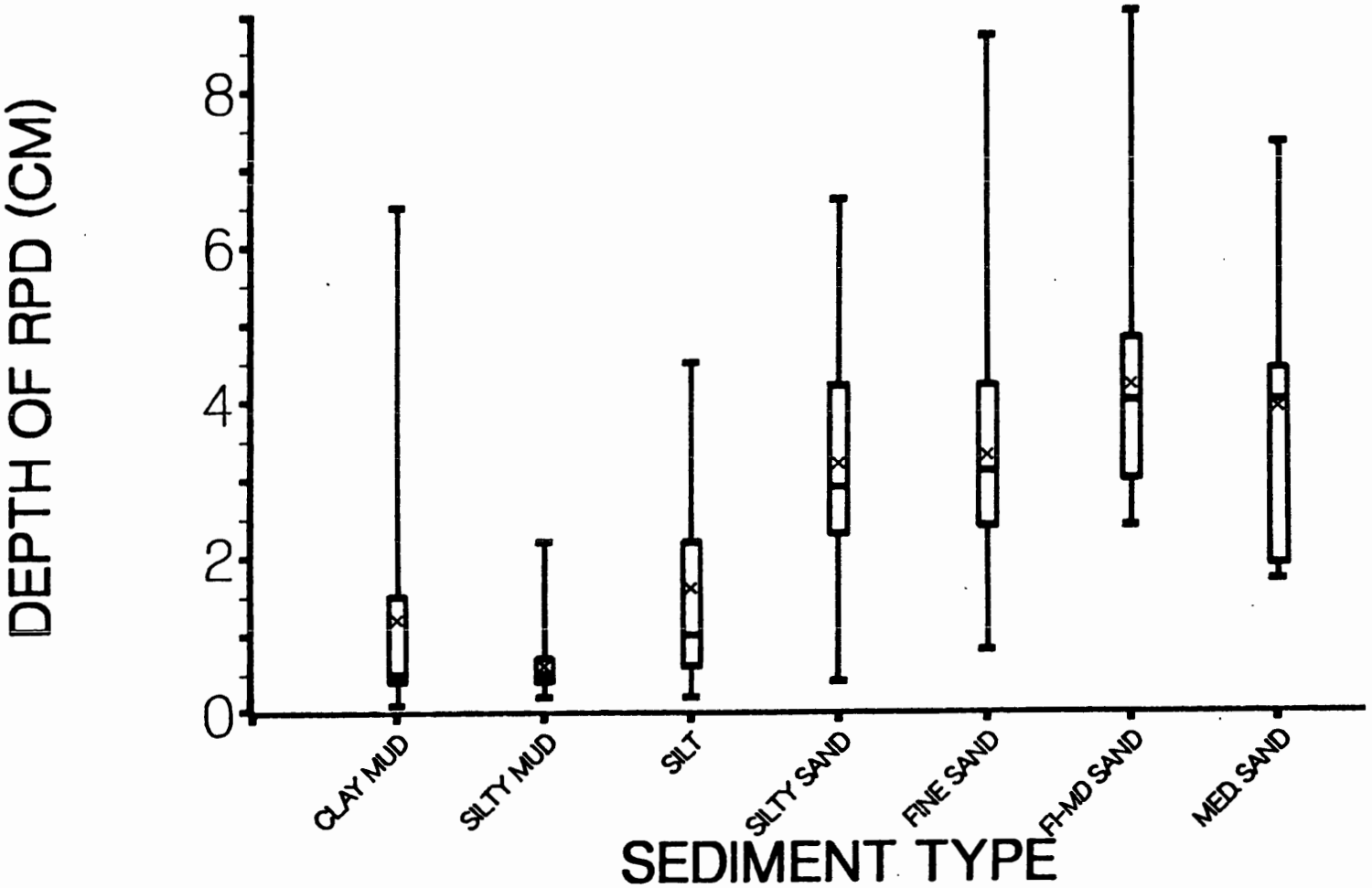


Figure 5. Depth of apparent RPD, from profile camera images, by sediment type. Bar is median, x is mean, box is interquartile range, and end bars are total range.

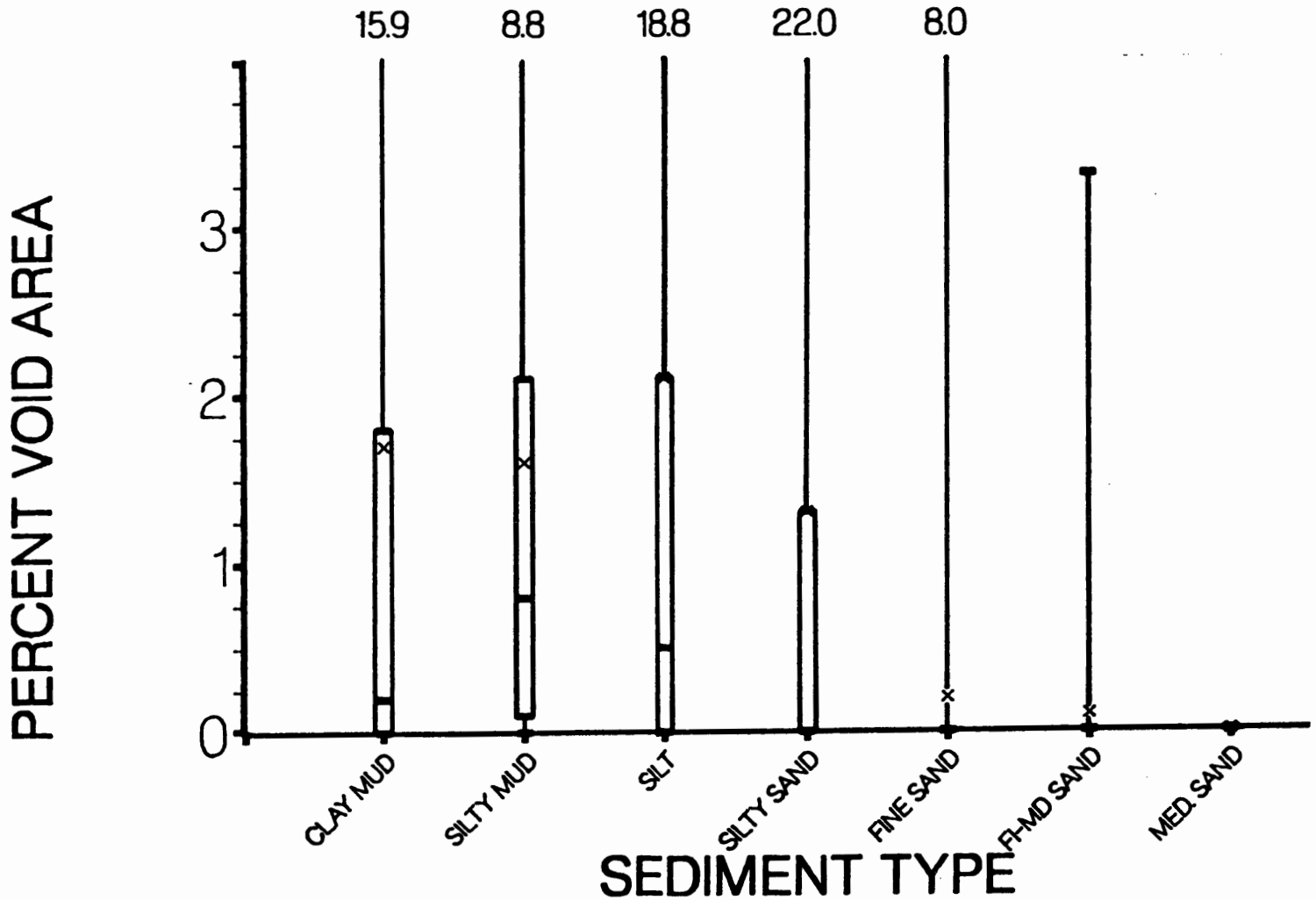


Figure 6. Percentage of image area that was voids, standardized to 15 cm prism penetration. Bar is median, x is mean, box is interquartile range, and end bars are total range.

communities. Within each salinity zone, as defined, the basic patterns of surface relief and apparent RPD depth were similar by sediment type. At salinities above 5 ppt patterns in void area by sediment type were also similar. At salinities less than 5 ppt functional groups of infauna capable of producing subsurface feeding voids are limited in abundance (Schaffner et al. 1987a). See Figure 7 for representative images.

In sediments ranging from mud to silty sands, the complexity of surface relief and apparent RPD depth increases with increasing grain size. This is due mainly to the increasing dominance of infauna in sediment mixing processes along this sediment gradient (Schaffner et al. 1987b). With the transition to sand sediments physical forces dominate surface relief and RPD depth. In sands, bed forms are the predominant surface relief and the apparent RPD layer tends to be more uniform, not following the surface contours provided by bed forms. Apparent RPD layers in clayey and silty muds tended to be broadly uniform, following the contour of the surface sediments, upon which a smaller scale (on the order of mm's) convolution is superimposed. In silts and silty sands the apparent RPD is most complex and convoluted providing a greatly increased biologically reactive interface.

The degree of biogenically-induced structural complexity in Chesapeake Bay surface sediments, as documented by surface and profile imagery, might have important effects on cycling of dissolved and particulate substances at and through the sediment water interface. For example, consider the processes associated with geochemical cycling across the RPD layer. While flux rates are typically based on simple areal measurement and the RPD is considered to be a simple contact plane between aerobic and anaerobic environments (Fenchel 1969), over most of the Chesapeake Bay's sediment landscape this assumption would lead to an underrepresentation of the actual area of the RPD layer. The results of numerous studies clearly demonstrate that biogenic structures are regions of enhanced biological and geochemical activity (Aller 1982, Aller and Yinst 1978, Aller and Aller 1986) and that the activities of infaunal organisms can increase flux across the oxic-anoxic sediment interface (Henriksen, Hanson and Blackburn 1980, Aller and Yinst 1978). Our documentation of the apparent RPD layer, a complicated surface much greater in actual area than a simple areal measurement would estimate, strongly suggests the need for further evaluation of the effects of infaunal benthos on sediment-water interface flux processes in the Chesapeake Bay.

CONCLUSIONS

There are broadscale patterns in the sediment landscapes of the Chesapeake Bay with regards to data collected by surface and profile imaging. General trends noted are:

- Biogenic voids are common and an integral part of sediment structure, except in sand and tidal freshwater and oligohaline habitats.

- Surface roughness increases concordant with increasing grain size. In fine grain sediments roughness is primarily biogenic and best developed in silts and silty-sands. In sands roughness is from current generated bed forms.

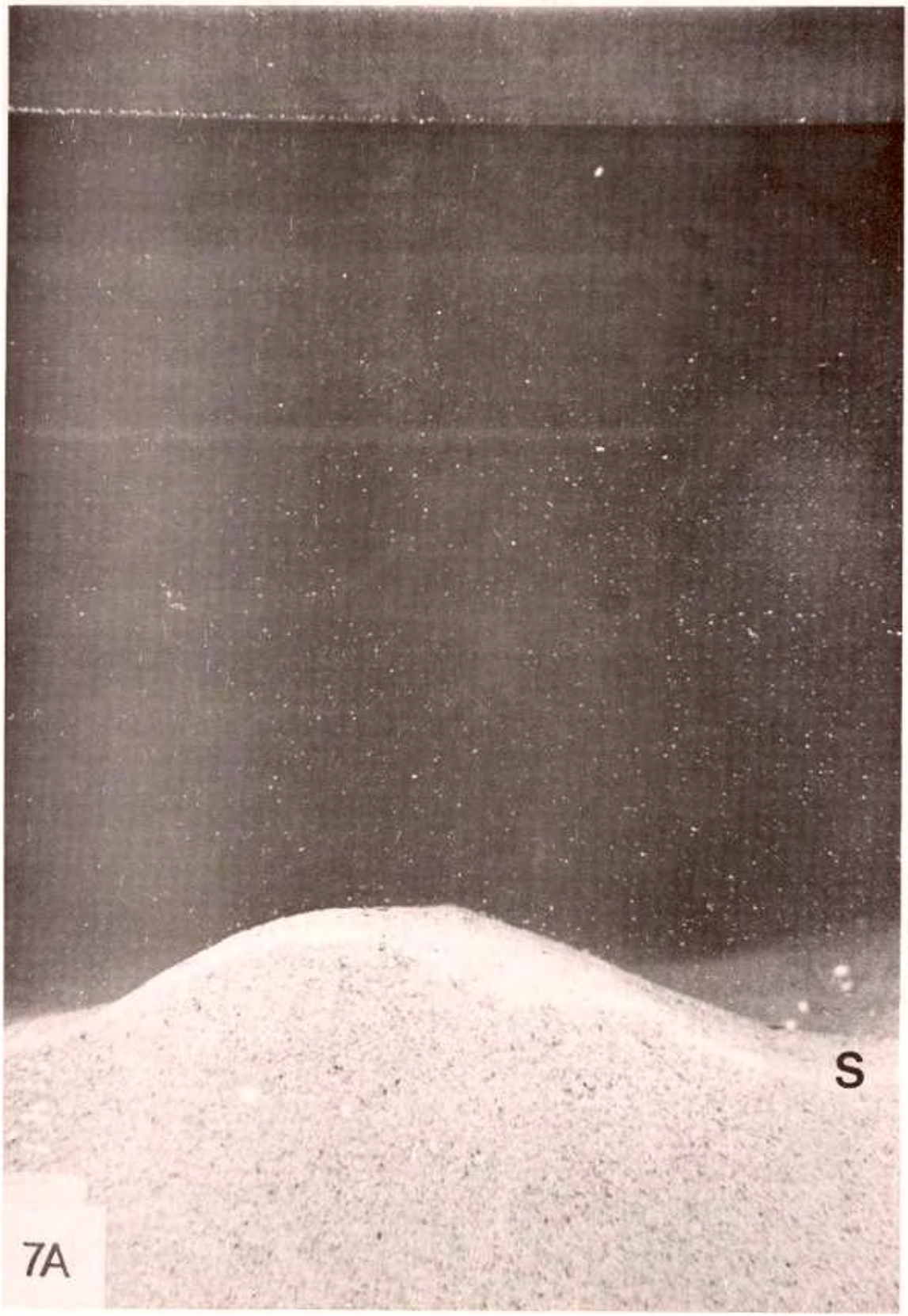
Figure 7. Examples of sediment profile images. Scale is 1X. S - sediment water interface, T - worm tube, V - feeding void.

a. 2 cm high bed form in medium sand at 5 m depth off Cape Charles in the Lower Chesapeake Bay.

b. Muddy sediments off Broome's Island, Patuxent River, showing thin (less than 1 cm) apparent RPD. Notice highly mottled appearance of subsurface sediment which may result from biogenic mixing. Also notice polychaete tubes at surface of sediment.

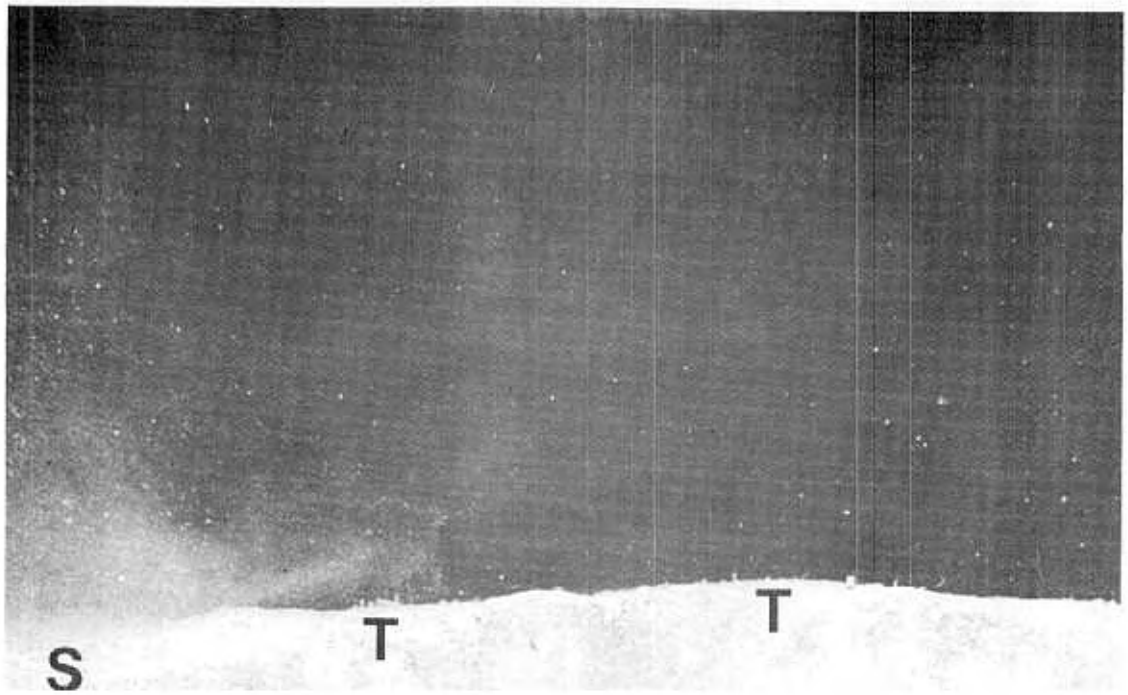
c. Silty sediments along Eastern Shore south of Cape Charles at 22 m depth. Apparent RPD is deeply convoluted and along the right of the image it extends down below the penetration of the camera prism. This type of apparent RPD is due to biogenic reworking by deep dwelling fauna. Surface relief in this image is all from biogenic activities. Notice small polychaete tubes at the surface.

d. Silty sediments near York River entrance channel at 10 m depth. Apparent RPD is deep in sediments and convoluted from biogenic activities. Large void is from head down deposit feeding of maldanid polychaetes.



7A

7B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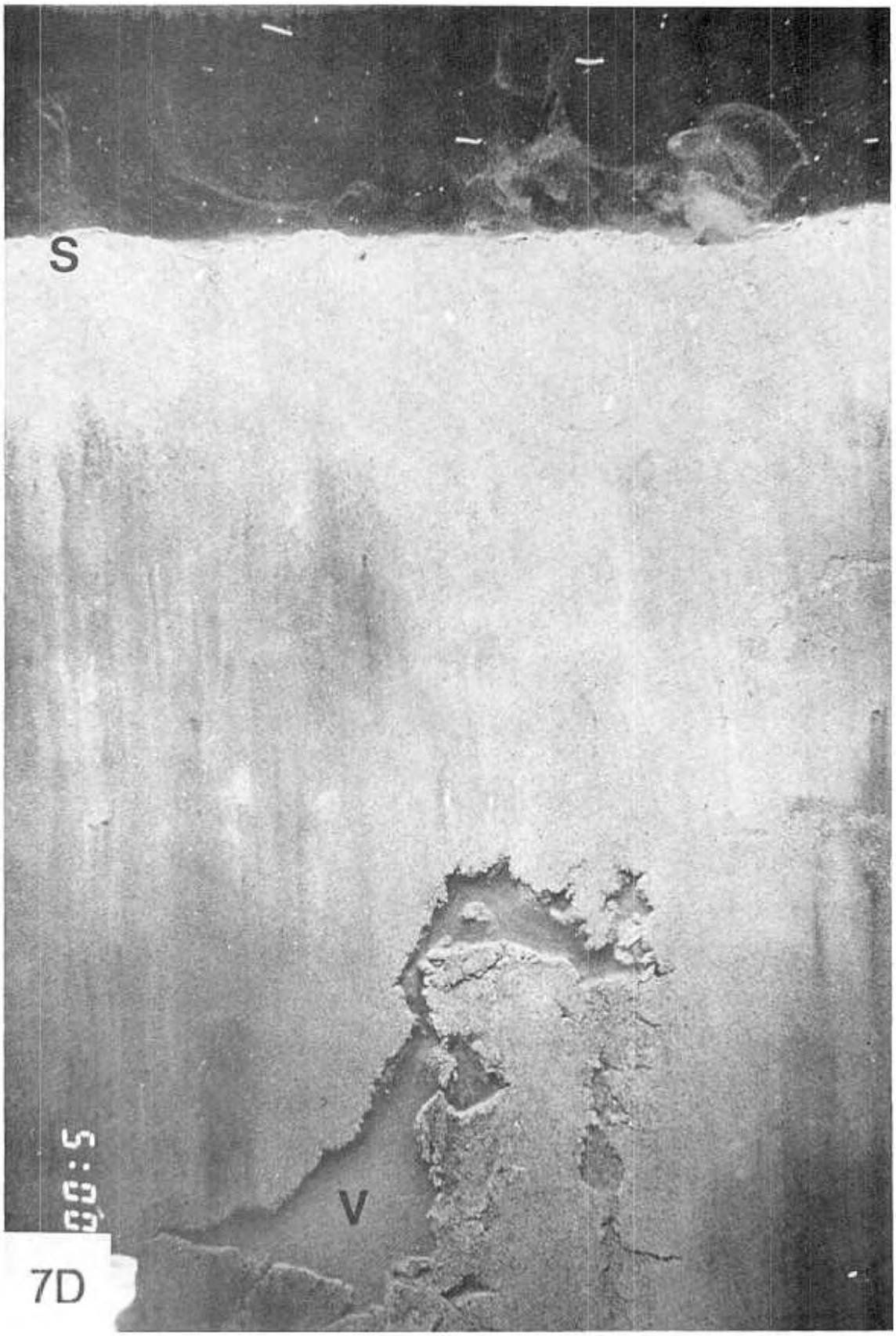

7D

- The biologically reactive interface, as represented by the apparent RPD, is greater than predicted by surface area alone. Deepest and most complex RPD's are found in silts and silty-sands at meso- and polyhaline salinities.

- Except when very thin (< 0.5 cm) and there is no deep biogenic activity, or in sand sediments, the apparent RPD layer is not a simple contact plane between aerobic and anaerobic environments. The actual RPD area could be many times that described by simple surface area.

REFERENCES

Aller, R. C. 1982. The effects of macrobenthos on chemical properties of marine sediments and overlying waters. In P. L. McCall and M. J. S. Tevesz (eds.), Animal-Sediment Relations: The biogenic alteration of sediments. Plenum Press, NY. p. 53-102.

Aller, R. C. and J. Y. Yinst. 1978. Biogeochemistry of tube-dwellings: a study of the sedentary polychaete Amphitite ornata (Leidy). Journ. Mar. Res. 36:201-254.

Aller, J. Y. and R. C. Aller. 1986. Evidence for localized enhancement of biological activity associated with tube and burrow structures in deep-sea sediments at the HEBBLE site, western North Atlantic. Deep-Sea Res. 33:755-790.

Boynton, W. R. and W. M. Kemp. 1985. Nutrient regeneration and oxygen consumption by sediments along an estuarine salinity gradient. Mar. Ecol. 23:45-55.

Day, M. E., L. C. Schaffner, and R. J. Diaz. (in press). Long Island Sound sediment quality survey and analyses. NOAA Tech. Memor. NOS OMA, 113 pp.

Fenchel, T. 1969. The ecology of marine microbenthos IV. Structure and function of the benthic ecosystem, its chemical and physical factors and the microfauna communities with special reference to the ciliated Protozoa. Ophelia 6:1-182.

Folk, R. L. 1974. Petrology of sedimentary rocks. Austin, Texas, Hemphill's. 170 p.

Garber, J. H. 1987. Benthic-pelagic coupling in the Chesapeake Bay. In M. P. Lynch and E. C. Krome (eds.), Perspectives on the Chesapeake Bay: Advances in estuarine sciences. CRC Pub. No. 127. Chesapeake Research Consortium, Gloucester Pt. VA. p. 11-34.

Henriksen, K., J. I. Hansen and T. H. Blackburn. 1980. The influence of benthic infauna on exchange rates of inorganic nitrogen between sediment and water. Ophelia. Suppl. 1:249-256.

Rhoads, D. C. and S. Cande. 1971. Sediment profile camera for in situ study of organism-sediment relations. Limnol. Oceanogr. 16:110-114.

Rhoads, D. C. and J. D. Germano. 1986. Interpreting long-term changes in benthic community structure: a new protocol. Hydrobiologia 142:291-308.

Olsen, C. R., N. H. Cutshall and I. L. Larsen. 1982. Pollutant-particle associations and dynamics in coastal marine environments: A review. Mar. Chem. 11:501-533.

Schaffner, L. C., R. J. Diaz, C. R. Olsen and I. L. Larsen. 1987. Faunal characteristics and sediment accumulation processes in the James River estuary, Virginia. Estuarine Coast. Shelf Sci. 25:211-226.

Schaffner, L. C., R. J. Diaz and R. J. Byrne. 1987b. Processes affecting recent estuarine stratigraphy pp. 584-599. In Coastal Sediments '87, Waterways Div., ASCE, New Orleans, LA.

SPSS. 1986. SPSSX user's guide, 2nd ed. SPSS Inc., Chicago, IL, 988 pp.

Stroup, E. D. and R. J. Lynn. 1963. Atlas of salinity and temperature distributions in Chesapeake Bay. Chesapeake Bay Institute Report 2, 410 pp.

Wright, L. D., D. B. Prior, C. H. Hobbs, R. J. Byrne, J. D. Boon, L. C. Schaffner and M. O. Green. 1987. Spatial variability of bottom types in the lower Chesapeake Bay and adjoining estuaries and inner shelf. Estuarine Coast. Shelf Sci. 24:765-784.

Understanding the Estuary: Advances in Chesapeake Bay Research. Proceedings of a Conference. 29-31 March 1988. Baltimore, Maryland. Chesapeake Research Consortium Publication 129. CBP/TRS 24/88.

Use of In Situ Nutrient Addition and Dilution Bioassays to Detect Nutrient Limitation in the Tidal Freshwater Potomac

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INTRODUCTION

Nuisance blooms of blue-green algae have regularly invaded the tidal freshwater Potomac and its embayments since 1983. Many of these blooms have been concentrated in the river stretch between Marshall Hall (River Mile 22) and Quantico (RM 40). Due to the expense of major upgrades of waste treatment in the Washington area, these unpredicted blooms have been a major cause of concern in water quality management. A blue-ribbon panel commissioned following the 1983 bloom concluded that the river system was phosphorus-limited and that the 1983 bloom was due to phosphorus that became available not from current waste discharges, but from an unexpected source, the river's sediment (Thomann et al. 1985). The mechanism for this release was hypothesized to be enhanced pH due to photosynthetic activity, a positive feedback mechanism. Similar, but more limited blooms have been experienced in 1985 and 1986.

Fundamental questions remain to be answered. Is phosphorus always limiting algal growth in the river? Are carbon or light limitation important? What about the role of climatic factors? If the system is not limited by phosphorus, how much phosphorus must be removed before phosphorus limitation and subsequent water quality improvement takes place?

Nutrient addition bioassays have been used by numerous workers as a method of determining growth limiting factors in nutrient-limited situations. In these experiments soluble nutrients (usually N and P) are added to closed containers of receiving water containing either native algae or representative cultures and growth is measured over a period of time against unspiked controls. Experimental designs have ranged from laboratory studies with single species growing in filtered, autoclaved water (Miller et al. 1978) to field designs in which nutrients are added to containers of native phytoplankton and incubated in the field (Goldman 1963, 1978). In very eutrophic systems where nutrients are not limiting,

little information can be gleaned from nutrient addition bioassays. Paerl and Bowles (1987) have proposed that nutrient dilution bioassays are needed in these rich waters to determine levels to which nutrients must be diluted before nutrient limitation can occur. It is only after nutrient limitation is attained that incremental removal of nutrients will result in decreases in algal crops.

The phytoplankton is a community, that is an assemblage of species. Each species has its own growth characteristics and each may respond differently to environmental characteristics. In addition, they compete for scarce resources and as these resources change so too do the competitive relationships among the species. Based on these considerations it is not hard to visualize that complex responses could occur when communities are confined in enclosures and pulsed with nutrients. Thus, in a study such as this it is useful to consider not only the aggregate response (chlorophyll, photosynthetic rates, total cell density), but also the responses of various species.

The purpose of this study was to conduct nutrient addition and nutrient dilution bioassays on Potomac river water from 2 sites (Gunston Cove and Indian Head) on two dates in September 1986. The method used was a modification of that of Paerl and Bowles (1987) with changes as noted below. Three questions were to be addressed:

(1) Are phytoplankton at these sites nutrient limited?
(2) If not, how much nutrient must be removed (in this case, diluted) to attain nutrient-limited status?
(3) Is there a difference in response among different major groups or even species of phytoplankton and do these differences help account for the aggregate response of the community? Do "nuisance" species and "desirable" species of phytoplankton differ in their response?

METHODS

Study sites were located in the tidal fresh portion of the Potomac River below Washington, D.C. One site identified as GC was located in Gunston Cove, a shallow embayment on the Virginia side of the river about 13 miles downstream of the Woodrow Wilson Bridge. The second site was located in the river mainstem off Indian Head about 19 miles downstream of the Wilson Bridge. This station was situated near the river channel in a stretch of the river in which blue-green algal blooms have developed in recent years. Experiments were conducted at each station during early and late September 1986.

At each site twelve polyethylene containers (5-gallon Cubitainers) were used. Six treatments were established in duplicate as follows: control, three levels of phosphorus addition (+P, +10P, +11P), 10% dilution with synthetic river water (-10%), 50% dilution with synthetic river water (-50%). Phosphorus additions were to be made by adding 20 ml of a stock solution containing 0.56 g K_2HPO_4/L (+P, +11P) and/or 6.07 g K_2HPO_4/L (+10P, +11P) giving a final added P concentration of 0.1 mg P/L (+P), 1.1 mg P/L (+10P), or 1.2 mg P/L (+11P). Dilutions were performed using a synthetic river water made up with ACS grade laboratory chemicals to simulate the major ionic composition of Potomac River water (Jones 1987).

Nutrient and synthetic water additions indicated above were made prior to filling of the bags with river water. In the field river water was pumped from a depth of 0.3 m (1 ft) directly into the bags in the following order: CTRL, +10P, +P, +11P, -10%, -50%. After filling and mixing the contents of each bag well, water was poured from each for analysis. Three samples were taken from each cubitainer: one for nutrient and pH analysis, a second for chlorophyll and photosynthetic rate determination at GMU, and a third for phytoplankton analysis. The phytoplankton sample was stored in an amber glass bottle to which 2 ml of acid Lugol's iodine solution was added, while the other two samples were held in 500 ml polyethylene bottles in a dark, insulated container at ambient temperature.

After sampling each bag was sealed with its plastic screw lid, lowered into the water, and attached with a metal clip to an eye screw on a floating wooden frame. After two days, each bag was removed from the water, mixed thoroughly, and sampled as on Day 0. During the first experiment several bags were lost at the Indian Head site due to exposure of the apparatus to constant wave action. The site was also a long run from the launching ramp in Gunston Cove and we could not monitor it every day. Thus, for the second experiment bags were filled at Indian Head, covered, and moved to a station in the outer part of Gunston Cove near the Coast Guard station for incubation.

Samples for chlorophyll and photosynthetic rate were returned to George Mason University within a few hours of collection. Chlorophyll was measured by filtration, extraction with acetone and DMSO, and fluorometric determination of pigments (Jones 1987). Light saturated photosynthetic rate was measured using C-14 uptake in the lab at ambient temperature under artificial illumination (Jones 1987). Phytoplankton were enumerated by species using the inverted microscope-settling chamber technique (Lund et al. 1958, Jones 1987). At least 300 cells (1500 in GC samples) were counted. To minimize variation between counters, one person counted all GC samples (Ms. Vicki Andrele) and another person all IH samples (Dr. Claire Buchanan). Biovolumes were determined for dominant species by measuring cell dimensions and approximating cell shape to an appropriate geometric solid for which volume can be easily calculated. It should be noted that cells may vary in size and that small changes in dimensions lead to large changes in cell volume. Nutrient analysis was conducted by Lower Potomac Pollution Control Plant lab.

Changes in each parameter were subjected to one-way analysis of variance to determine if treatment means were significantly different from one another. The Tukey HSD test was used to compare treatment means with the control. The SYSTAT statistical package was used to conduct all statistical tests.

RESULTS

Gunston Cove

At the beginning of the September 2-4 experiment in Gunston Cove enclosures reflected the large algal populations present in the cove. Initial chlorophyll levels in undiluted treatments (+P, +10P, +11P, and CTRL) were 233-295 ug/L indicating bloom conditions. Photosynthetic

rates were 1079-1367 ug C/L/hr in undiluted treatments. Chlorophyll and photosynthetic rate were proportionately lower in the -10% and -50% dilutions. Blue-green algae were the dominant phytoplankton group in terms of cell density comprising 84% of all cells counted. The most abundant species was Merismopedia tenuissima comprising 60-80% of the total count in initial samples. Chroococcus dispersus and Oscillatoria planctonica were found in substantial numbers (2.8-19.8%). The bloomformer Microcystis aeruginosa was present, but not abundant (0.4-2.1% of total cells). Cryptophytes composed about 7% of the total cell density. When biovolumes were computed, cryptophytes became dominant comprising about two-thirds of the total biovolume. This was due to their much larger size when compared to the blue-greens.

After two days cell density increased in all treatments (Figure 1a). Change in cell density was significantly different among treatments, but no treatment was significantly different from the control. The increase was greatest in the control and +P treatments and least in the +10P treatment. Changes in biovolume were not significant among treatments. After two days chlorophyll values had decreased in all treatments. In the control, chlorophylls decreased by 22%. Similar decreases were observed in all other treatments except -50% where little change in chlorophyll was observed. None of the responses in chlorophyll were significantly different from the control. Photosynthetic rates in the control decreased 24%. All treatments to which P had been added showed increases in photosynthetic rate of from 4 to 44% which were significantly greater than the control. Dilutions showed decreases in photosynthetic rate which were not different from the control.

Green algae increased in most treatments, but showed no significant differences among treatments (Figure 1b). Diatoms showed little change under most treatments, but were significantly enhanced in the -50% dilution. Cryptophytes increased slightly in the control and decreased in all three nutrient addition treatments, but were much higher in the -50% dilution. Although not significantly different than the control, the -50% dilution did result in significantly more cryptophytes than the nutrient addition treatments. The blue-greens were the group most strongly responsible for the observed increase in cell density. Merismopedia tenuissima continued to be the overwhelming dominant comprising 45-62% of cells counted in all treatments. Chroococcus dispersus and Oscillatoria planctonica remained subdominant with 7-26% of cells counted. Microcystis aeruginosa density increased at about the same rate as the community as a whole retaining a 1-2% share of total cells.

Initial conditions at Gunston Cove during the September 23-25 experiment were similar to those observed earlier in September. Initial chlorophyll levels in the undiluted treatments were even higher than those found in early September ranging 317-358 ug/L. Photosynthetic rates were also higher: 1845-2041 ug/L. Chlorophyll and photosynthetic rate were proportionately less in dilution treatments. Initial phytoplankton densities in the late September experiment were similar to those observed in early September (300,000-400,000 cells/mL) as was species composition. Blue-greens were again the most abundant group at the beginning of the experiment comprising 82% of total algal cells. As in early September Merismopedia tenuissima was the dominant species accounting for 44-63% of all cells counted. Chroococcus dispersus and Oscillatoria planctonica

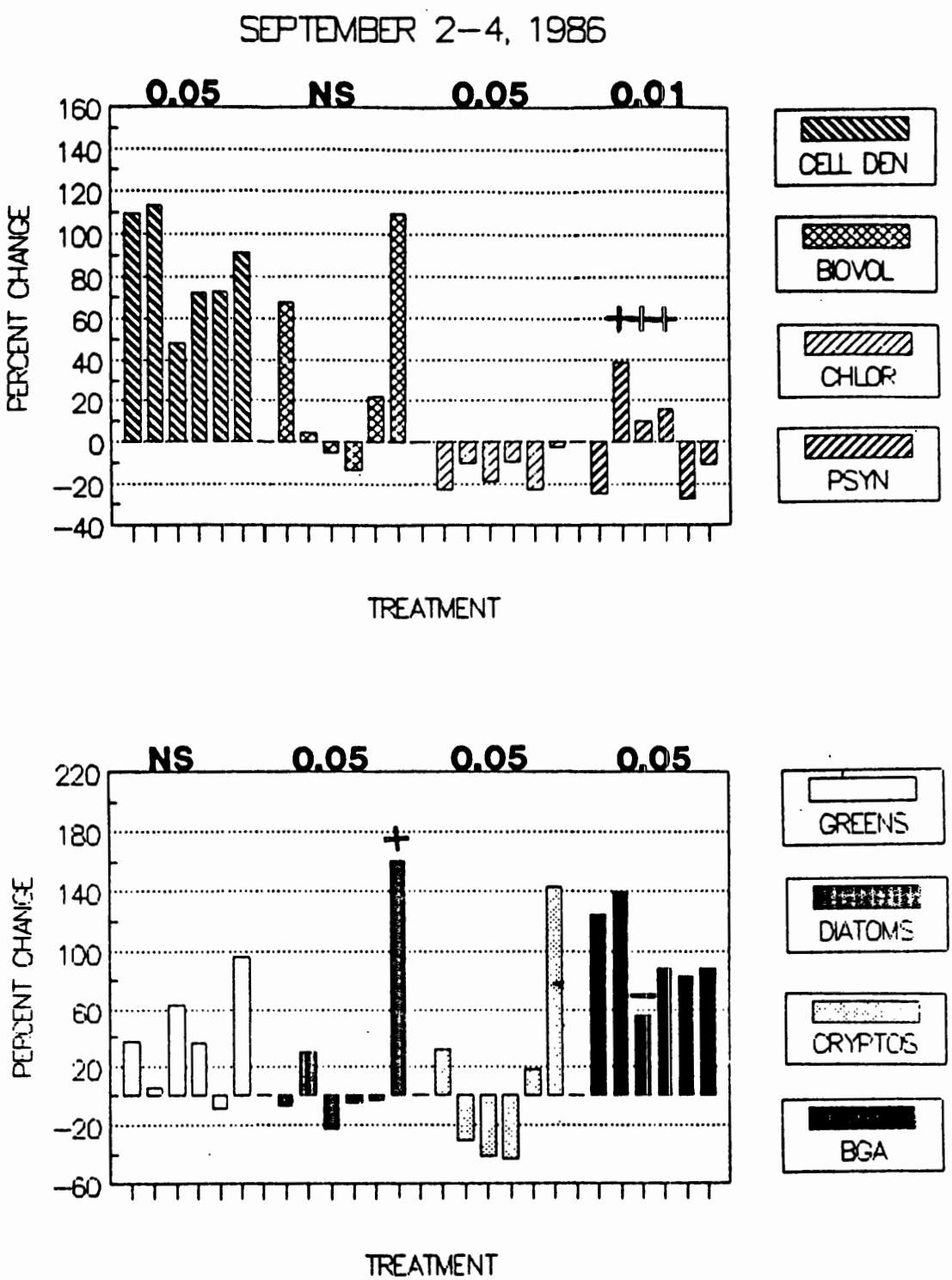

Figure 1. Response of phytoplankton in <u>in situ</u> bioassays. Gunston Cove, September 2-4, 1986. a. (upper) Percent change in cell density (cells/mL), biovolume (um^3/L), chlorophyll (ug/L), photosynthetic rate (ug/L/hr). b. (lower) Percent change in density of green algae, diatoms, cryptophytes, and blue-greens (cells/mL). Six bars for each parameter indicate response in control, +P, +10P, +11P, -10% dilution, and -50% dilution respectively. Numbers across top of graph indicate significance of one-way ANOVA among treatments for each parameter. + and - connote treatment means significantly different from the mean for each parameter.

continued to be subdominant (5-25%). Again Microcystis aeruginosa was present, but not particularly abundant (0.5-1.3%). In terms of algal biovolume, cryptophytes were again the most important group comprising 71% of the total.

Cell density increased significantly in two of the three nutrient addition treatments; nutrient dilution treatments showed little change relative to the control. Biovolume increased in most treatments, but was significantly greater than the control only in +10P. Chlorophyll levels decreased in almost all treatments during the late September GC experiment. Decreases of 14-16% were observed in the controls with similar declines in most other treatments except -50% where a small increase was measured. Photosynthetic rate decreased in all treatments. The decrease in the controls was 26-29% while that in the nutrient additions was 2-18%. More substantial declines were observed in the nutrient dilution treatments.

Greens increased in all treatments especially those with large additions of P with positive changes of up to 200% being found (Fig. 2b). Scenedesmus bijuga was the most abundant chlorophyte accounting for as much as 2% of the total count. Diatoms did poorly in all treatments, decreasing 45-80% in all nutrient addition and control treatments and somewhat less in most dilution treatments. Cryptophytes declined 0-40% in all treatments except -50% dilution, where a significant increase (about 80%) was registered. Blue-greens remained the dominant group and followed trends similar to those in total cell counts. Increases were significantly greater than the mean in two of three nutrient addition treatments. Merismopedia tenuissima remained the dominant comprising 34-54% of all cells counted, a slight drop in dominance. Chroococcus dispersus and Oscillatoria planctonica were subdominant (6-30%). Microcystis aeruginosa increased slightly in dominance to 1.7-2.7%.

Indian Head

Initial conditions for the September 23-25 experiment at Indian Head reflected much lower algal standing crop. Chlorophyll levels were similar in all undiluted samples at the beginning of the experiment ranging from 29-34 ug/L. Photosynthetic rates were varied from 290-370 ug C/L/hr. Dilution treatments showed proportionately lower chlorophyll and photosynthetic rate values. Initial cell densities were 60,000-90,000 cells per mL, blue-greens comprising 46%. Diatoms, greens, and small flagellates were also important. Merismopedia tenuissima (15-36%), Chroococcus dispersus (9-13%), and Oscillatoria planctonica (4-12%) were the dominant blue-green species. Discoid centric diatoms were also a dominant group numerically (20-24%). Diatoms were dominant in biovolume (39%) followed closely by the unidentified flagellates.

Phytoplankton densities increased strongly in all treatments (Fig. 3a) with highest increases of about 500% in the +P treatment (significantly greater than the control) and lowest increase of 100-200% (significantly less than the control in -50% dilution. Phytoplankton biovolume followed a similar trend with increases found in all treatments and significant relationships with the control identical to those in densities. The greatest increase in biovolume was 600% in +P and the least was slightly less than 200% in the -50% treatment. After 2 days chlorophyll increased in all samples, but none were statistically

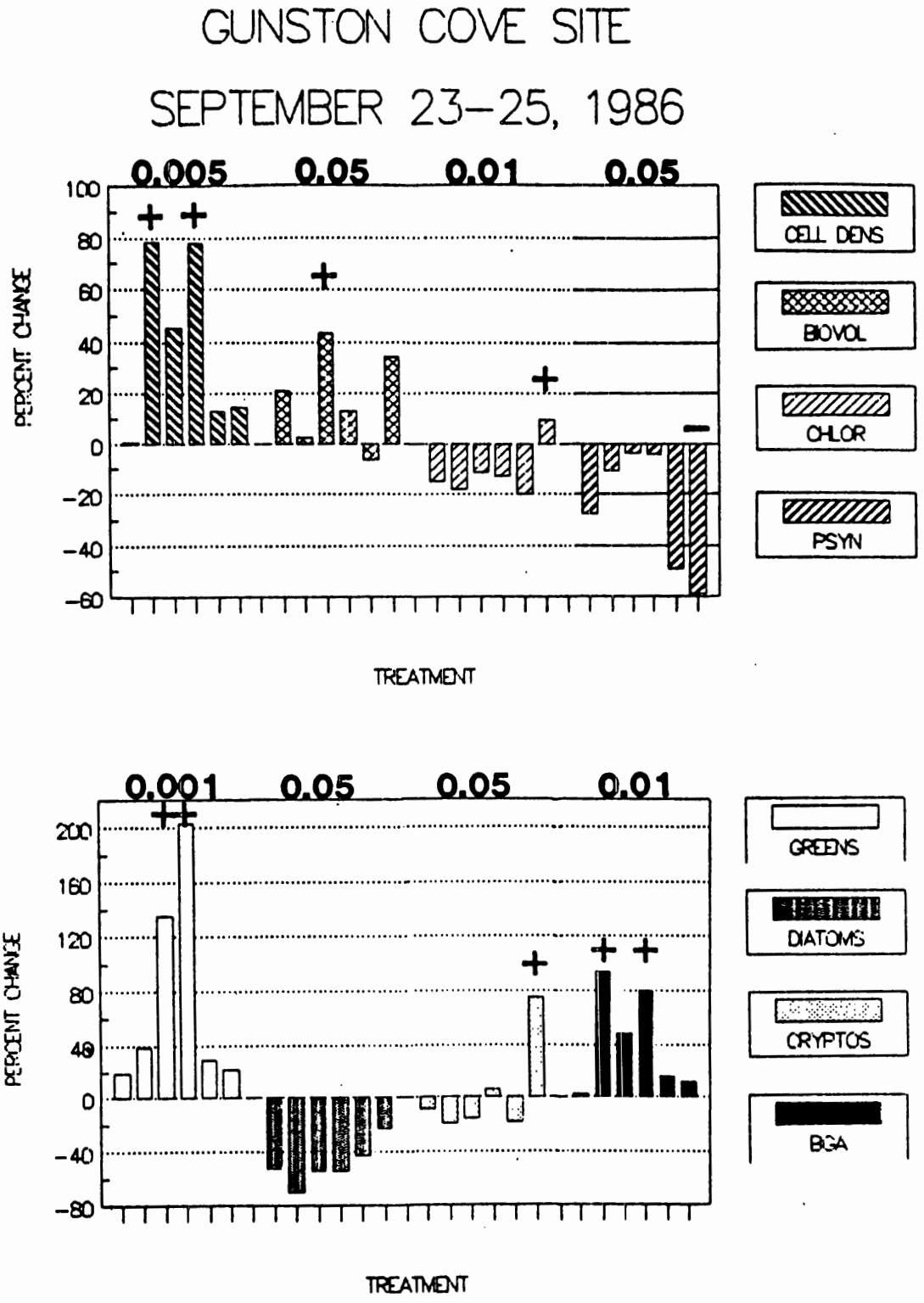

Figure 2. Response of phytoplankton in <u>in situ</u> bioassays. Gunston Cove, September 23-25, 1986. a. (upper) Percent change in cell density (cells/mL), biovolume (um^3/L), chlorophyll (ug/L), photosynthetic rate (ug/L/hr). b. (lower) Percent change in density of green algae, diatoms, cryptophytes, and blue-greens (cells/mL). Six bars for each parameter indicate response in control, +P, +10P, +11P, -10% dilution, and -50% dilution respectively. Numbers across top of graph indicate significance of one-way ANOVA among treatments for each parameter. + and - connote treatment means significantly different from the mean for each parameter.

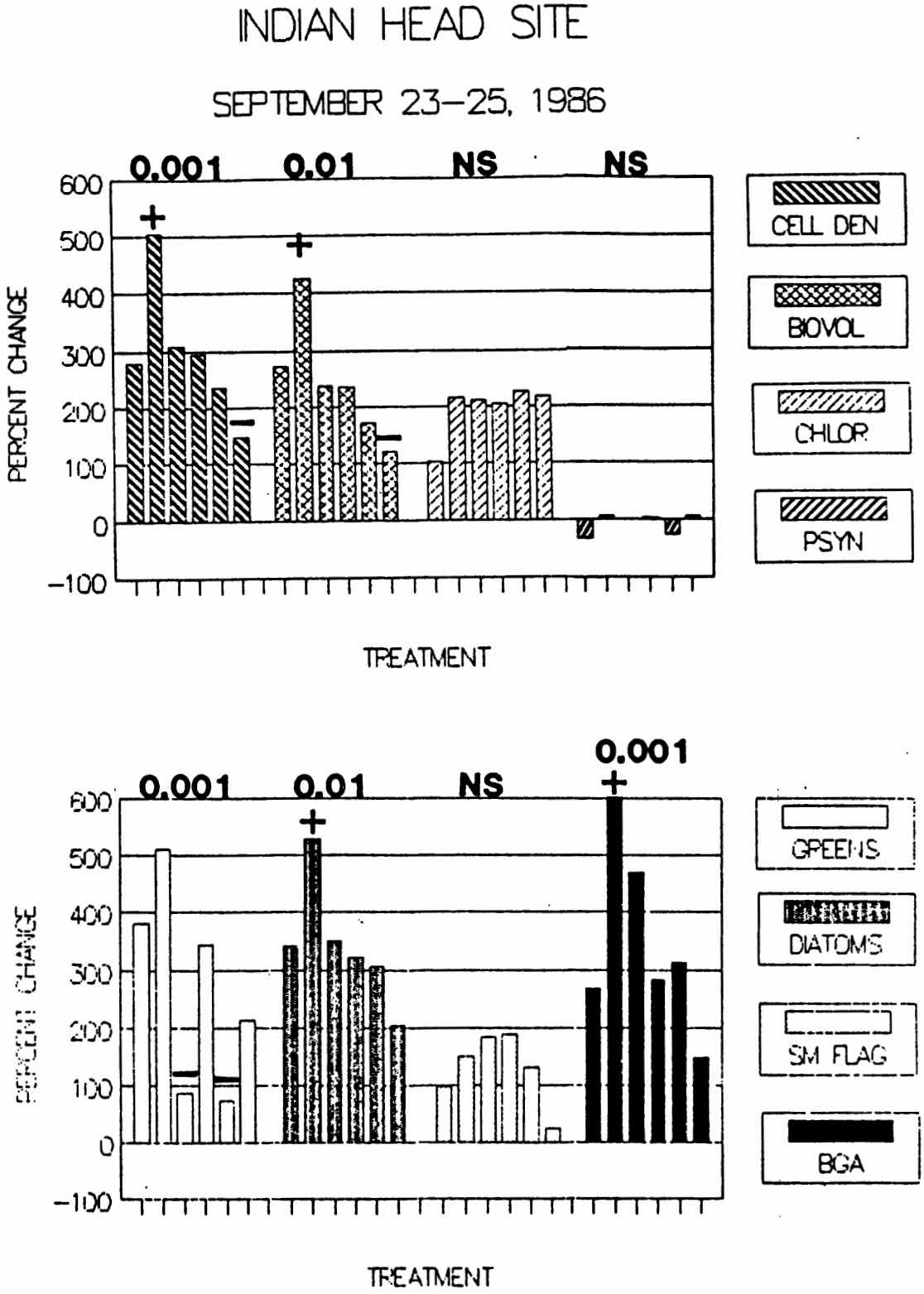

Figure 3. Response of phytoplankton in in situ bioassays. Indian Head, September 23-25, 1986. a. (upper) Percent change in cell density (cells/mL), biovolume (um^3/L), chlorophyll (ug/L), photosynthetic rate (ug/L/hr). b. (lower) Percent change in density of green algae, diatoms, small flagellates, and blue-greens (cells/mL). Six bars for each parameter indicate response in control, +P, +10P, +11P, -10% dilution, and -50% dilution respectively. Numbers across top of graph indicate significance of one-way ANOVA among treatments for each parameter. + and - connote treatment means significantly different from the mean for each parameter.

different from the control. Photosynthetic rate decreased markedly in the controls and in the -10% dilution treatment. Other treatments showed little consistent change from the initial readings; no changes were significantly different from the control.

All major groups increased in all treatments (Fig. 3b). Increases in chlorophytes were much lower at +10P and -10% dilution than in other treatments, being significantly less than the control. Diatom increases were highest in +P (500%, significantly greater than the control) and were generally 200-400% in other treatments. Discoid centrics increased as overall numbers climbed maintaining 23-27% of cell numbers and probably a greater percentage of overall biomass. Blue-green increases were similar to those in total phytoplankton with greatest increase in +P (significantly greater than the control) and lowest increase in -50% dilution (100-200%). Merismopedia tenuissima remained the dominant blue-green (29-39%); Chroococcus dispersus (7-11%) and Oscillatoria planctonica (5-10%) were subdominant cyanophytes.

DISCUSSION

Nutrient addition bioassays in various forms have been used by numerous investigators to determine which nutrient element is limiting algal growth in natural waters. These tests assume that if X is the limiting nutrient, then addition of X to a water sample containing algae will cause an increase in algal growth and/or standing crop. If X is not limiting, then no response will occur. In hypereutrophic waters, all nutrient elements may be present in excess. In these cases algal growth is probably limited by carbon or light and nutrient levels in receiving waters may have to be reduced markedly before any detectable change in algal activity is observed. Thus, expensive nutrient removal schemes may be implemented without a detectable response in algal biomass. Nutrient dilution bioassays (Paerl and Bowles 1987) offer a means of ascertaining to what levels nutrients must be reduced to achieve nutrient limitation; at this point further nutrient reductions will have proportional effects on algal biomass.

In this work nutrient addition and nutrient dilution bioassays were both utilized in situ to detect the degree of nutrient limitation at Potomac River embayment and mainstem stations subject to algal blooms. During the experiment an algal bloom was in progress at the embayment site in Gunston Cove with chlorophyll levels in excess of 200 ug/L. The bloom in was composed of cyanophytes Chroococcus and Merismopedia with a substantial component of green algae and cryptophytes compared with previous Gunston Cove blooms dominated by Microcystis aeruginosa. At the Indian Head site chlorophyll levels (30-50 ug/L) were well below bloom levels observed there in some previous years. Merismopedia, Chroococcus, and Oscillatoria were the dominant blue-greens. Discoid centric diatoms were very important in the river. Other parameters reflected the magnitude of algal standing crops. Initial pH at Gunston Cove was highly elevated (9.0-9.6) compared with Indian Head (7.6-7.9).

It is important to recall the differences in depth at the two sites. At the GC site with water only 1.5 m deep, the algae circulate only through a shallow, near-surface water column. At the IH site, phytoplankton circulate through 5-10 m of water column. Thus the IH

phytoplankton spend more time in the dark deeper waters and thus would be more likely to be light-limited. Light limitation could occur even at the GC site if large phytoplankton populations build up. One result of enclosing phytoplankton in bags is that they spend more time near the surface and probably experience higher light levels. Another result is that being enclosed, turbulence is decreased and increased sedimentation may occur. In addition decreased turbulence may lead to an increase in the boundary layer around individual cells and more difficulty in obtaining nutrients. Finally, enclosure with only a small amount of air and decreased surface turbulence may lead to carbon limitation as carbon dioxide is depleted from the enclosed water and air.

Phytoplankton communities at Indian Head responded very favorably to enclosure. Chlorophyll increased in all treatments and photosynthetic rate responded favorably in the first experiment. Total cell density and biovolume and the density of all major groups also increased in all treatments at Indian Head. This response probably reflects increased light availability when the algae are held in an enclosure near the water's surface as opposed to circulating over the entire water column. Since algal crops were not excessive and final pH was less extreme (less than 9.4), carbon would be a less serious limiting factor than at Gunston Cove.

Indian Head communities responded positively to +P addition, although the response to +10P and +11P was not statistically significant. Cell density, biovolume, diatoms, and blue-greens all increased significantly faster in the +P treatment than in the controls. Indian Head communities responded negatively to -50% dilution with cell density and biovolume significantly less than the controls. In addition to data reported here, particulate phosphorus and organic nitrogen also increased faster in nutrient addition treatments than in the control (Jones 1987).

Phytoplankton communities at Gunston Cove did not respond as favorably to enclosure as did those at Indian Head. In almost every case, chlorophyll levels decreased during the incubation period. The water column at Gunston Cove is much shorter and thus algae are already spending a large portion of their time near the surface. Negative factors associated with enclosure may have overwhelmed any slight light advantage. Due to the higher standing crops already present in the bags, dissolved CO_2 was exhausted much more rapidly in the enclosure. At its extreme in this study, dissolved CO_2 reached a low of 0.034 umoles/L in GC enclosures (as compared with a low of 0.65 uM at Indian Head). This is well below the level thought to end photosynthesis by green algae and would be expected to severely stress even blue-greens (King 1970). On the other hand cell densities increased in virtually all treatments in both GC experiments. This increase was largely accounted for by very small cyanophytes. When algal biovolume was computed, this strong positive effect of enclosure disappeared. Large-bodied algae, particularly cryptophytes and diatoms, were negatively affected by enclosure with nutrient addition and caused biovolume to decline in many cases. Numerous other studies have shown that low dissolved CO_2 concentrations and high pH inhibit photosynthesis by greens, diatoms, and other algae and enhance the dominance of cyanophytes (Shapiro 1973, King 1970, Talling 1976, Moss 1973). Interestingly, cryptophytes and diatoms were actually stimulated by nutrient dilution in the Gunston Cove samples. This may also indicate the effect of carbon limitation which is alleviated by the high

alkalinity and moderate pH of the synthetic river water and the lower photosynthetic activity in the diluted medium. Chlorophyll behaved similarly to large-bodied forms while photosynthetic activity mirrored changes in small cyanophytes. This is consistent with the general observation of a negative correlation between size and growth rate (Reynolds 1984).

Photosynthetic rates in GC experiments often increased more or at least decreased less in nutrient addition treatments compared with the controls. In the first experiment a strong increase was found with nutrient addition when compared with the controls. In the second experiment the decrease in photosynthetic rate was less in the nutrient additions than in the controls. Chlorophyll levels were less supportive of the nutrient stimulation hypothesis particularly in the second experiment. Cell densities were consistently higher in nutrient enriched cultures in the first experiment, but not the second. Cell biovolume was negatively affected by nutrient addition in the second experiment; no trends were obvious in the first experiment. Increases in particulate phosphorus and organic nitrogen were consistently and often significantly greater in nutrient addition treatments than in the control (Jones 1987). The stronger response to nutrient addition in the second GC experiment may have reflected an increase in P limitation as the bloom proceeded. As discussed above a clear response to nutrient addition was obstructed by carbon limitation which severely inhibited the dominant cryptophytes.

In summary, the phytoplankton of the freshwater tidal Potomac responded to *in situ* nutrient bioassay in a manner consistent with phosphorus limitation of growth. Responses of major groups of algae differed markedly especially when high algal crops rendered carbon limitation a problem in the enclosed containers. Future studies should ensure against carbon limitation by using open enclosures, shorter incubations, and/or inorganic carbon additions.

ACKNOWLEDGEMENTS

I wish to thank Hans Paerl for providing a prepublication draft of his methods paper on nutrient dilution bioassays and for advice on setting up these experiments. Vicki Andrle and Claire Buchanan provided phytoplankton enumerations and Sue Touart and Ann Powel assisted in laboratory analyses. Allan Hide provided field support. Fairfax County's Lower Potomac Pollution Control Plant provided nutrient analyses. Funding for this study was provided by the Metropolitan Washington Council of Governments. The assistance of Wendy Chittenden and Stuart Freudberg is greatly appreciated.

LITERATURE CITED

Goldman, C.R. 1963. The measurement of primary productivity and limiting factors in freshwater with C-14. Pages 103-113 in M.S. Doty (ed.) Proceedings of the conference on primary productivity measurement, marine and freshwater. U.S. Atomic Energy Commission T.I.D.-7633.

Goldman, C.R. 1978. The use of natural phytoplankton populations in bioassay. Mitteilungen. Internationale Vereinigung fur Theoretische und Angewandte Limnologie 21: 364-371.

Jones, R.C. 1987. A study of nutrient limitations to algal growth in the Potomac River Estuary. Final Report submitted to Metropolitan Washington Council of Governments. Washington, D.C.

King, D.L. 1970. The role of carbon in eutrophication. Journal Water Pollution Control Federation 42: 2035-2051.

Miller, W.E., J.C. Greene, and T. Shiroyama. 1978. The Selenastrum capricornutum Printz algal assay bottle test: experimental design, application, and data interpretation. EPA-600/9-78-018.

Moss, B. 1973. The influence of environmental factors on the distribution of freshwater algae: an experimental study. II. The role of pH and the carbon dioxide-bicarbonate system. Journal of Ecology 61: 157-177.

Paerl, H.W. 1983. Factors regulation nuisance blue-green algal bloom potentials in the lower Neuse River, N.C. Water Resources Research Institute of the University of North Carolina. Report 188.

Paerl, H.W. and N.D. Bowles. 1987. Dilution bioassays: their application to assessments of nutrient limitation in hypereutrophic waters. Hydrobiologia 146: 265-273.

Reynolds, C.S. 1984. The Ecology of Freshwater Phytoplankton. Cambridge University Press.

Shapiro, J. 1973. Blue-green algae: why they become dominant. Science 179: 382-384.

Talling, J.F. 1976. The depletion of carbon dioxide from lake water by phytoplankton. Journal of Ecology 64: 79-121.

Thomann, R.V., N.J. Jaworski, S.W. Nixon, H.W. Paerl, and J. Taft. 1985. The 1983 Algal Bloom in the Potomac Estuary. Prepared for the Potomac Strategy State/EPA Management Committee.

Understanding the Estuary: Advances in Chesapeake Bay Research. Proceedings of a Conference. 29-31 March 1988. Baltimore, Maryland. Chesapeake Research Consortium Publication 129. CBP/TRS 24/88.

Abstract only

Numerical Tagging of Phosphorus in the James Estuary

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In estuarine eutrophication analysis, one of the key questions often asked is to where nutrients from particular source(s) would be transported. For example, in the James Estuary, phosphorus input to the upper estuary could be incorporated into the biomass of phytoplankton in the water column, deposited into the sediments, or transported to the lower estuary. Perhaps a more meaningful question is: how much phosphorus in the peak algal biomass is from a given source? A simple component analysis is not appropriate for the eutrophication modeling analysis simply because of the nonlinear relationship of the phytoplankton growth-nutrient limitation dynamics in the system. A numerical tracer was added to a water quality model of the James Estuary to determine the fate of phosphorus in the system. That is, a source or sources of phosphorus was numerically labeled and added to the James Estuary. The model was then used to quantify the amount of such labeled phosphorus in different components of the water column: organic phosphorus, orthophosphate, and algal biomass. Results of the analysis using the water quality data collected in September 1983 indicated that municipal wastewater discharges in the upper estuary (i.e., from Richmond, Falling Creek, Proctors Creek, and Hopewell plants) contributed to about 75% of phosphorus in the peak algal biomass (as chlorophyll a) in the water column. Upstream (nonpoint) input provided another 15% of phosphorus in the peak algal biomass. Industrial wastewaters played a very small role in contributing to the algal biomass in the James Estuary. Finally, the Appomattox River which receives wastewater discharges from the City of Petersburg contributed an insignificant amount of phosphorus to the peak algal biomass in the mainstream of the estuary.

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Abstract only

Some Response Patterns of Phytoplankton to Nutrification Based on Enrichment Experiments and Apparent Influence of Silicon

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Nutrient-spike experiments evaluated the effect of various concentrations of N+P and N+P+Si on phytoplankton community structure, speciation, biomass, growth rate and primary production. Seven test levels, including maximal NH_4 doses equivalent to secondary effluent, were used and attenuated in a graded series to simulate a nutrient gradient. Enrichment with NH_4, PO_4 and SiO_2 stimulated carbon production rates, chlorophyll biomass yield and species growth rates. Lag effects, mortality, suppression of growth and production, and differential species responses were commonplace, however. Enrichment did not trigger blooms of nuisance species or algal groups during the 48-hour experiments.

Enrichment with various N+P+Si concentrations tended to be more stimulatory than enrichment with N+P alone. In an experiment designed to evaluate kinetics, a highly significant yield-dose relationship occurred between chlorophyll production over a 48-hour period and initial NH4 concentrations (Y = 5.17X0.62; r2 = 0.94). The relationship between chlorophyll yield and NH_4 uptake was described by Y = 3.33 + 2.06X (r2 = 0.85). There was a significant correlation between NH_4 uptake and its concentration, with the relationship influenced by SiO_2. The presence of SiO_2 in excess of 5 mg-at m^{-3} stimulated NH_4 uptake by about 60% at NH_4 concentrations greater than 12.5 mg-at m^{-3} (r2 = 0.98). In the absence of, or at < 5.0 mg-at m^{-3}) the relationship was described by Y = 0.93X0.91 (r2 = 0.94). Between 5.0 and 18.5 mg-at m^{-3} NH_4, $\geq$ 90% of the NH_4 was taken up over the 48-hr period. Above 18.5 mg-at m^{-3} NH_4, the percentage uptake fell off rapidly, with the percentage utilization influenced by SiO_2 concentration. This apparent influence of SiO_2 on the various responses of natural phytoplankton communities to nitrogen enrichment was an unexpected finding which warrants further study to evaluate its potential role as a regulator and/or mediator of phytoplankton community responses to estuarine nutrification.

The enrichment experiments were carried out using summer phytoplankton communities from Massachusetts Bay in outer Boston Harbor. The results would appear to be relevant to Chesapeake Bay nutrification issues; hence, submission of this abstract.

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The Importance of Submarine Groundwater Discharge to Nutrient Flux in Coastal Marine Environments

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INTRODUCTION

The purpose of this investigation was to examine the role of submarine groundwater discharge (SGWD) on nutrient flux into the nearshore area of two coastal marine environments on Virginia's eastern shore. The movement of water across sediment/water interfaces is important biologically because it sets the microclimatic conditions for sediment inhabiting micro and macro organisms. SGWD, therefore, is a site specific phenomenon. Such movement also contributes to benthic-pelagic coupling (Nixon 1981; Rowe et al. 1975; Rowe and Smith 1977).

Historically, nutrient flux across sediment/water interfaces has been measured by two methods: in situ and diffusion gradient methods (Zeitzschel 1979). This paper reports the movement of water and dissolved material across sediment/water interfaces in marine environments by a third method, that of bulk flow, using seepage meters and mini-piezometers (Lee 1977, 1980; Lee and Cherry 1978).

Initial interest in groundwater discharge into marine habitats focused on the relationship between fresh, groundwater inflow and/or seawater intrusion into fresh, groundwater supplies (Reilly and Goodman 1985). Over a period of several decades, numerous mathematical equations and models have been used to relate the interactions between these two systems and their interface properties. Glover (1959) developed a mathematical expression to describe the interface between fresh and salt groundwater in a coastal aquifer that accounted for the movement and discharge of freshwater. Cooper (1959) developed a hypothesis to explain the mixing zone, or zone of dispersion, the continuous circulation of seawater observed in various field studies, and he attempted to quantify the amount of mixing due to tidal fluctuations. Henry (1959, 1964) advanced the concepts further by using an advection - diffusion equation to account for

hydrodynamic dispersion. Kohout (1960, 1964) was one of the first investigators to quantify and suggest the continuous cycling of seawater as a result of SGWD. The important summary to these developments from an ecological perspective is the realization that fresh, groundwater does move into coastal marine environments and that seawater cycles through the sediments as a result of the SGWD (Fig. 1).

Kohout (1966) and Kohout and Kolipinski (1967) appear to be the first investigators to study the importance of freshwater seepage into shallow marine ecosystems. Their studies, conducted along the shore of Biscayne Bay, Florida, showed a definite relationship between biological zonation and SGWD into the bay. Johannes (1980) also presented significant information on the ecological significance of SGWD. While acknowledging the fact that SGWD to the sea is widespread, "... overlooking the fact could lead to serious misinterpretation of ecological data in studies of coastal pollution, of benthic zonation and productivity, and of the flux of dissolved substances between bottom sediments and overlying water." Harden Jones (1980) has even suggested groundwater seepage as a landmark for identifying spawning grounds for plaice.

In the past, direct measurements of bulk water flow across sediment/water interfaces in marine environments would have been cost prohibitive due to drilling requirements. However, with the development of seepage meters and mini-piezometers by Lee (1977) and Lee and Cherry (1978), such information can be obtained easily and cost effectively. Studies dealing with seepage flux have been estimated in lakes and streams (Lee 1977; Lee and Hynes 1978; Lee and Cherry 1978; Lock and John 1978; Lee 1980; Erickson 1981), estuarys (Valiela et al. 1978; Bokuniewicz 1980; Zimmerman et al. 1985; Capone and Bautista, 1985), and coral reefs (D'Elia et al. 1981; Oberdorfer and Buddemeier 1985, 1986; Lewis 1987; Simmons and Love 1987; Simmons and Netherton 1987; Simmons in press).

METHODS

Two sites were used in this study and both were located on Virginia's eastern shore (Fig. 2) in conjunction with land-based wells established by the U.S. Geological Survey. The first site, located off Chincoteage Island, represented a high density human population area with large volumes of freshwater input to the groundwater system due to human consumption and septic tank usage. Steelman's Landing represented an agricultural site in which SGWD to Magothy Bay was buffered by woodland and marshland areas (Fig. 3) each ~350 m wide at the research site. In general, seepage meters were established ~10-15 cm beneath the water's surface at low tide, and 10m and 100m offshore the low tide mark. Samples of SGWD were collected in acid washed bags that were rinsed in deionized/distilled water. Water samples were then transferred to acid washed plastic bottles, kept on ice, and returned to the laboratory for analyses. Standards for nutrient analyses were made in substitute ocean water. Nitrate was measured by cadmium reduction following E.P.A. Method 353.3 (E.P.A. 1983). Ammonia determinations were made using the phenate method (E.P.A. Method 350.1) (E.P.A. 1983 and American Public Health Assoc. et al. 1976). Total and dissolved phosphate were measured by the ascorbic acid method (E.P.A. Method 365.2) (E.P.A. 1983).

The specific gravity of samples was measured with a hydrometer read to the fourth decimal place. Corresponding density/salinity tables were used to convert hydrometer measurements to salinity values (American Public Health Assoc. et al. 1976). Temperature measurements were made with calibrated long stem thermometers. Oxygen measurements were made using either a macro or micro-winkler techniques and titration with the appropriate normality of thiosulfate.

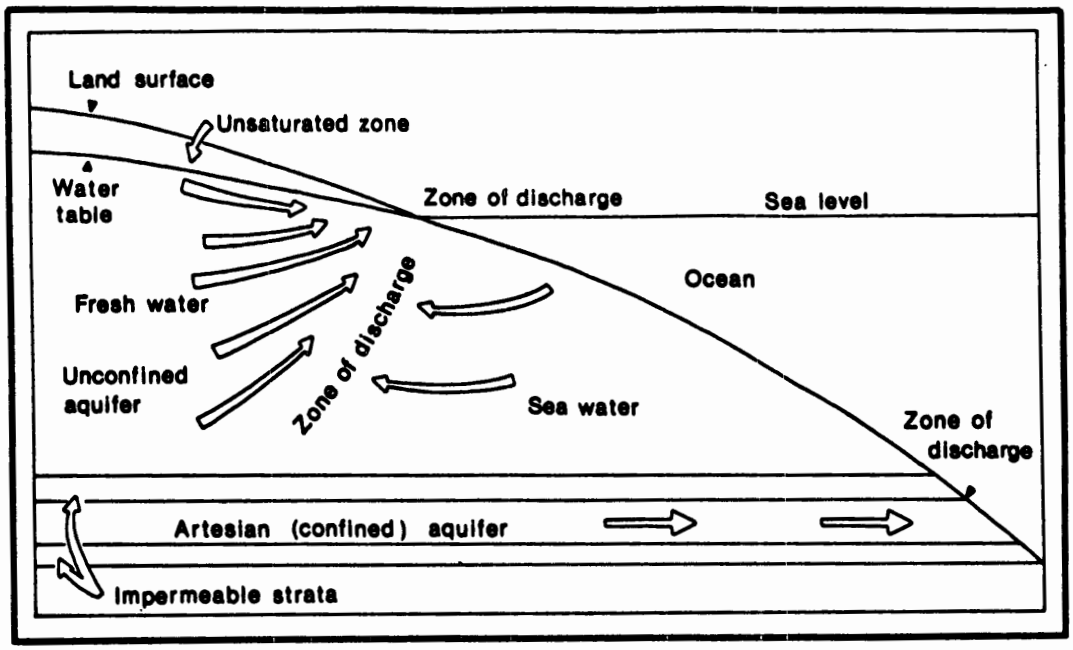

Figure 1. Schematic diagram of the interface between fresh and salt groundwater systems in a coastal environment (from Johannes 1980).

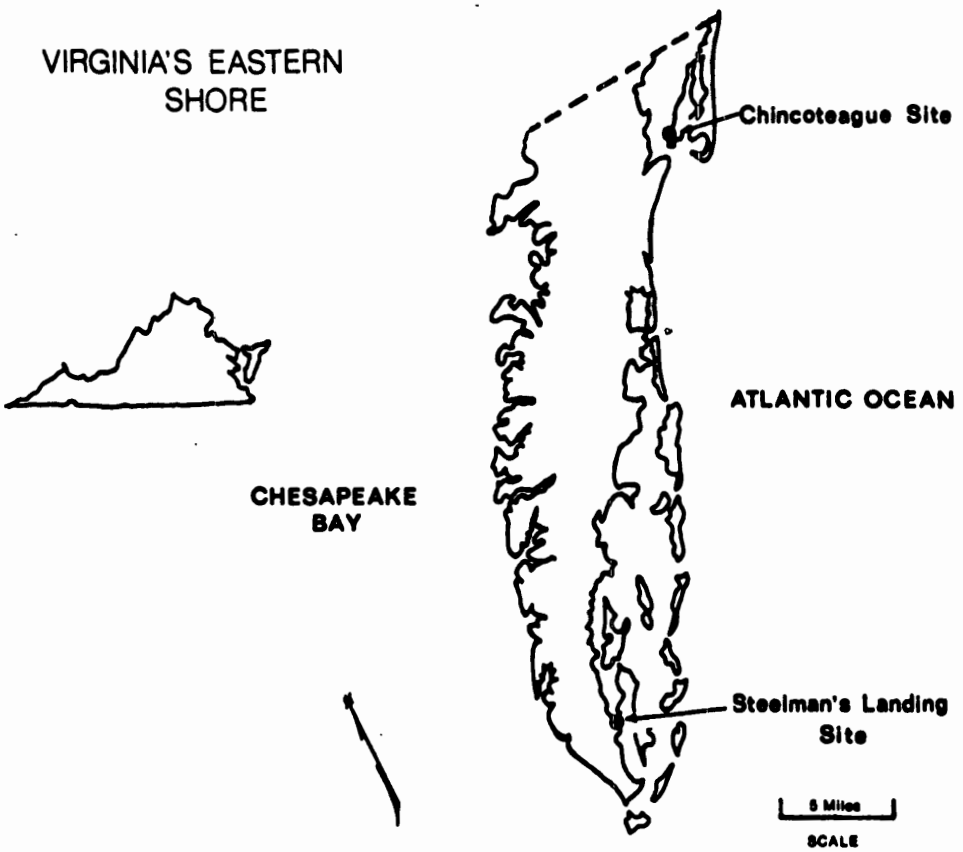

Figure 2. Study sites for submarine groundwater discharge on Virginia's eastern shore.

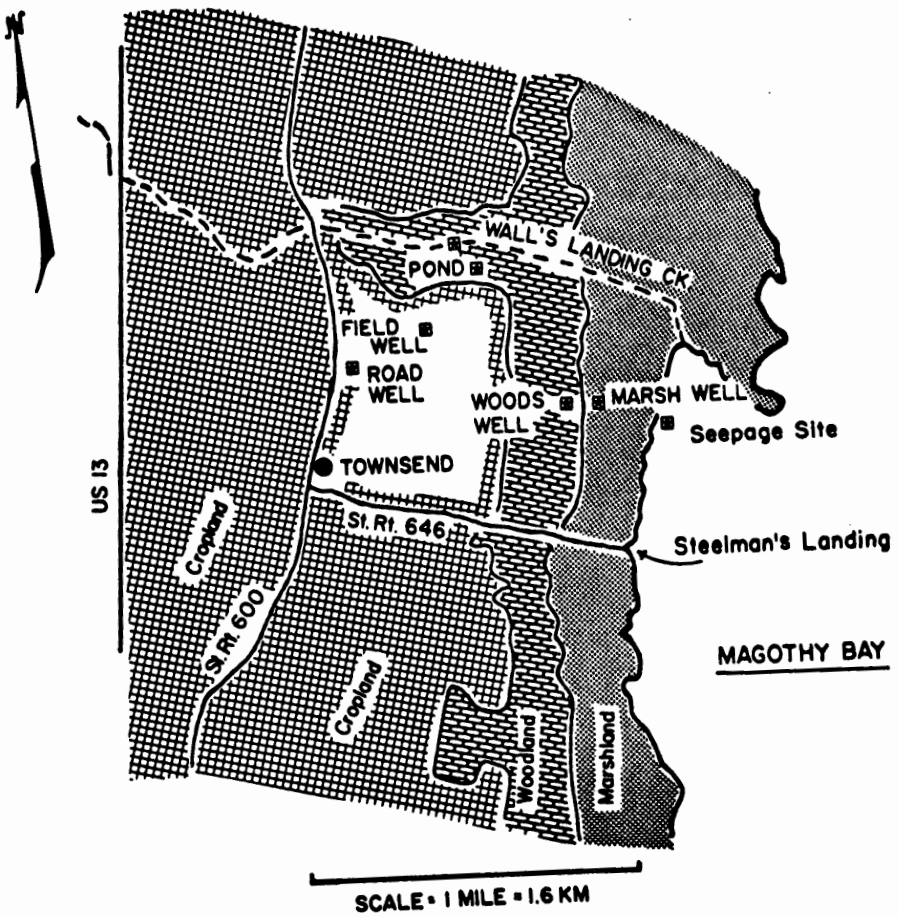

Figure 3. Study site at Steelman's Landing.

The seepage meters used in this study were 0.25 m^2. The flux of SGWD was measured with seepage meters. The volume of water collected was divided by the time interval of collection and multiplied by 4.0 to approximate the amount of water passing across one square meter of bottom. The quantity of SGWD collected was expressed as $L \cdot m^{-2} \cdot day^{-1}$. Knowing the concentration of a particular nutrient and multiplying this by the seepage flux resulted in nutrient flux expressed as $mg \cdot m^{-2} \cdot day^{-1}$.

RESULTS

Seepage meter discharge rates for the two sites are summarized in Table 1 and show that discharge rates were higher at the Chincoteague Site than the Steelman's Landing Site. The SGWD at the Chincoteague Site showed a classical decrease in discharge rate with distance from shore, but the highest discharge at Steelman's Landing varied with distance from shore. Tidal fluctuation off Chincoteague was fairly consistent during the study periods, but off Steelman's Landing, changes in wind direction often left seepage meters exposed during low tide, or inundated to the extent that sample recovery at high tide was not feasible. The inability to keep the shallow meters submerged and to collect at needed time intervals may have contributed to some of the variability at the Steelman's Landing Site.

Salinity measurements from the two sites are summarized in Table 2. Even though all wells sampled from Chincoteague are summarized in Table 2, only one well represented salinity values of groundwater that discharged into the research site. This well had a mean salinity of 1.3 ppt. The salinity of SGWD in both seepage meters and piezometers indicated the presence of fresh, groundwater entering the channel at least to a point 10m offshore the low tide mark. The influence of SGWD off Steelman's Landing was equally apparent in both seepage meters and piezometers. Further evidence of fresh, groundwater seepage entering Magothy Bay was found in water pools above the high tide mark in the marshland area (Fig. 2). Woods Well was located on the edge of the woodland area next to the marshland and had a mean salinity of 2.2 ppt. Marsh Well was located ~85m into the marshland area and contained a mean salinity of 26.7 ppt. Water sampled from pools between the Marsh Well and the Bay contained a salinity of 2.1 ppt. During a field trip in February at the Steelman's Landing site, a small, shallow stream of water (~1cm deep by 5m wide) was observed flowing from the marshland into the bay. The time of collection was almost at mean low tide and the flowing water was believed to be residual Bay water left in the marsh from the previous high tide. The salinity of this water, however, was 4.8 ppt.

Concentrations and flux of ammonia are summarized in Table 3. No nitrate was measured in any of the water samples from the Chincoteague Site. All of the inorganic nitrogen was in the form of ammonia. Wells were anaerobic and the well nearest the study site had a mean ammonia concentration of 74.01 mg/L ($\pm$ 70.0, N = 3). In contrast to the Chincoteague Site, the distribution of nitrate and ammonia was very different at the Steelman's Landing Site. No ammonia was measured in any of the wells, but nitrate was present ($\bar{X} = 11.55 \pm 5.22$; N = 7). Moreover, these well waters were highly oxygenated (8-10 $mg \cdot L^{-1}$). Conversely, nitrate was not found in any of the SGWD samples, but ammonia was present. Table 3 shows that ammonia levels in the SGWD was considerably lower than that measured off Chincoteague, but concentrations were higher than that measured in ambient bay water. In two of the wells sampled, Road and Field Wells, concentrations of nitrate at times exceeded 10 $mg \cdot L^{-1}$. On the February collecting trip, a concentration of 22.91 $mg \cdot L^{-1}$ was measured in the Road Well. High concentrations also were recorded in the pond water and to a certain extent in the stream water ($\bar{X} = 4.68 \pm 3.30$; N = 6). Interestingly, Woods Well and Marsh Well did not show any nitrate presence, but did show ammonia presence on only one occasion. A greater number of piezometers were used off

TABLE 1

SEEPAGE METER DISCHARGE RATES ($L \cdot m^{-2} \cdot day^{-1}$)

Chincoteague Site - 1987				Steelman's Landing Site - 1987-88		
SITE	MONTHS			MONTHS		
SHORE	MAR	APR	MAY	DEC	JAN	FEB
Mean	20.26	28.16	13.59	2.27	5.95	1.16
S.D.	16.53	13.71	14.98	1.02	2.33	1.08
N	6	19	11	5	3	6
MID (~10m offshore)						
Mean	13.23	9.83	8.83	1.47	6.59	2.08
S.D.	22.11	15.21	7.26	0.97	4.11	1.21
N	3	14	5	3	6	8
DEEP (~100m offshore)						
Mean	1.71	1.84			9.33	2.00
S.D.	0.40	0.69			7.68	1.43
N	3	3			9	12

TABLE 2

SALINITY (ppt)

Chincoteague Site

Reference Water

	Wells	Chincoteague Sound
Mean	9.7	28.7
Range	0.4-23.7	25.0-30.3
N	3	3

Submarine Groundwater Discharge

	Shallow		Mid
	SM	Piez.	SM
Mean	21.2	16.7	30.4
Range	8.5-30.1	6.2-27.1	29.4-30.8
N	11	2	4

Steelman's Landing Site

Reference Water

Field Wells, Pond, Creek		Woods Well	Marsh Well	Magothy Bay
Mean	0.7	2.2	26.7	31.5
Range	0.0-1.9	1.6-2.9	21.7-35.4	30.4-32.8
N	12	3	3	7

Submarine Groundwater Discharge

	Shallow		Mid	Deep	
	SM	Piez	SM	Piez	SM
Mean	30.8	30.9	25.5	29.7	31.0
Range	29.8-32.4	25.6-32.7	24.7-27.5	27.2-31.4	29.8-33.5
N	8	12	6	8	9

TABLE 3

Concentration and Flux of Ammonia (NH_3-N)

Chincoteague Site:

Reference Water ($mg \cdot L^{-1}$)

	Wells	Chincoteague Channel
Mean	52.83	None Detected
Range	4.61-150.66	
N	6	

Submarine Groundwater Discharge

	Conc($mg \cdot L^{-1}$)		Flux ($mg \cdot m^{-2} \cdot day^{-1}$)
Shallow			
Mean	61.95	Mean	787.98
Range	2.09-188.83	S.D.	801.68
N	9	N	8
Mid			
Mean	1.70	Mean	14.93
Range	0.31-3.60	S.D.	13.68
N	4	N	4
Deep			
Mean	4.8	Mean	9.10
Range	3.1-6.5	S.D.	—
N	2	N	2

Steelman's Landing Site:

Reference Water

	Wells	Magothy Bay
Mean	None Detected	0.19
Range		0.00-0.42
N	19	5

Submarine Groundwater Discharge

	Conc($mg \cdot L^{-1}$)		Flux ($mg \cdot m^{-2} \cdot day^{-1}$)
Surface			
Mean	0.19	Mean	1.27
Range	0.00-0.44	S.D.	1.47
N	7	N	4
Mid			
Mean	1.32	Mean	15.84
Range	0.00-1.94	S.D.	13.33
N	4	N	3
Deep			
Mean	0.33	Mean	3.24
Range	0.00-0.67	S.D.	3.36
N	5	N	5

Steelman's Landing and they showed a higher concentration of ammonia than the seepage meters. An extreme example occurred in December when the mean amonia concentration from six piezometers, one meter into the sediments, was 4.03 ± 2.16 $mg \cdot L^{-1}$, and no ammonia was detected in the seepage meters even after their pumping for over 24 hrs at a rate of ~1-5 $L \cdot m^{-2} \cdot day^{-1}$.

Total phosphate levels at the Chincoteague Site are summarized in Table 4. The mean concentration of total phosphate in the well closest to the research site showed a mean concentration of 0.34 mg/L (± 0.33; N = 3). Dissolved phosphate was ~80-95% of the total phosphate measured in well and seepage meter samples from Chincoteague. As with the ammonia samples, too few piezometer samples were taken off Chincoteague to be meaningful. However, samples were within the range measured by the seepage meters suggesting little dissolution from the sediments.

The amount of total phosphate measured in well, surface, and seepage water samples off Steelman's Landing was considerably lower than measured at the Chincoteague Site except for the deep site (Table 4). Furthermore, the mean concentration of dissolved and total phosphate in seepage meter samples from the mid and deep water sites was lower than piezometer samples from the same depths (.10 vs .17 $mg \cdot L^{-1}$, respectively). This suggests that 1) dissolved phosphate was being fixed at the sediment/water interface and 2) the presence of the seepage meters did not cause dissolution of phosphate from the sediments. In contrast to the Chincoteague Site, the proportion of dissolved phosphate was ~30% of the total phosphate concentration.

Due to water conditions, only a few hydraulic head readings could be taken which were considered to be reliable. At the Chincoteague Site, mean values of 8.4 and 12.1 cm were measured with a manometer during the March and April field trips. At the Steelman's Landing Site, a hydraulic head of 12.5 and 22.5 cm was measured on the January and February collecting trip, respectively, using water level height from flowing piezometers at low tide.

DISCUSSION

The data collected off Chincoteague and Steelman's Landing show that water below ambient salinity did move into shallow, nearshore marine environments. The discharge values reported here are similar to those reported by Bokuniewicz (1980) and Zimmerman et al. (1985). Moreover, it could be demonstrated that the SGWD was being driven by a positive hydraulic head. Other investigators have expressed caution and concern when using nutrient data obtained from seepage meters (Brock et al. 1982), and such concern between the real and experimental world is valid in this study. However, nutrient concentrations were usually higher in piezometer water than seepage meter water, and dissolved oxygen concentrations taken from ports in the seepage meters indicated an aerobic environment. Therefore, an assumption has been made that, in these cases, there was little or no seepage meter effect.

In comparing nutrient flux data, the problem arises with the units of expression. For comparative purposes, I have converted other reported values to $mg \cdot m^{-2} \cdot day^{-1}$ for the elemental species and tabulated these in Table 5. For those investigators familiar with the original citation values, these numbers also are listed for comparative purposes.

Nutrient transport into marine environments has been reported by other investigators using several methods. A partial summary of some of these studies has been presented by Nixon (1981). In addition, Phoel, et al. (1981) studied nutrient flux in the York River at three different depths. They reported ammonia to be the major inorganic nitrogen species moving into the water column through diffusion flux and their flux rates ranged

TABLE 4

Concentration and Flux of Total Phosphate (PO_4-P)

Chincoteague Site:

Reference Water ($mg \cdot L^{-1}$)

	Wells	Chincoteague Channel
Mean	0.37	0.04
Range	0.01-0.07	0.00-0.07
N	6	11

Submarine Groundwater Discharge

	Conc($mg \cdot L^{-1}$)	Flux ($mg \cdot m^{-2} \cdot day^{-1}$)	
Shallow			
Mean	0.62	Mean	15.82
Range	0.19-1.42	S.D.	17.84
N	12	N	9
Mid			
Mean	0.79	Mean	3.55
Range	0.00-1.62	S.D.	2.98
N	8	N	8
Deep			
Mean	0.06	Mean	0.09
Range	0.05-0.06	S.D.	0.03
N	4	N	4

Steelman's Landing Site:

Reference Water ($mg \cdot L^{-1}$)

	Wells	Magothy Bay
Mean	0.06	0.11
Range	0.02-1.60	0.08-0.12
N	19	5

Submarine Groundwater Discharge

	Conc($mg \cdot L^{-1}$)	Flux ($mg \cdot m^{-2} \cdot day^{-1}$)	
Shallow			
Mean	0.08	Mean	0.25
Range	0.04-0.14	S.D.	0.24
N	9	N	9
Mid			
Mean	0.10	Mean	0.59
Range	0.05-0.19	S.D.	0.66
N	9	N	8
Deep			
Mean	0.10	Mean	0.98
Range	0.05-0.13	S.D.	0.92
N	9	N	8

TABLE 5

Comparative Data — Other Nutrient Flux Studies

STUDY	ORIGINAL CITATION VALUE	CONVERTED VALUES
Callender and Hammond (1982)	0.6-6.5mMole·m^{-2}·day^{-1} NH_3	9.6-88.8 mg·m^{-2}·day^{-1} NH_3-N
	1-21 mMole·m^{-2}·day^{-1} NH_3	14.4-285.6 mg·m^{-2}·day^{-1} NH_3-N
Capone and Bautista (1985)	10μM NO_3	0.12-600 mg·m^{-2}·day^{-1} NO_3-N
Nixon (1981)	13-250μMoles·m^{-2}·hr^{-1} NH_4^+	4.4-84.0mg·m^{-2}·day^{-1} NH_4-N
Phoel, et al. (1981)	3m:22-191 ugatN·m^{-2}·hr^{-1} NH_3	7.2-64.8mg·m^{-2}·day^{-1} NH_3-N
	7-1678 ugatN·m^{-2}·hr^{-1} NH_3	2.4-564.0 mg·m^{-2}·day^{-1} NH_3-N
	9m:162 ugatN·m^{-2}·hr^{-1} NH_3	55.2 mg·m^{-2}·day^{-1} NH_3-N
	16m:511 ugatN·m^{-2}·hr^{-1} NH_3	172.8 mg·m^{-2}·day^{-1} NH_3-N
	288 ugatN·m^{-2}·hr^{-1} NH_3	96.0 mg·m^{-2}·day^{-1} NH_3-N
	11 ugatN·m^{-2}·hr^{-1} NO_3	4.8 mg·m^{-2}·day^{-1} NO_3-N
Callender and Hammond (1982)	.02-0.3mMole·m^{-2}·day^{-1} PO_4-P	.72-9.36mg·m^{-2}·day^{-1} PO_4-P
	0 .1-2.0mMole·m^{-2}·day^{-1} PO_4-P	3.12-62.4mg·m^{-2}·day^{-1} PO_4-P
Nixon (1981)	2-50 μMoles·m^{-2}·hr^{-1} PO_4-P	1.44-37.7mg·m^{-2}·day^{-1} PO_4-P
Zimmerman, et al. (1985)	29-70 x10^{-6}g·m^{-2}·day^{-1} DRP*	.02-.07mg·m^{-2}·day^{-1} PO_4-P
	3-25 x10^{-6}g·m^{-2}·day^{-1} DRP*	.003-.02mg·m^{-2}·day^{-1} PO_4
	7.8 x10^{-3}g·m^{-2}·day^{-1} DRP*	7.92 mg·m^{-2}·day^{-1} PO_4-P

*DRP = Dissolved Reactive Phosphate

between 2.4 - 172.8 $mg \cdot m^{-2} \cdot day^{-1}$. The lowest values occurred in the deepest depths (16m) under stratified conditions, and the highest values occurred in the shallowest depth (3m) when little or no light was present. Other elevated values of ammonia flux have been reported by Callender and Hammond (1982) from Potomac River sediments and they estimated that such rates were approximately one-third of that coming from the Blue Plains sewage treatment plant. Capone and Bautista (1985) noted that ammonia inputs from sediments result from nitrate reduction in anoxic zones and, as such, represent new, rather than recycled, inputs to bay waters.

As expected, there is little information available on nitrate flux into coastal water because it appears the nitrate in fresh, groundwater is converted to ammonia by microbial activity. However, Phoel et al. (1981) and Capone and Bautista (1985) did report nitrate flux in their studies. Phoel et al. (1981) found the lowest values to occur at the 16m depth ($4.8 mg \cdot m^{-2} \cdot day^{-1}$) and the highest values to occur at the 3m depth under maximal light conditions ($564.0 mg \cdot m^{-2} \cdot day^{-1}$). Capone and Bautista (1985) reported a mean concentration of 0.120 mg NO_3-N at the sediment water interface in Great South Bay, New York, and when coupled with the discharge rates of Bokuniewicz (1980), the flux rate should be $\sim 0.12 - 6.00 mg \cdot m^{-2} \cdot day^{-1}$.

Phosphate is another important nutrient species that has been investigated. Again, some of the highest values reported are those by Callender and Hammond (1982) ($285.6 mg \cdot m^{-2} \cdot day^{-1}$). Zimmerman et al. (1985) conducted a study in a nearshore estuarine environment and concluded that > 99% ($7.92 mg \cdot m^{-2} \cdot day^{-1}$) of the dissolved reactive phosphate entering the water was due to groundwater seepage.

Several investigators have attributed nutrient flux to invertebrate irrigation, and there is no doubt these animals play an important role in sediment irrigation and nutrient transport. However, the data to date suggest that SGWD in shallow environments plays the dominating role. In other studies off Key Largo, Florida I have observed seepage meters to become partially anaerobic, benthic invertebrates to succumb to the lowered oxygen levels and expire, and yet the discharge rate remained the same (unpublished data). The relationship between SGWD with and without the presence of invertebrates deserves closer attention, particularly in estuarine environments.

The data reported here show that ammonia flux into the shallow zone off Chincoteague is the highest reported thus far in the literature. Higher flux rates for phosphorus (DRP) have been reported by other investigators (Table 5). The important comparison is between the Chincoteague and Steelman's Landing sites. Chincoteague represents an environment where fresh, groundwater is pumped from the Wallop's Island area for human consumption in Chincoteague (Fennema and Newton 1982). The water, after being used, is discharged into septic tanks and then into the groundwater on the island. The water is then able to make its way directly into the bay. At our study site, there were no woods or marsh buffer and the phosphate and ammonia data collected were correspondingly high. At Steelman's Landing, groundwater beneath cropland showed high nitrate levels (22.0 mg/L), but these high concentrations did not translate into high ammonia (or nitrate) levels in the SGWD. This possibly may have been due to the buffering activity of the plant communities in the woodland and marshland areas. Furthermore, the fact that the concentration of the different nutrient species from the piezometers was generally greater than that collected from the seepage meters suggests that the microbial community at the sediment/water interface plays an important role in converting the nutrients to biomass. Submerged rooted aquatic vegetation would play the same role unless the concentration of nutrients bathing the root system were toxic.

SUMMARY

1. Two sites on Virginia's eastern shore were studied over a short time period with regard to the role of SGWD as a vehicle for nutrient flux. One area represented a site of high density human population with little or no plant buffer of groundwater entering the bay. The other represented an agricultural area buffered from the bay by a woodland and marshland area.

2. The data showed that water of lower salinity, under a positive hydraulic head, did enter the nearshore bay environment with varying concentrations of nutrient species. Off Steelman's Landing, water of low salinity also was found on the surface in the marsh habitat. This suggests that much of the fresh, groundwater may be exiting in the marshland.

3. Higher concentrations of ammonia and phosphate were measured off Chincoteague. The data suggest that more attention should be given to the role of SGWD on nearshore coastal environments when considering the establishment of high density human populations.

4. Even though the total amount of nutrients entering shallow marine environments may be less than that carried in by rivers, the effect of SGWD is site specific at the sediment/water interface. Moreover, SGWD also could be a vehicle for synthetic chemicals and toxic metals.

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REFERENCES CITED

American Public Health Association, American Water Works Association, Water Pollution Control Federation. 1976. Standard methods for the examination of water and wastewater. Fourteenth Edition. American Public Health Association, Washington, D.C. xxxix + 1193 p.

Bokuniewicz, H. 1980. Groundwater seepage into Great South Bay, New York. Estuar. Coast. Mar. Sci. 10:437-444.

Brock, T.D., D.R. Lee, D. Janes, and D. Winek. 1982. Groundwater seepage as a nutrient source to a drainage lake; Lake Mendota, Wisconsin. Water Res. 16:1255-1263.

Callender, E. and D.E. Hammond. 1982. Nutrient exchange across the sediment - water interface in the Potomac River estuary. Estuarine, Coastal and Shelf Sci. 15:395-413.

Capone, D.G. and M.F. Bautista. 1985. A groundwater source of nitrate in nearshore marine sediments. Nature 313:214-216.

Cooper, H.H. 1959. A hypothesis concerning the dynamic balance of freshwater and saltwater in a coastal aquifer. J. Geophys. Res. 64:461-467.

D'Elia, C.F., K.L. Webb, and J.W. Porter. 1981. Nitrate-rich ground water inputs to Discovery Bay, Jamaica: a significant source of N to local reefs? Bull. Mar. Sci. 31:903-910.

Environmental Protection Agency. 1983. Methods for chemical analysis of water and wastes. EPA-600/4-79-020.

Erickson, D.R. 1981. A study of littoral groundwater seepage at Williams Lake, Minnesota using seepage meters and wells. M.S. Thesis, the University of Minnesota. ix + 135 p.

Fennema, R.J. and V.P. Newton. 1982. Ground water resources of the eastern shore of Virginia. Plan. Bull. 3332. Commonwealth of Virginia. State Water Control Board. Tidewater Regional Office. 74p. + Appendices.

Glover, R.E. 1959. The pattern of freshwater flow in a coastal aquifer. J. Geophys. Res. 64:457-459.

Harden Jones, F.R. 1980. The migration of plaice (Pleuronectes platessa) in relation to the environment, p. 383-399. In Bardach, J.E., J.J. Magnuson, R.C. May and J.M. Reinhart (eds.) Fish Behavior and Its Use in the Capture and Culture of Fishes. ICLARM Conf. Proc. 5, 512 p. International Center for Living Aquatic Resources Management, Manila, Philippines.

Henry, H.R. 1959. Salt intrusion into fresh - water aquifers. J. Geophys. Res. 64:1911-1919.

_____. 1964. Effects of dispersion on salt encroachment in coastal aquifers. In: Sea Water in Coastal Aquifers. U.S. Geol. Surv., Water - Supply Pap. 1613-C, pp. 70-84.

Johannes, R.E. 1980. The ecological significance of the submarine discharge of ground water. Mar. Ecol. Prog. Ser. 3:365-373.

Kohout, F.A. 1960. Cyclic flow of salt water in the Biscayne Aquifer of southeastern Florida. J. Geophys. Res. 65:2132-2141.

_____. 1964. The flow of fresh water and salt water in the Biscayne Bay Aquifer of the Miami area, Florida. Seawater in coastal aquifers. U.S. Geol. Surv., Water Supply Pap. 1616-C:12-32.

_____. 1966. Submarine springs: a neglected phenomenon of coastal hydrology. Hydrology and Water Resources Development, Symposium. Central Treaty Org. (CENTO), Ankara, Turkey. Feb. 7-12, 1966. pp. 391-413.

_____ and Kolipinski. 1967. Biological zonation related to ground water discharge along the shore of Biscayne Bay, Miami, Florida, pp. 488-499. In G.H. Lauff (ed.) Estuaries. Conf. on Estuaries, Jekyll Island, GA.

Lee, D.R. 1977. A device for measuring seepage flux in lakes and estuaries. Limnol. Oceanogr. 22:140-147.

_____. 1980. Groundwater - solute influx. Limnol. Oceanogr. 25:183-186.

_____ and J.A. Cherry. 1978. A field exercise on groundwater flow using seepage meters and mini-piezometers. J. Geol. Ed. 27:6-10.

_____ and H.B.N. Hynes. 1978. Identification of groundwater discharge zones in a reach of Hillman Creek in Southern Ontario. Poll. Res. Can. 13:121-133.

Lewis, J.B. 1987. Measurements of groundwater seepage flux onto a coral reef: spatial and temporal variations. Limnol. Oceanogr. 32:1165-1169.

Lock, M.A. and P.J. John. 1978. The measurement of groundwater discharge into a lake by a direct method. Int. Revue ges. Hydrobiol. 63:271-275.

Nixon,S. 1981. Remineralization and nutrient cycling in coastal marine ecosystems, p. 111-138. In B.J. Neilson and L. E. Cronin (ed.) Estuaries and Nutrients. Humana Press, Clifton, N.J.

Oberdorfer, J.A. and R.W. Buddemeier. 1985. Coral reef hydrology. Proc. 5th Int. Coral Reef Symp. 3:307-312.

_____ and _____. 1986. Coral reef hydrology: Field studies of water movement within a barrier reef. Coral Reefs 5:7-12.

Phoel, W.C., K.C. Webb, and C.F. D'Elia. 1981. Inorganic nitrogen regeneration and total oxygen consumption by the sediments at the mouth of the York River, Virginia, p. 607-618. In B.J. Neilson and L.E. Cronin (ed.) Estuaries and Nutrients. Humana Press, Clifton, N.J.

Reilly, T. E. and A.S. Goodman. 1985. Quantitative analysis of saltwater - freshwater relationships in groundwater systems - a historical perspective. J. Hydrol. 80: 125-160.

Rowe, G.T. and K.L. Smith, Jr. 1977. Benthic - pelagic coupling in the mid - atlantic bight, p. 55-66. In B.C. Coull (ed.) Ecology of Marine Benthos, Univ. of South Carolina Press, Columbia, S.C.

_____, C.H. Clifford, and K.L. Smith, Jr. 1975. Benthic nutrient regeneration and its coupling to primary productivity in coastal waters. Nature 255:215-217.

Simmons, G.M., Jr. Groundwater discharge quality in a deep coral reef habitat. NOAA Technical Rept. Ser., Off. Ocean and Coastal Resource Mgt., Sanctuary Programs Div. (In Press).

_____ and J. Netherton. 1987. Groundwater discharge in a deep coral reef habitat - evidence for a new biogeochemical cycle?, pp. 1-12. In C.T. Mitchell (ed.) Diving for Science - 86. Proc. Amer. Assoc. Underwater Sci. Amer. Acad. Underwater Sciences, 947 Newhall St., Costa Mesa, CA 92627.

_____ and F.G. Love. 1987. Water quality of newly discovered groundwater discharge into a deep coral reef habitat, pp. 155 - 164. In R.A. Cooper and A.N. Shepard (eds), Science Applications of Current Diving Technology on the U.S. Continental Shelf. NOAA Symp. Ser. Undersea Res. 2(2):7-16. NOAA Undersea Research Program, Rockville, MD.

Valiela, I., J.M. Teal, S. Volkmann, D. Shafer, E.J. Carpenter. 1978. Nutrient and particulate fluxes in a salt marsh ecosystem: Tidal exchanges and inputs by precipitation and groundwater. Limnol. Oceanogr. 23:798-812.

Zeitzschel, B. 1979. Sediment - water interaction in nutrient dynamics, p. 195-218. In K.R. Tenore and B.C. Coull (ed.) Marine Benthic Dynamics. Univ. of South Carolina Press, Columbia, S.C.

Zimmerman, C.F., J.R. Montgomery, and P.R. Carlson. 1985. Variability of dissolved reactive phosphate flux rates in nearshore estuarine sediments: effects of groundwater flow. Estuaries. 8:228-236.

Understanding the Estuary: Advances in Chesapeake Bay Research. Proceedings of a Conference. 29-31 March 1988. Baltimore, Maryland. Chesapeake Research Consortium Publication 129. CBP/TRS 24/88.

Abstract only

Nitrogen Cycling in Chesapeake Bay Sediments: Balance Between Regeneration and Denitrification

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Nitrogen transformation and recycling processes were investigated at two stations (10 m, 25 m) in the mesohaline region of Chesapeake Bay in relation to seasonal patterns of phytoplankton production and bottom water 0_2 depletion. Intensive sampling at 1-2 wk intervals was done in spring 1986, and additional measurements were made during spring, summer and mid autumn of 1986-1987. Fluxes of DIN across the sediment-water interface were measured with core incubations, and nitrification (NI) and denitrification (DN) were estimated using specific inhibitors in intact cores. Annual cycles of DIN (NH_4 plus NO_3^-) flux from sediments to over-lying water followed the seasonal temperature cycle, with summer rates ca. 5-fold greater than those in early spring. A mid-spring peak in DIN flux corresponded to an earlier event of particulate organic nitrogen (PON) deposition, with a temporal lag related to ambient temperature. Pools of NH_4^+ in sediment pore-waters increased by more than an order-of-magnitude from Apr. to Aug. A decline in macrofauna and associated bioturbation during this period may have contributed to the NH_4^+ accumulation, which was quantitatively equivalent to ca. 10% of the mean DIN flux across the sediment-water interface. Seasonal patterns of NI and DN were opposite that observed for DIN flux, with highest rates in spring and fall. NI rates, which were correlated with sediment redox, approached zero during incipient hypoxia (< 1 mg 0_2 L^{-1}) of bottom waters in spring. Rates of both NI and DN were low in summer even at the shallower station which experienced hypoxia only rarely. It is likely that the reduced depth of 0_2 penetration into sediments and accumulation of toxic anaerobic metabolites (e.g., sulfide) both contributed to loss of NI and DN. Prelininary budgets of N inputs and outputs to and from the sediment surface revealed that in spring inputs of PON exceeded outputs of both NH_4 and N_2, which fluxes were themselves similar to one another. By Aug., recycling flux of NH_4 to overlying water was greater

than the input of PON, and DN was virtually eliminated, especially at the deeper (anoxic) station. The importance of NH_4^+ recycling in support of phytoplankton production is demonstrated, as is the effect of DN in reducing NH_4^+ availability for algal growth. It is suggested that any increase in bottom water O_2 concentration in summer (through, for example, reduced eutrophication) would have the effect of enhancing DN and further decreasing NH_4^+ availability for phytoplankton.

Understanding the Estuary: Advances in Chesapeake Bay Research. Proceedings of a Conference. 29-31 March 1988. Baltimore, Maryland. Chesapeake Research Consortium Publication 129. CBP/TRS 24/88.

Variability of Groundwater Nitrate Concentrations in Non-Agricultural Ecosystems

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ABSTRACT

Results from several recent watershed-scale studies suggest that removal of fertilizer NO_3^- from laterally moving, shallow groundwater occurs in forest and marsh lands adjacent to agricultural systems, thus mitigating the impact of NO_3^- pollution of both ground- and surface waters by agriculture. To date, the precise mechanisms responsible for this observation remain unclear. This study was initiated to investigate the removal of groundwater NO_3^- in non-agricultural ecosystems and to study the mechanisms controlling this process. A site was located on Maryland's Eastern Shore, and contained an established agricultural field, abutted by a grass buffer strip, a forest, and a marsh, all adjacent to the Wye River. Each ecosystem was sampled with groundwater wells located along transects which followed the downgrade of the surface topography, with the agricultural field at the highest elevation and the marsh at the lowest elevation. Nitrate analyses of these wells indicated a general trend of decreased groundwater NO_3^- concentration under the non-agricultural ecosystem. However, in 3 of the 8 wells located in the forest decreased NO_3^- concentrations were not observed. Predicted NO_3^- concentrations in these wells based on Cl^- concentrations indicate that, even in a relatively

homogeneous sandy aquifer, preferential flow paths of groundwater may exist, which contribute to the variability observed. Therefore, caution must be used in defining the role of non-agricultural ecosystems in mitigating NO3- of groundwater when data are based on only a few well transects.

INTRODUCTION

The fate of nitrogen fertilizers applied to soil is an area of primary importance in agricultural research, and there is an increasing concern over the impact of nutrients derived from agricultural systems on surface and groundwater quality. Nitrates from agriculture have been implicated in the declining quality of the Chesapeake Bay (U.S. Environ. Protection Agency, 1983). These conclusions have come primarily from surface runoff studies of heavily manured agricultural lands. Less attention has been given to investigating the impact of groundwater nitrate loading to the Chesapeake Bay from adjoining cropland.

Several recent studies have shown that non-agricultural ecosystems can act as filters in removing NO_3^- from shallow groundwater draining agricultural lands. Nitrate movement from agricultural fields into forested land was monitored along nine well transects in a 1568 ha. watershed by Lowrance, et al. (1984). Seasonal mean concentrations of NO_3-N were always found to be significantly higher in field wells than in the non-agricultural areas. No conclusive evidence was given regarding the process by which the riparian forest reduced the NO_3^- levels, however, decreasing ratios of NO_3-N:Cl from the field to the streams indicated that biological processes (plant uptake and denitrification) were important in the removal of NO_3-N.

Along two groundwater well transects from a corn field into a forested area, Peterjohn, et al. (1984) found NO_3^- concentrations of shallow groundwater inside the forest to be about an order of magnitude lower than in the corn field. These NO_3^- losses were estimated to be about 1/3 uptake by forest vegetation and 2/3 by denitrification, however, direct measurements of denitrification were not done.

In another study Jacobs and Gilliam (1985) measured NO_3-N concentrations of 7 to 8 mg/L in the subsurface drainage water at the edge of a forested area. Apparent passage through about 16 to 47 m of the forested area resulted in a drop to <0.1 mg/L NO_3-N. It was speculated that denitrification was the primary mechanism of NO_3^- removal, rather than N uptake by vegetation or dilution by deep seepage water.

Other studies have also reported similar findings of decreased groundwater NO_3^- levels in riparian zones (Schnabel, 1986; Cooper, 1986; Davidson et al., 1986). Despite the general similarity of results of these previous studies, the precise mechanisms have not been clearly determined. Possible mechanisms responsible for these observations may include: i) dilution of shallow, high NO_3^- groundwater by upwelling of deeper, low NO_3^- groundwater, ii) dilution by low NO_3^- recharge percolate in the forest area, iii) microbial denitrification in the shallow groundwater, and iv) plant uptake of NO_3^-. The objective of this report is to describe the extent of variability of groundwater NO_3^- concentrations in non-agricultural ecosystems bordering agricultural land. Our data are preliminary but are important because of their implications on future sampling strategies of this and other similar studies.

MATERIAL AND METHODS

The study site is located adjacent to the Wye River in the Eastern Shore region of Maryland at the Wye Research and Education Center near Queenstown, Md. The agricultural soil is a well-drained Matapeake silt loam (fine-silty, mixed, mesic, Typic Hapludult) underlaid by sand. The site consists of an agricultural field, abutted by a narrow grass buffer strip, forested land and a marsh which is located on the river's edge. The agricultural land had been cropped to corn in 1984 (moldboard plow tillage) and has been in soybeans since 1985. The grass buffer strip is used as an access road, and the forest ecosystem is a small patch of deciduous woods which is bordered by pine trees on the eastern and western edge and a Phragmities marsh to the south (Fig. 1). In 1985 a preliminary investigation of the site was initiated. Piezometers were installed in each of the 4 ecosystems (field, grass, forest, and marsh) at the top of the water table (located ca. 0.5 to 1.5 m below the soil surface). These piezometers consisted of ceramic candles attached to 1.5 m lengths of 3/4" polyvinyl chloride (pvc) pipe. Groundwater samples were collected for NO_3^- analyses by drawing a vacuum on the pvc pipe and subsequently collecting groundwater which accumulated in the pipe with a tube attached to a syringe. In 1987 a network of groundwater wells was installed at the site. These wells were pvc pipe (3/4" diameter, screened 60-120 cm) which were installed below the surface of the water table. Water samples were collected from the wells in the summer and fall of 1987 and in the winter of 1988 for NO_3^--N and Cl^- analyses. Groundwater heights were also determined at these sampling times. Groundwater NO_3-N (NO_3 + NO_2) levels were determined by an automated cadmium reduction method (Technicon Autoanalyzer II, Industrial Method # 100-70W, 1973). Chloride was determined using the colormetric ferricyanide procedure (Am. Pub. Health Assoc., 1985).

RESULTS AND DISCUSSION

From preliminary sampling of the study site, conducted in 1985, it was observed that at the edge of the agricultural field groundwater NO_3-N concentrations were approximately 13 mg-N/L, and concentrations decreased markedly in the adjoining grass buffer strip where levels of ca. 6 mg-N/L were observed. The water below the forested area contained very low NO_3^- levels (ca 0.5 mg-N/L), and groundwater NO_3^- concentrations below the marsh were undetectable. These results corroborate earlier work which indicates that non-agricultural ecosystems may act as buffers to remove agriculturally derived NO_3^- in shallow, laterally moving groundwater (Jacobs and Gilliam, 1985; Lowrance et al., 1984; Peterjohn, 1984).

Based on these preliminary results, the site was extensively instrumented with groundwater wells in 1987. Wells were established in a pattern which roughly followed the surface topography from the field, grass, forest, and marsh (Fig. 1).

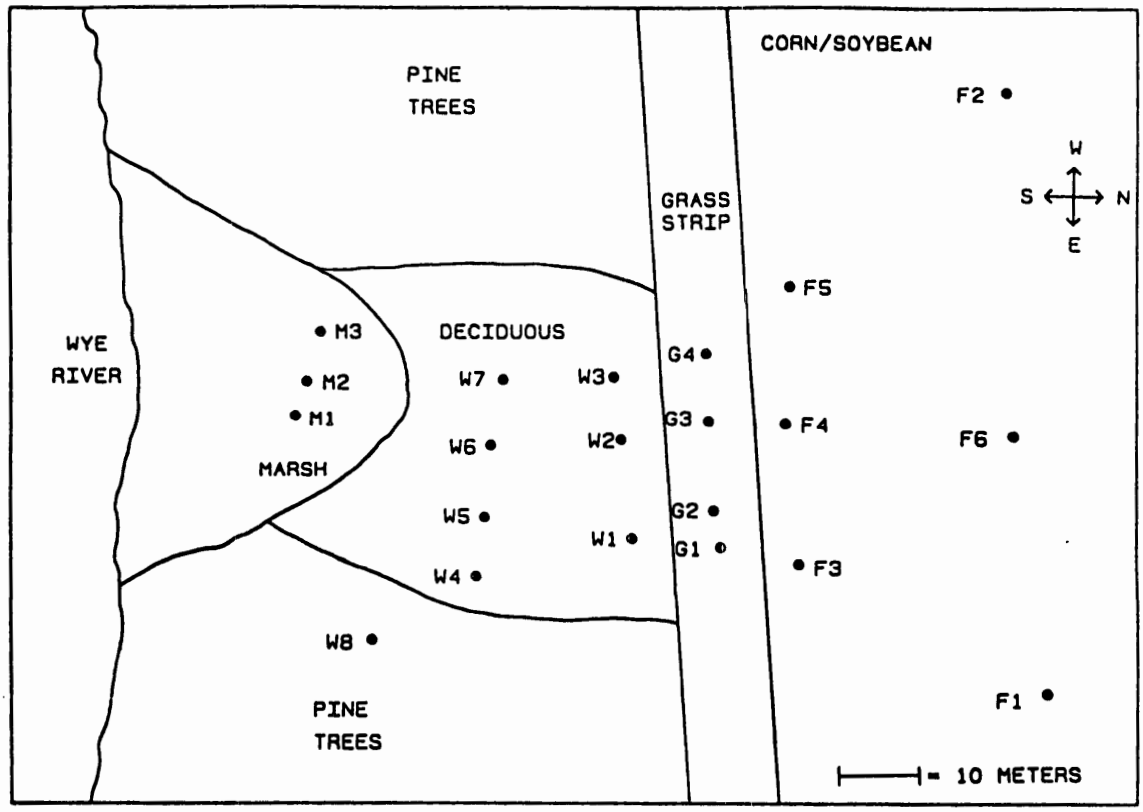

Figure 1. Map of study area indicating groundwater well locations.

The general slope of the land surface topography along with water table heights at different times of the year are shown in Fig. 2. Groundwater gradients were found to follow the general direction of the surface

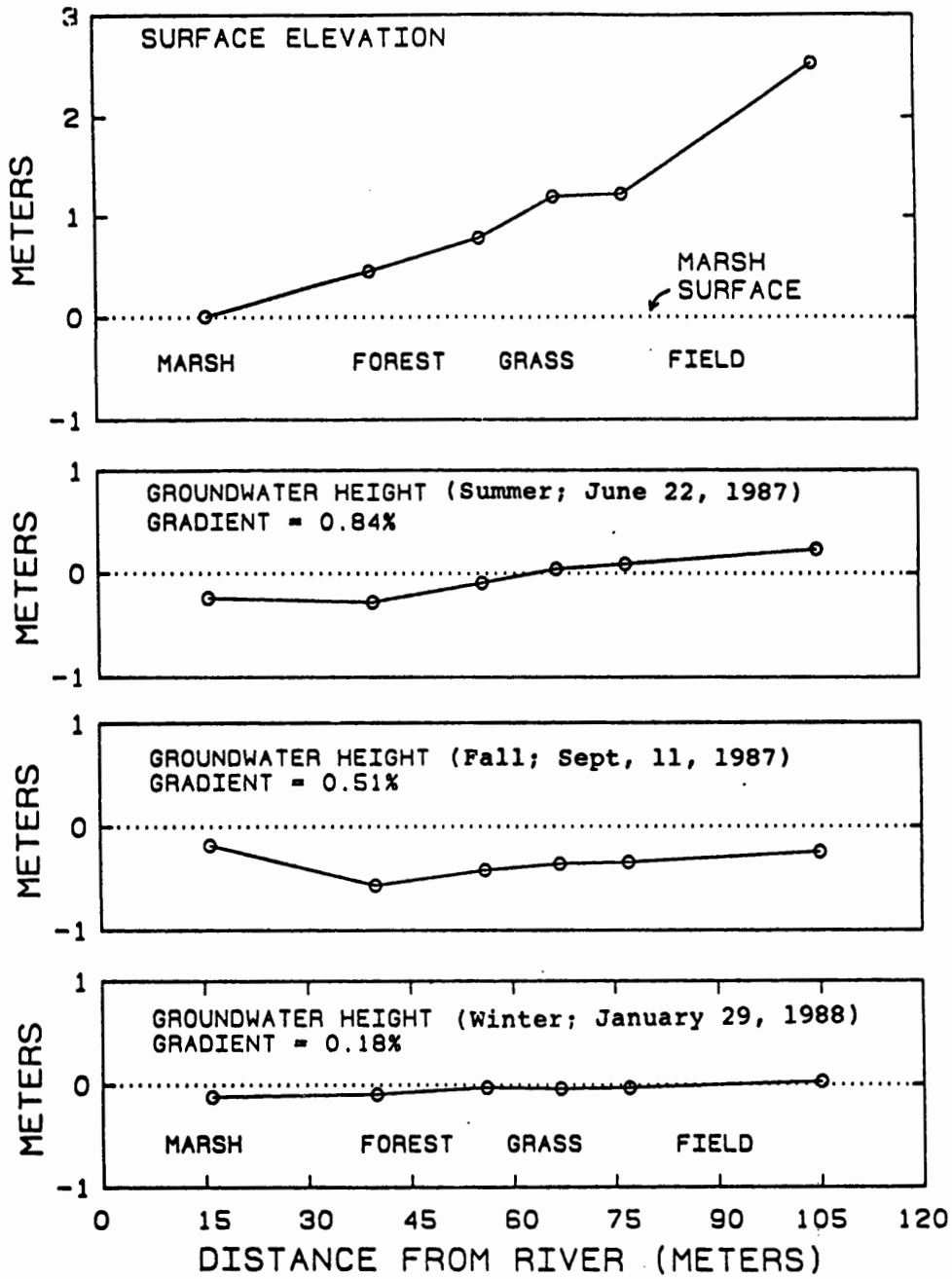

Figure 2. Surface elevation of land topography and seasonal water table heights relative to the height of the marsh.

topography, however, neither water table heights nor gradients were constant over the year. In the summer, groundwater gradients were greatest (0.8 %) and a low point in the water table was observed in the second series of wells located in the deciduous forest (wells W5, W6, and W7). During the Fall, groundwater levels in all wells (except the marsh wells) dropped, and the low water table zone in the forest wells became more pronounced. In the winter, recharge water caused the water levels in the wells to rise and the gradient from

field to forest decreased to 0.18%. It is hypothesized that the low forest water table zone observed during the summer and fall resulted from low rainfall (1987 precipitation was 30 cm less than the 30 year average) combined with high transpiration rates by the deciduous trees. Also, in the forested area, laterally moving groundwater may be coming from both the agricultural lands and the river.

Chloride concentrations in the wells support this hypothesis (Fig. 3). In the center and west edge of the sample site Cl^- levels in the forest wells ranged from 600 to 1000 mg Cl^-/L which are approximately 20 to 40 fold higher than Cl- concentrations in the field (average concentration 26 mg/L). However, along the east edge of the site, Cl- concentrations in the forest wells were similar to those observed in the agricultural field. Chloride concentrations in the river were the same as those of the marsh wells.

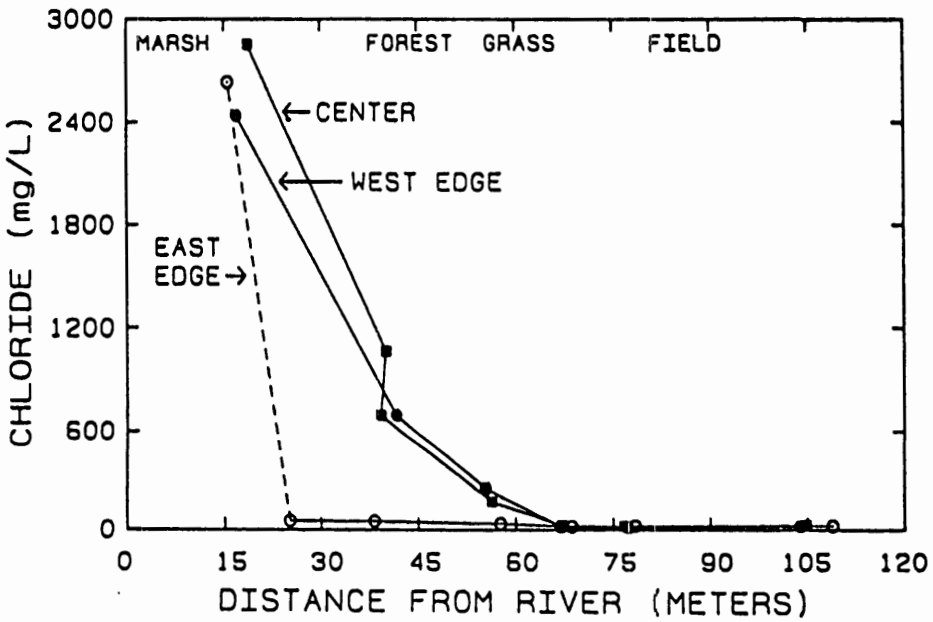

Figure 3. Chloride concentrations of groundwater along the East edge, West edge, and center of the study site.

Average NO_3^- concentrations in the shallow groundwater below the agricultural field are presented in Table 1. In this area NO_3^- exhibited a higher degree of spatial variation than temporal variation. Coefficients of variation in this area ranged from 34 to 46% on any given sample date, but variability over time for a given well was low (CVs 8-19%).

Table 1. Nitrate concentrations of individual agricultural field wells for each sampling date.

Well[a]	Sampling date 6/22/87	9/11/87	10/25/87	1/29/88	Mean	%CV[b]
			NO_3^- mg-N/L			
F1	8.6	7.9	7.8	7.0	7.8	8.5
F2	9.5	8.6	9.8	8.2	9.0	8.3
F3	3.5	5.4	4.7	4.3	4.5	18
F4	3.1	3.2	3.0	3.3	6.5	19
F5	8.5	8.3	8.1	10.8	8.9	15
F6	4.5	dry	dry	4.0	4.3	-
Mean	6.3	6.7	6.7	6.3		
%CV	46	34	42	46		

[a]Well designations refer to those indicated on Fig 1.
[b]Coefficient of variation (%).

Variability of NO_3^- in the grass strip was lower than the field and was relatively constant both temporally and spatially with CVs in the range of 25% (Table 2).

Table 2. Nitrate concentrations of individual grass strip wells for each sampling date.

Well[a]	Sampling date 6/22/87	9/11/87	10/25/87	1/29/88	Mean	%CV[b]
			NO_3^- mg-N/L			
G1	3.7	4.5	5.1	2.7	4.0	25
G2	4.0	5.1	3.7	3.1	4.0	21
G3	5.5	5.0	2.7	3.7	4.2	31
G4	3.1	3.9	4.0	2.5	3.4	21
Mean	4.2	4.6	3.9	3.0		
%CV	24	12	25	17		

[a]Well designations refer to those indicated on Fig 1.
[b]Coefficient of variation (%).

Groundwater NO_3^- concentrations in the forest showed a higher degree of spatial variability (Table 3). Coefficients of variation for the two Fall sample dates were ca. 20%, however in the summer and winter, CVs exceeded 100%. The reason for this appears to be a spatial heterogeneity in the forest area. The wells in the center and along the west edge of the deciduous forest area were dry during the fall (wells W2, W3, W5, W6, W7). In the summer and winter, when water was present, these wells typically contained low NO_3^- levels. However, the wells located on the east edge of the forest area (wells W1, W4, W8) contained water on all sample dates and supported nitrate levels similar to those observed in the agricultural field.

Groundwater NO_3^- in the marsh were undetectable on all sample dates (detection limit 0.1 mg-N/L).

Table 3. Nitrate concentrations of individual forest wells for each sampling date.

Well[a]	Sampling date 6/22/87	9/11/87	10/25/87	1/29/88	Mean	%CV[b]
			NO_3^- mg-N/L			
W1	5.1	8.6	9.5	7.1	7.6	24
W2	0.3	dry	dry	0.2	0.3	-
W3	1.2	dry	dry	0.1	0.7	-
W4	8.2	6.2	5.5	6.1	6.5	19
W5	0.2	dry	dry	3.7	0.3	-
W6	0.2	dry	dry	0.7	0.5	-
W7	0.2	dry	dry	0.3	0.3	-
W8	6.0	6.0	6.5	5.8	6.10	4.9
Mean	2.7	6.9	7.2	3.0		
%CV	120	20	29	100		

[a]Well designations refer to those indicated on Fig 1.
[b]Coefficient of variation (%).

Groundwater NO_3-N concentrations at the site are inversely correlated to chloride levels (Fig. 4). This correlation indicates that water from the river may be entering the groundwater wells located in the forest, thus, diluting the NO_3^-.

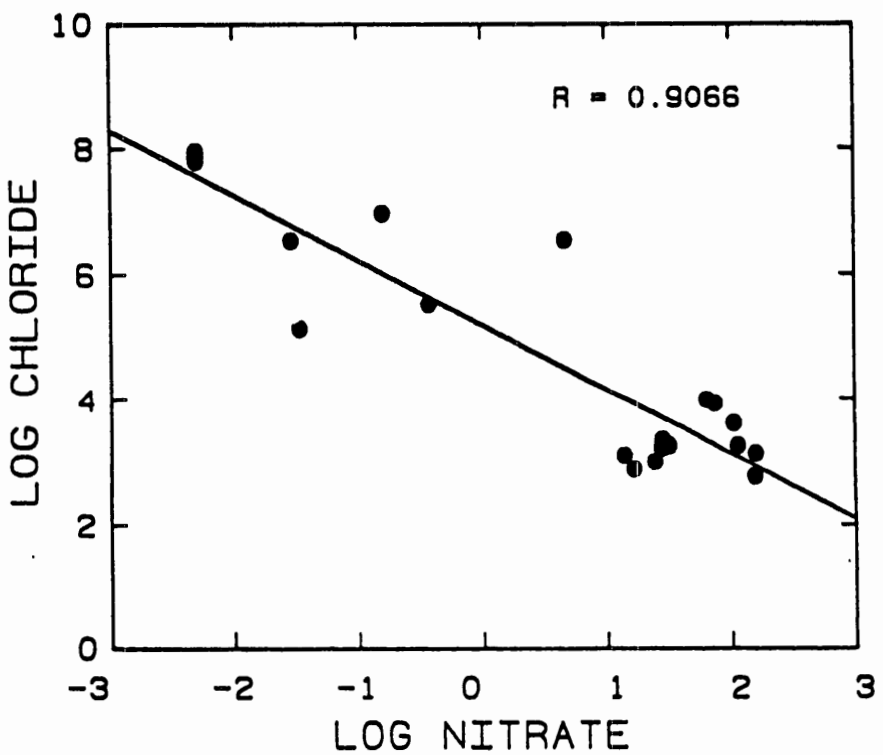

Figure 4. Correlation between NO_3^- and Cl^- concentrations for individual wells in the study site. Regression equation: $\log(Cl^-) = -1.03 * \log(NO_3^-) + 5.01$.

To investigate the magnitude of this dilution effect, predicted NO_3^--N concentrations were calculated for the wells based on the average Cl^- concentration of

each well and the NO_3-N concentrations observed in the agricultural field (Fig. 5). This calculation corrects the NO_3-N concentrations in the wells for dilution by river water.

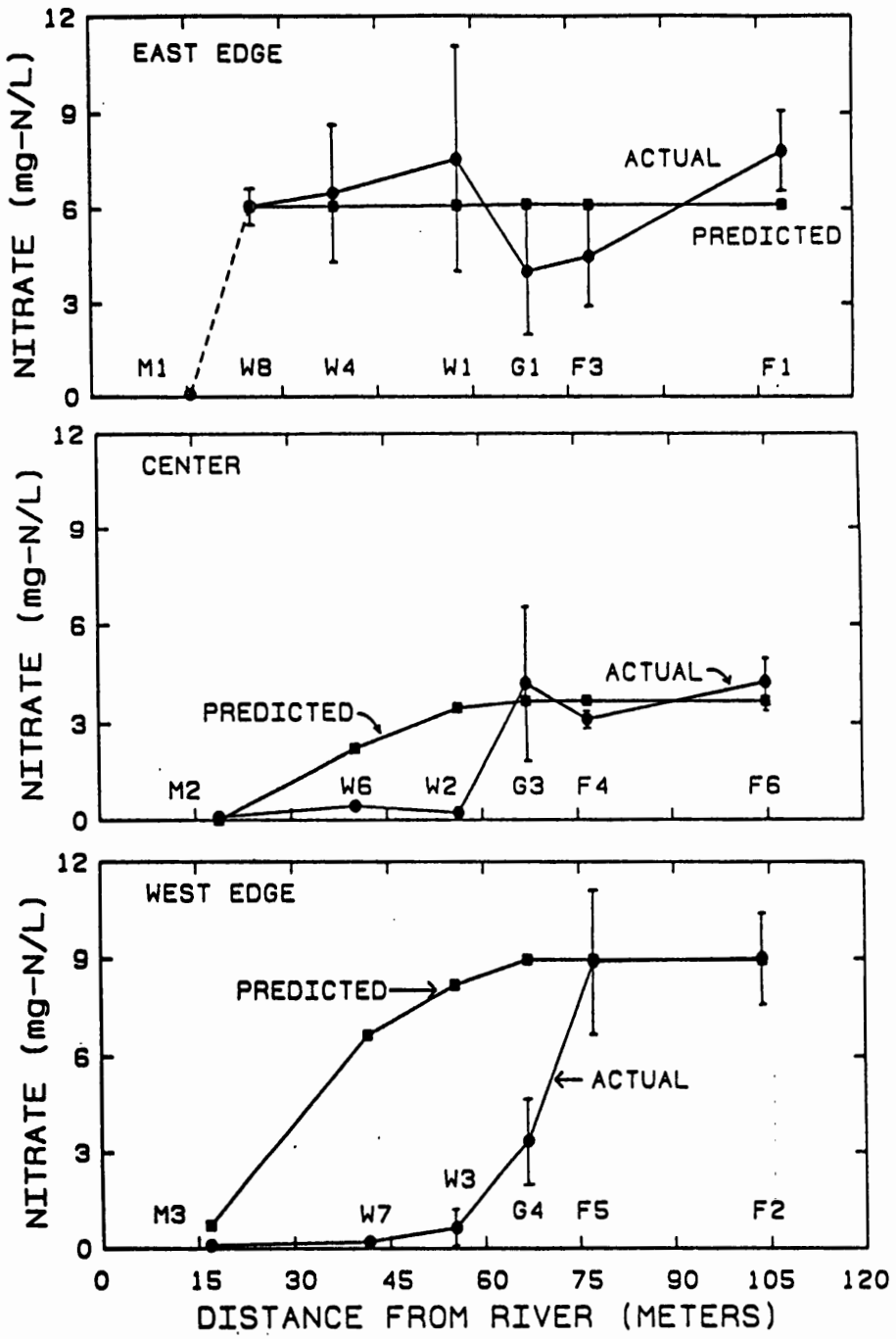

Figure 5. **Measured (circles) and predicted (squares) NO_3^- concentrations along the Eastern edge, Western edge, and in the center of the study site. Vertical bars are 95% confidence intervals about the measured average NO_3^- concentrations. Sample wells indicated correspond to locations designated in Fig. 1.**

Along the east edge of the site, predicted NO_3-N concentrations roughly corresponded with measured NO_3-N concentrations, indicating that nitrate is essentially unaltered as it moves from the field through the forest. Implicit in this interpretation is the assumption that the forest wells are actually being fed by groundwater which contains NO_3^- levels similar to those observed in field wells F1 and F3. An expanded well network is needed to confirm or refute this assumption. In the center and west edge of the field a discrepancy exists between actual and predicted NO_3^- concentrations. Since dilution of agriculturally derived NO_3^- by salt water intrusion from the river has been accounted for, the difference between actual and predicted NO_3^- concentrations are due to either : i) denitrification, ii) plant uptake, or iii) dilution by low NO_3^- recharge. Studies are currently underway to differentiate the relative importance of these mechanisms.

SUMMARY

Along the west edge and center of our study site we observed decreased groundwater NO_3-N concentrations in the grass and forest ecosystems. This observation is similar to the pattern of decreased NO_3-N concentrations in non-agricultural ecosystems reported in previous studies. However, along the east edge of the site, NO_3-N concentrations were similar in the field, grass, and forested areas, indicating that non-agricultural ecosystems had little effect on groundwater NO_3-N levels. The reason for this discrepancy is unclear but is is possible that: i) along the east edge of our site, the forest wells are being fed with groundwater which is higher in NO_3^- than that present in the agricultural field or ii) nutrient cycling processes are different in the pine forest. In any case it is evident that our system is not as simple as indicated by previous studies reported in the literature. Past studies may have may have relied on inadequate sampling or, on the other hand, a real difference could exist between our site and others. Before any firm conclusions can be drawn regarding the mechanisms of NO_3^- removal by the non-agricultural ecosystems at our site, the reasons for the spatial heterogeneity of groundwater NO_3^- in the forest ecosystem must be elucidated. This will require a more detailed hydrological description of the site.

ACKNOWLEDGEMENTS

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BIBLIOGRAPHY

American Public Health Association. 1985. Standard methods for the examination of water and wastewater. 16 ed.

Cooper, J.R., J.W. Gilliam, and T.C. Jacobs. 1986. Riparian areas as a control of nonpoint pollutants. In David L. Correll (ed.) Watershed Research Perspectives. Smithsonian Inst. Press. Wash., D.C. pp. 166-192.

Davidson, E.A., and W.T. Swank. 1986. Environmental Parameters regulating gaseous nitrogen losses from two forested ecosystems via nitrification and denitrification. Appl. & Env. Microb. 52:1287-1292.

Jacobs, T.C., and J.W. Gilliam. 1985. Riparian losses of nitrate from agricultural drainage waters. J. Env. Qual. 14:472-478.

Lowrance, R. Richard, Robert L. Todd, and Loris E. Asmussen. 1984. Nutrient cycling in an agricultural watershed: I. Phreatic Movement. J. Env. Qual. 13:22-27.

Peterjohn, Williams., T., and David L. Correll. 1984. Nutrient dynamics in an agricultural watershed: Observations on the role of a riparian forest. Ecology 1466-1475.

Schnabel, R.R. 1986. Nitrate concentrations in a small stream as affected by chemical and hydrologic interactions in the riparian zone. In David L. Correll (ed.) Watershed Research Perspectives. Smithsonian Inst. Press. Wash., D.C. pp. 263-282.

U.S. Environ. Protection Agency, 1983. Chesapeake Bay Program technical studies: A Synthesis. U.S. Government Printing Office, 606-490, pp. 634

Understanding the Estuary: Advances in Chesapeake Bay Research. Proceedings of a Conference. 29-31 March 1988. Baltimore, Maryland. Chesapeake Research Consortium Publication 129. CBP/TRS 24/88.

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 between 1984 and 1985 was due to increased sediment oxygen demand (SOD), phytoplankton resperation, and bacterial oxidation.
- Bottom sediments were the largest source of dissolved inorganic phosphorus (DIP) and ammonia nitrogen during the summers of 1984 and 1985 (the simulated time period).
- Bay DO and algae are controlled largely by SOD and sediment nutrient flux.

- 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

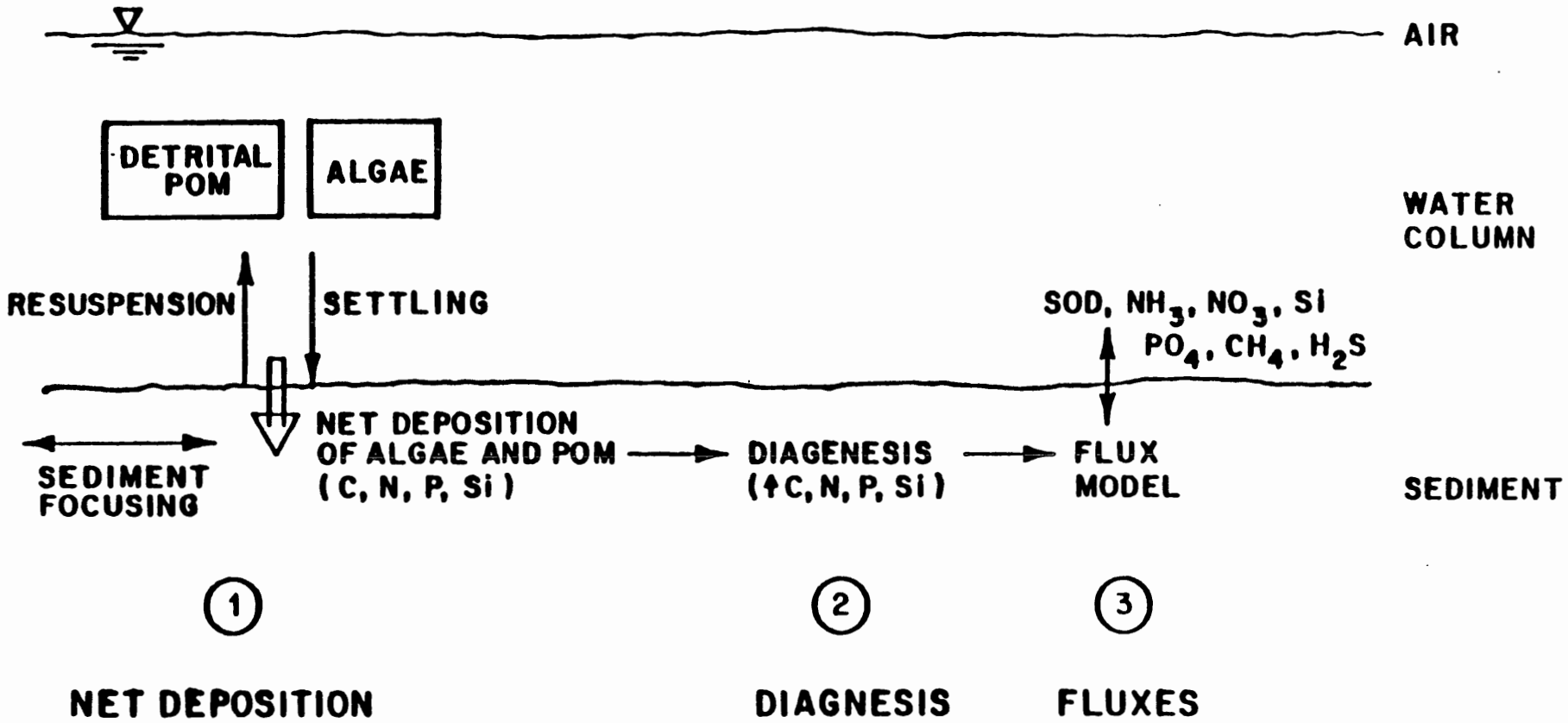


Figure 1. Principal Components of the Sediment Model (From HYDROQUAL, 1988)

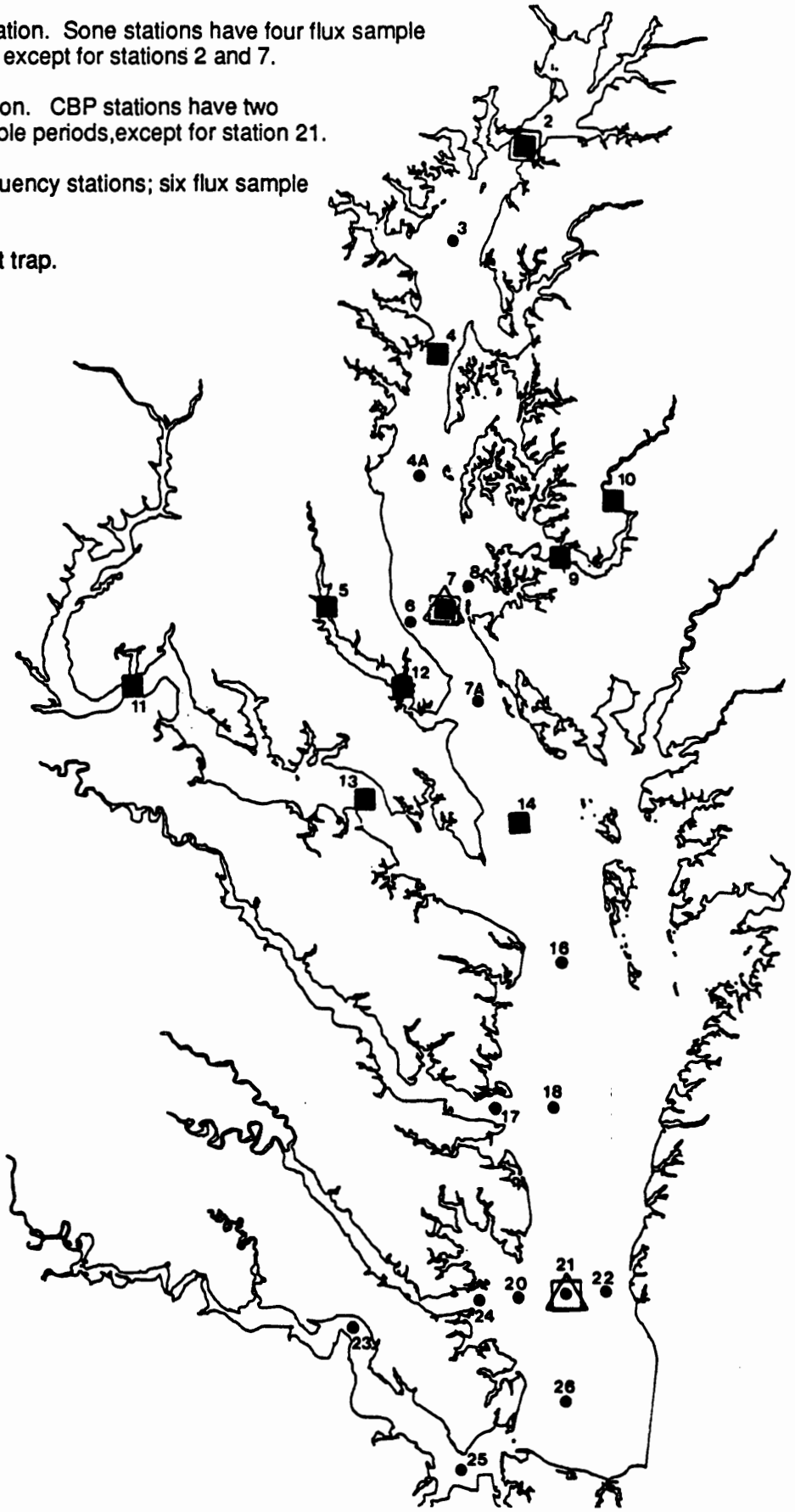
Figure 2. Location of sediment monitoring stations.

■ = 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.



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

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

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.

REFERENCES

Boynton, W.R.; Kemp, W.M.; Garber J.; Barnes, J.M. Sediment and water column interaction in the Chesapeake Bay. In: State of the Chesapeake Bay Second Annual Monitoring Report. 1987.

Brush, G.S.; Martin, E.A.; DeFries, R.S.; Rice, C.A. Comparisons of ^{210}Pb and pollen methods for determining rates of estuarine sediment accumulation. Quater. Resear. 18:196-217; 1982.

Brush, G.S. Stratigraphic evidence of eutrophication in an estuary. Wat. Res. Resea. 20(5):531-541; 1984.

Garber, J.H. Benthic-Pelagic Coupling in Chesapeake Bay. In: Perspectives on the Chesapeake Bay: Recent Advances in Estuarine Sciencies. Eds: M.P. Lynch and E.C. Krome, Chesapeake Research Consortium. Gloucester Point, VA. 1987.

HydroQual, Inc. Development of a coupled hydrodynamic/water quality model of the eutrophication and anoxia process of the Chesapeake Bay. EPA Contract No. 68-03-3319. Annapolis, MD. 1987.

HydroQual, Inc. Workshop Number 2: Sediment processes and sediment modeling workshop, December 2-3, 1987. USCOE Contract No. DACW39-88-C-0004. U.S. Army Corps of Engineers, Baltimore District. Baltimore, MD. 1988.

Malone, T.C.; Kemp, W.M.; Ducklow, H.W.; Boynton, W.R; Tuttle, J.H.; Jonas, R.B. Lateral variation in the production and fate of.

Twilly, R.R.; W. M. Kemp. Preliminary results on the significance of sediment denitrification to the fate of nitrogen in the Chesapeake Bay. Final report to the EPA Chesapeake Bay Program. Annapolis, MD. 1985.

Wright, L.D.; D.B. Prior; C.H. Hobbs; R.J. Byrne; J.D. Boon; L.C. Schaffner; M.O. Green. Spatial variability of bottom types in the lower Chesapeake Bay and adjoining estuaries and inner shelf. Estuarine, Coastal and Shelf Science, Vol. 24, P. 765-784. 1987.

CONCURRENT SESSIONS AND POSTER SESSION:

PHYSICAL PROCESSES

Chairs:

William Boicourt
Horn Point Environmental Laboratory
University of Maryland

Evon P. Ruzecki
Virginia Institute of Marine Science
College of William and Mary

Understanding the Estuary: Advances in Chesapeake Bay Research. Proceedings of a Conference. 29-31 March 1988. Baltimore, Maryland. Chesapeake Research Consortium Publication 129. CBP/TRS 24/88.

Abstract only

Hampton Roads Circulation: The Combined Effects of General and Meso-Scale Features

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Historical data sets from the Hampton Roads region of the James River estuary and the James River Hydraulic Model obtained in 1964 and 1968 respectively are re-examined with respect to hypothesized net cyclonic transport in the lower James and a meso-scale circulation feature associated with a more recently examined (1987) frontal feature in this region. The 1964 data illustrate the development and decay of an upstream-directed subsurface jet-like feature during flood tide which coincides with the newly described front while subsequent hydraulic modal data indicated net cyclonic transport in the lower James over several tidal cycles. The combined effects of both circulation features are examined with respect to recirculation of oyster larvae spawned upstream and the upstream directed injection of salt and municipal/industrial contaminants from the Hampton Roads region. Effects of these circulation patterns are illustrated by sequential 'openings' of upstream shellfish beds which were closed as a result of a recent (1985) sewage treatment plant incident in the lower James.

Understanding the Estuary: Advances in Chesapeake Bay Research. Proceedings of a Conference. 29-31 March 1988. Baltimore, Maryland. Chesapeake Research Consortium Publication 129. CBP/TRS 24/88.

Abstract only

A Theory of Tidal Intrusion Front and its Practical Application

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A one-dimensional analysis for a two-layer fluid was formulated to describe the characteristics of a tidal intrusion front. The analytical result relates the vertical transport through the front (i.e., the diving depth of the heavier fluid) to the densimetric Froude number of the approaching flow and the depth change across the front. It also defines a critical Froude number, above which no steady state solution exists. Interpretation of the result provides information on the movement and maximum transport capacity of the front.

The theory was used to interpret the characteristics of the front observed off Newport News Point in the lower James/Hampton Roads of Virginia. Because of the phase difference between the tidal currents on the two sides of Newport News Point, a convergent zone is formed at early stage of flood tide. The density difference between the two water masses is large enough to make the more saline Hampton Roads water dive beneath the fresher water of the lower James, thus form a tidal intrusion front. The front moves upriver as flood current strength increases, slows down and intensifies as it encounters a steep drop of bottom elevation. The observed diving depths of heavier water were explained by the theoretical result.

The theory was used to predict the impact of a proposed man-made island on the frontal characteristics, particularly its ability to entrain oyster larvae to the lower portion of water column in which the net transport is upriver toward seed oyster beds. A 400 acre island was proposed to be constructed on the Hampton Flats, a shallow embayment on the north side of Hampton Roads, which is a broad water body at the mouth of the James River. The proposed island would be located immediately downriver of the front. Its effect on the flood current approaching the front was quantified with a two-dimensional (in horizontal plane) numerical model, when combined with inferences from oyster larvae studies, indicate that the transport capacity of the front would be markedly reduced by island construction at the proposed sites.

Understanding the Estuary: Advances in Chesapeake Bay Research. Proceedings of a Conference. 29-31 March 1988. Baltimore, Maryland. Chesapeake Research Consortium Publication 129. CBP/TRS 24/88.

Changes in Circulation and Salinity from Increased Channel Depth in the Baltimore Harbor

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1. INTRODUCTION

This paper presents results of numerical simulations of the circulation within the Baltimore Harbor system, in an effort to determine the quantitative effects of channel enlargement on the circulation and salt distribution within the Harbor, subject to the external environmental conditions found at various times throughout a normal year. The incentive for this study comes from the plan to dredge the principal navigation channels in the Patapsco River and in certain segments of the main stem of Chesapeake Bay. Within the Baltimore Harbor system, the dredging program calls for enlarging the navigation channels from their present dimensions, approximately 12.8 meters deep and 244 meters wide in the main ship channel, to the planned dimensions of approximately 15.3 meters deep and 244 meters wide. The long term average flow pattern of Baltimore Harbor includes a three-layered gravitational circulation (Stroup et al., 1961; Boicourt and Olson, 1982; Olson et al., 1982). The magnitude of this circulation depends upon the depth of the channel and on the vertical stratification imposed on the Harbor at its mouth by the main stem of Chesapeake Bay, both of which increase with dredging.

Figure 1 is a location map, showing points referred to in this paper. The mouth of the Harbor, which is taken to be the cross section at which the Patapsco River joins the main stem of Chesapeake Bay, is chosen to be the line joining Cedar Point on the south shore of the Patapsco with the North Point on the north shore. Ranked in terms of volume, the Harbor system has four major branches: Bear Creek, Curtis Creek, Middle Branch and Northwest Branch. We designate the Middle Branch as being the head of the Harbor. The mean low water volume of the Harbor system thus defined is 468×10^6 m^3. The largest Harbor tributaries, Bear and Curtis Creeks, have volumes of 40×10^6 and 26×10^6 m^3, respectively (Cronin and Pritchard, 1975). If we exclude all harbor branches, then the main channel of the Harbor through Middle Branch is 21 km long and

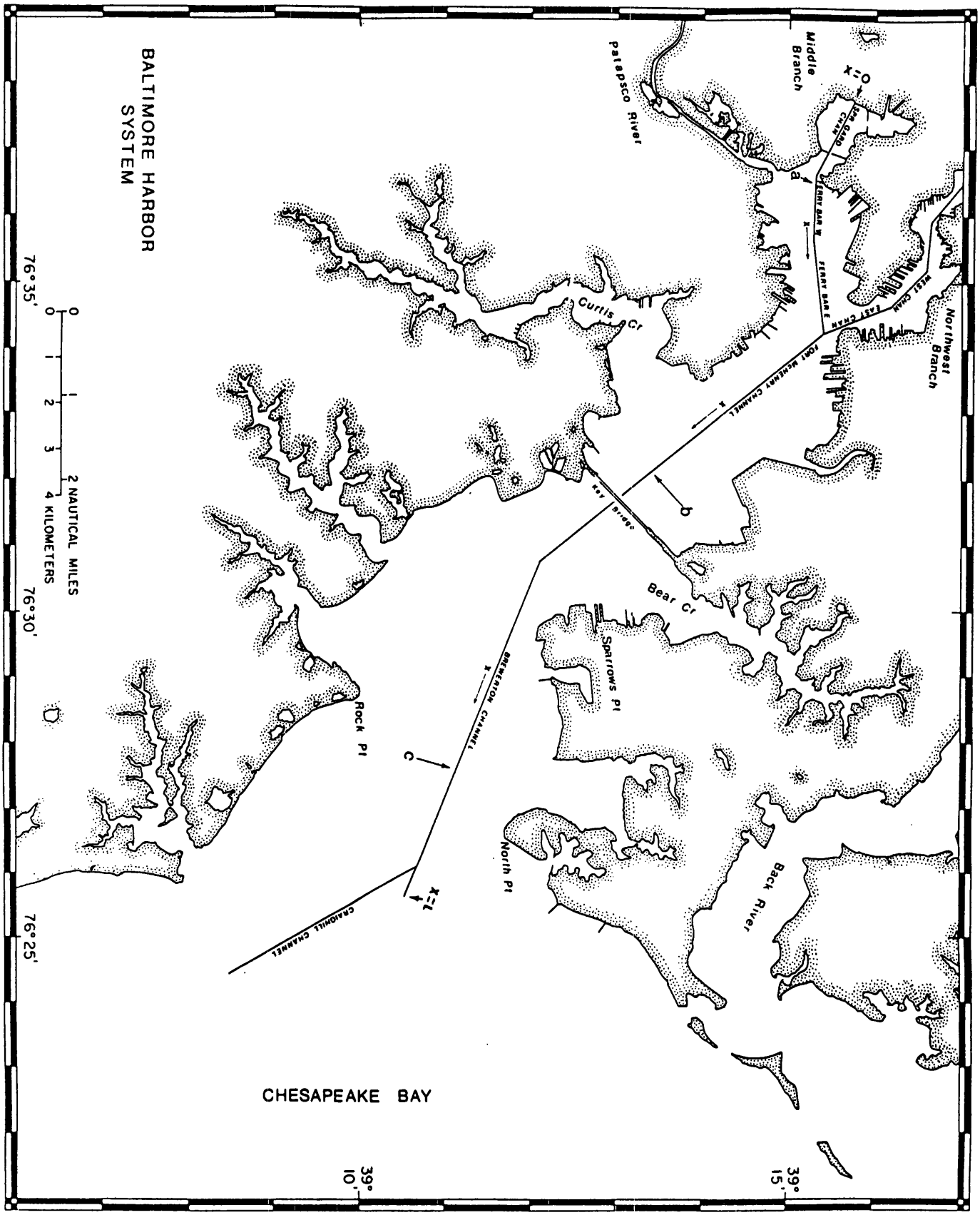

Figure 1: Reference map of the Baltimore Harbor system showing major navigation in channels, plus locations referred to in the text.

has a volume of approximately 364×10^6 m^3. In the outer portion of the Harbor, beyond Bear Creek, the cross sectional area is approximately 20,000 m^2. The average surface width in the outer portions of the Harbor is approximately 3 km and the average depth is 7 m. However, average depth is not a good measure of the Harbor bathymetry, because the greatest depths occur within relatively narrow ship channels.

Baltimore Harbor circulation has been the subject of study for several decades (Hachey, 1934; Garland, 1952; Stroup et al., 1961; Wilson, 1970; Hansen and Rattray, 1972; Hansen and Festa, 1974; Long, 1977; Boicourt and Olson, 1982; Olson et al., 1982). In addition to the interest in Baltimore Harbor circulation generated by commercial and environmental concerns, it has received considerable attention from physical oceanographers because it represents a mean flow induced primarily by small scale mixing processes. The essential physics of the three layer circulation is represented in the diagrams in figure 2. The low salinity surface water and the high salinity bottom water from nearby portions of the Chesapeake Bay enter the Harbor, where they are partially mixed by the action of the tides and winds. The resulting water is of intermediate salinity and density. It discharges outward into the Chesapeake Bay as a mixed layer, at a depth intermediate between the two incoming layers. Under equilibrium conditions the total salt content is the same for every cross section, and there is zero net salt and volume flux through any cross section. Nevertheless, there is a large volume transport in each of the individual layers. Field measurements by W. Boicourt (Boicourt and Olson, 1982) indicate three layer flow velocities range from 2 to 8 cm/sec when averaged over 3 to 7 day intervals.

The classical two-layer estuarine circulation, which is found within the main stem of the Chesapeake Bay, is evidently not present to any significant degree within the Baltimore Harbor system. This circulation is induced by partial mixing between salt water and fresh water derived from terrestrial run-off (Pritchard and Carpenter, 1960). The absence of an important contribution from this type of circulation can be explained by the low volume transport typical of the Patapsco River drainage. The average discharge from the Patapsco River is approximately 2.6 m^3/sec and the discharge from the Jones Falls into the Northwest Branch is significantly smaller than this. If we project the Patapsco discharge onto the mean cross sectional area of the outer portion of the Harbor, the resulting mean velocity is only .013 cm/sec. Thus, even allowing for the enhancement of a two layer circulation by mixing (which can amplify the circulation in each layer by a factor 10 or 20) the flow induced by the Patapsco River discharge, on average, will amount to significantly less than 1 cm/sec. For this reason, it is justifiable to neglect the influence of Patapsco River discharge on the laterally averaged circulation. The exception to this would be brief times following storm run-off.

2. NUMERICAL MODEL

The numerical model is a finite difference representation of the governing equations for laterally-averaged flow in a stably stratified shallow channel. We fix a coordinate system with the horizontal x-axis parallel to the ship channel as shown in figure 1, and the vertical z-axis positive upward. The origin of coordinates is fixed by mean sea level at the head of Middle Branch. The domain consists of Middle Branch plus the main stem of the Patapsco estuary; we exclude from consideration all other tributaries and branches. The conservation of horizontal momentum, in laterally averaged form, is

$$\frac{\partial}{\partial t}(uB)+\frac{\partial}{\partial x}(uuB)+\frac{\partial}{\partial z}(uwB)=gB\frac{\partial}{\partial x}\left(\int_0^z\frac{\rho}{\rho_o}dz'-\eta\right)$$

$$+\frac{\partial}{\partial x}\left(BN_x\frac{\partial u}{\partial x}\right)+\frac{\partial}{\partial z}\left(BN_z\frac{\partial u}{\partial z}\right)-ku\left|u\frac{\partial B}{\partial z}\right| \qquad (1)$$

Figure 2: A schematic diagram showing essential physical processes for the three-layer density driven circulation. Upper Diagram: Low salinity surface water and high salinity bottom water from Chesapeake Bay enter Baltimore Harbor, partially mix, and discharge out of the Harbor as a mixed layer. Lower Diagram: Typical profiles for salinity and horizontal velocity.

The laterally averaged conservation of salt is

$$\frac{\partial}{\partial t}(sB)+\frac{\partial}{\partial x}(usB)+\frac{\partial}{\partial z}(wsB)=\frac{\partial}{\partial x}\left(BK_x\frac{\partial s}{\partial x}\right)+\frac{\partial}{\partial z}\left(BK_z\frac{\partial s}{\partial z}\right) \qquad (2)$$

The conservation of mass is

$$\frac{\partial}{\partial x}(uB)+\frac{\partial}{\partial z}(wB)=0 \qquad (3)$$

which has the depth integrated form of

$$\frac{\partial}{\partial t}[\eta B(x,0)]+\frac{\partial}{\partial x}\int_{-D}^{\eta} uB\,dz=0 \qquad (4)$$

Finally, the equation of state connecting salt to density may be written

$$\frac{\rho}{\rho_0}=(1+\beta s) \quad . \qquad (5)$$

In equations (1)-(5), B(x,z) is lateral width, D is depth, g is gravity β is the saline coefficient, ρ_0 is freshwater density, k is the drag coefficient, (K_z, K_x) and (N_z, N_x) are (vertical, horizontal) mixing coefficients for salt and momentum, respectively. Representation of vertical mixing by diffusion coefficients is consistent with mixing by sidewall boundary layer processes (Phillips, Shyu and Salmun, 1987). The horizontal diffusion coefficients provide numerical stability and do not strongly influence the solution.

The appropriate boundary conditions are:

i) On the channel bottom, z = -D

$$w=\frac{\partial s}{\partial z}=0; \qquad N_z\frac{\partial u}{\partial z}=ku|u| \qquad (6a)$$

ii) On the surface, z = 0

$$w=\frac{\partial s}{\partial z}=0 \qquad (6b)$$

iii) At the Harbor head, the freshwater influx is ignored, as it makes a negligible contribution to the circulation. Therefore we have x = 0

$$u=\frac{\partial s}{\partial x}=0 \qquad (6c)$$

iv) At the Harbor mouth, x = L, we specify the surface elevation

$$\eta=\eta_m(t) \quad . \qquad (6d)$$

The salinity of inflowing water is specified

$$s=s_m(z,t) \quad if \quad u<0, \qquad (6e)$$

and we assume an advective balance for outflow

$$\frac{\partial}{\partial t}(sB)=\frac{-\partial}{\partial x}(usB) \quad if \quad u>0 \quad . \qquad (6f)$$

In addition to the driving provided by mouth stratification and fluctuations in sea level, we specify the longitudinal component of the wind stress

$$N_z\frac{\partial u}{\partial z}=\tau(t) \quad at \quad z=0 \quad . \qquad (7)$$

The data necessary to initialize a run consist of the initial distribution of salinity and surface elevation

$$s_0 = s(x, z, t=0)$$

$$\eta_0 = \eta(x, t=0) \tag{8}$$

Calculations are made for three domains:

i) The base configuration, consisting of the main branch of the Patapsco estuary plus the Middle Branch, both with a dredged channel 12.8 m deep, 244 m wide, 21 km long, and having a total volume of 264 x 10^6 m^3.

ii) The plan configuration, which is identical to the base configuration except that the channel is deepened everywhere to 15.3 m, and 244 m wide. The total volume of the plan configuration is 276 x 10^6 m^3.

iii) The ideal rectangular channel. A sequence of calculations is carried out in a rectangular channel 12.8 m deep, 1 km wide and 21 km long. The purpose of these simulations is to check our parametric representation of the three layered flow in an idealized channel, in which topographic effects are minimized.

The grid used for both base and plan domains consists of 22 equally spaced cross sections, with 20 grid points in the vertical at each cross section. The details of the computational algorithm can be found in Wang and Kravitz, 1980 and Olson et al., 1982. We have carried out the set of calculations summarized in tables 1 and 2. Table 1 gives the pertinent data for the steady state calculations, which are driven by imposing at the Harbor mouth the top and bottom water salinities, and an astronomical tide. The astronomical tide data was taken from the Fort McHenry tidal station records. Top and bottom water salinities were taken from the published results of the Chesapeake Bay Hydraulic Model experiment (Granat and Gulbrandsen, 1981, plate no. 78) in which the effects of channel dredging in the main stem of Chesapeake Bay were modelled. We assume these results are indicative of salinity changes expected in the main stem, and we use them as boundary conditions for the plan geometry. Table 2 lists the parameters for the model runs which include meteorological forcing. The applied wind stress was calculated from hourly wind data recorded at Baltimore Washington International Airport during the appropriate season. Nontidal sea level fluctuations during the same time interval come from the recordings at the Fort McHenry tide station.

A number of useful parameters can be computed which characterize the overall Harbor response, including volume flux, residence (flushing) time and stratification in the Harbor interior. For the three layer flow, four quantities prove to be diagnostic measures of the Harbor response: the mixed layer volume flux Q, the residence time T, the interior stratification ΔS_h and δ, the length scale of the flow. These are defined as

$$Q = \int_{-D}^{\eta} uB\,dz \qquad x = L, \quad u > 0$$

$$T = \frac{V}{Q} \tag{9}$$

$$\Delta S_h = S(-D, 0) - S(0, 0)$$

$$\delta = \int_0^L \frac{u(x, -D/2)}{u(L, -D/2)}\,dx$$

Dimensional Analysis

In order to apply the numerical results to cases whose external conditions differ from those in table 1, it is desirable to establish scaling laws which correctly determine the Harbor response to arbitrary changes in both channel depth and mouth salinities.

For the three layered flow, in steady state, we introduce the following dimensionless variables (denoted by primes)

$$(x',z') = (x/L, z/D)$$

$$(u',w') = (Lw, Du)/K_z$$

$$s' = s/\Delta s_m \tag{10}$$

$$B' = B/\bar{B}$$

where $\bar{B}$ is the depth-averaged width, related to the cross-sectional area A by

$$A = \bar{B}D \quad .$$

Equations (1) and (2) become

$$\frac{1}{P}\left[\frac{\partial}{\partial x'}\left(u'^2 B'\right) + \frac{\partial}{\partial z'}(u'w'B') - \Delta k u'\left|u'\frac{\partial B'}{\partial z'}\right|\right] =$$

$$\frac{I_N}{\Delta}\frac{\partial}{\partial x'}B'\frac{\partial u'}{\partial x'} + \Delta\frac{\partial}{\partial z'}B'\frac{\partial u'}{\partial z'} - R\int_{z'}^{0}\frac{\partial s'}{\partial x'}dz \tag{11}$$

$$\frac{\partial}{\partial x'}(u's'B') + \frac{\partial}{\partial z'}(w's'B') = \Delta\left[\frac{\partial}{\partial z'}B'\frac{\partial s'}{\partial z'} + \frac{I_K}{\Delta^2}\frac{\partial}{\partial x'}B'\frac{\partial s'}{\partial x'}\right]$$

with parameters

$$P = N_z/K_z$$

$$\Delta = L/D$$

$$I_N, K_K = N_x/N_z, K_x/K_z \tag{12}$$

$$R = \frac{g\beta\Delta S_m}{K_z N_z}D^3 \quad .$$

In dimensionless form, the response parameters (9) become

$$Q' = \int u'B'dz'; \quad x' = 1, u' > 0$$

$$T' = \frac{V}{DA}\frac{1}{Q'}$$

$$\Delta s'_h = s'(0,1) - s'(0,0) \tag{13}$$

$$\delta' = \int_0^1 \frac{u'(x',-1/2)}{u'(1,-1/2)}dx'$$

The response of the three-layered flow depends strongly on the Rayleigh number R, which measures the relative importance of stratification versus mixing. Its value for Baltimore Harbor can range between 10^6 and 10^{12}, depending on external conditions. By contrast, the dependence on other parameters is weak. The aspect ratio Δ is fixed by Harbor geometry. Our representation of momentum and salt mixing by constant diffusivities requires, for internal consistency, that the turbulent Prandtl number P has a value near unity. The three-layered flow is insensitive to values of mixing anisotropy paramenters I_n, I_k less than 10^5.

3. STEADY STATE CALCULATIONS

In this set of calculations the flow is driven only by the astronomical tide and the vertical stratification of Chesapeake Bay imposed at the Harbor mouth. The surface and bottom water salinities used as boundary conditions are listed in table 1.

With no meteorological forcing present explicitly, model results indicate that the mean circulation settles into a tidally-averaged steady state three-layered flow within a period of 10-30 tidal cycles, depending on the specific initial conditions used. The structure of the three-layered circulation is most sensitive to the value assigned to the vertical mixing coefficient K_z. Numerical experiments indicate that a vertical mixing coefficient lying within the range

$$0.1 \leq K_z \leq 1.0 \quad cm^2/sec$$

best matches the observed Harbor salinity cross sections, as measured by Boicourt and Olson (1982). For purposes of comparison, we carry out all calculations with two values: K_z=0.3 and K_z=1.0. These values result in best agreement with observed salinity patterns under weak and vigorous mixing conditions, respectively.

When the freshwater discharge into the Chesapeake Bay reaches its seasonal low in summer and early autumn, the average salinity in Baltimore Harbor is highest, and the stratification is weakest. A weak three-layered flow is expected under these conditions. Results of calculations for base and plan channel configurations are shown in table 3, and in figures 3-4. The salinity patterns are essentially the same for both base and plan configurations; the isohalines diverge from the Harbor mouth in a fan pattern which is the characteristic signature of the three-layered circulation. The important difference between plan and base is found in the magnitude of the stratification. In the base configuration, the inner two-thirds of the Harbor is essentially homogeneous, while in plan configuration a stratification of 2-3ppt exists as far up as Key Bridge (point b). The stronger stratification found in the plan configuration is reflected in the magnitude of the three-layered flow. For the base configuration, a weak three-layered circulation (with velocities less than 2 cm/sec) is found near the Harbor mouth only; the gravitational circulation inside the Harbor is negligible. By contrast, in the plan configuration a vigorous three-layered flow is found at the mouth (with velocities up to 7 cm/sec) and a weak but still significant mean circulation exists within the Harbor interior.

Equivalent tendencies are seen in figure 4, which shows the same calculation carried out with a mixing coefficient of K_z=1.0. The enhanced mixing results in more homogeneity than the case illustrated in figure 3, but the relationship between base and plan response is preserved. In particular, the strength of the three-layered flow at the Harbor mouth is essentially the same for both choices of vertical mixing coefficients.

Table 3 summarizes the comparison between base and plan configuration response, in terms of the response parameters defined by equation (9). The increase in channel depth plus the concomitant increased mouth stratification increases the outflowing volume flux within the middle layer from approximately 87 m^3/sec to 210 m^3/sec. This translates into a decrease in residence time from approximately 48 days to 20 days, a

Table 1: Calculation Parameters - Steady State

Channel	Low Flow				High Flow			
	length	depth	ΔS (mouth)	duration	length	depth	ΔS (mouth)	duration
base	21 km	12.8 m	1.8 ppt	50 td cy	21 km	12.8 m	9.5 ppt	50 td cy
plan	21	15.3 m	4.85	50	21	15.3	15.75	50
ideal	21	12	many	variable	-----	-----	-----	-----

Table 2: Calculation Parameters - Variable Forcing

Channel	Low Flow				High Flow			
	length	depth	ΔS (mouth)	duration	length	depth	ΔS (mouth)	duration
base	21 km	12.8 m	1.8 ppt	50 td cy	21 km	12.8 m	9.5 ppt	50 td cy
plan	21	15.3 m	4.85	50	21	15.3	15.75	50

Table 3: Critical Response Parameters, Low Flow Conditions

Channel	Residence Time	Outward Flux	Interior Stratification	Length Scale	Mixing Coefficients
Base	48 days	87 m^3/sec	0.1 ppt	11.2 km	0.3 cm^2/sec
	47	89	0.02	8.3	1.0
Plan	20.5	210	1.3	7.5	0.3
	20	215	1.1	8.0	1.0

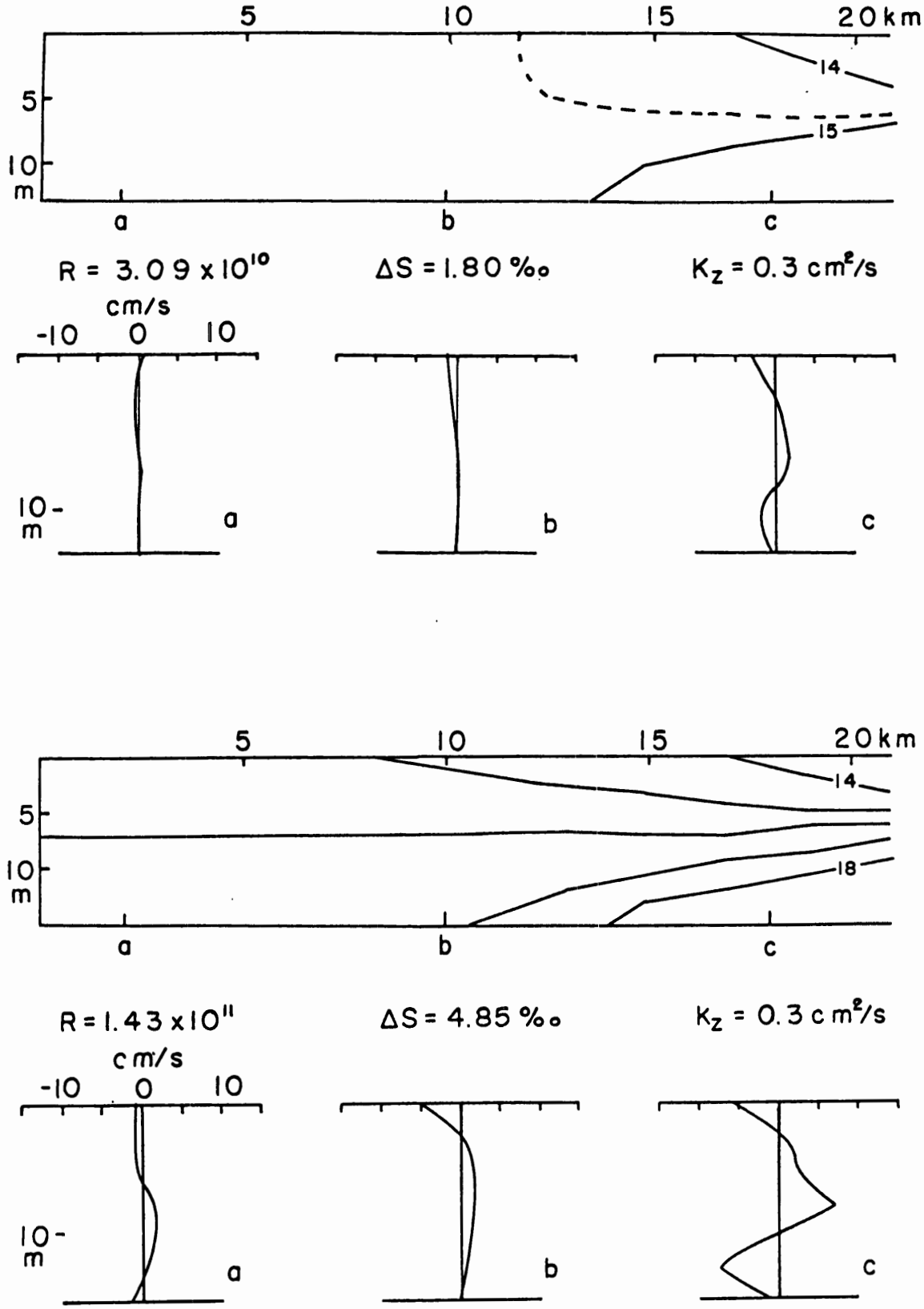

Figure 3: Comparison of base configuration (upper) versus plan configuration (lower) at steady state under low flow conditions. Shown are laterally averaged salinity cross sections contoured in ppt from Middle Branch to the Harbor mouth, plus three representative velocity profiles. Nominal locations: a=Middle Branch, b=Key Bridge-Curtis Bay, c=North Point. Calculations made with a vertical mixing coefficient K_z = 0.3cm²/sec.

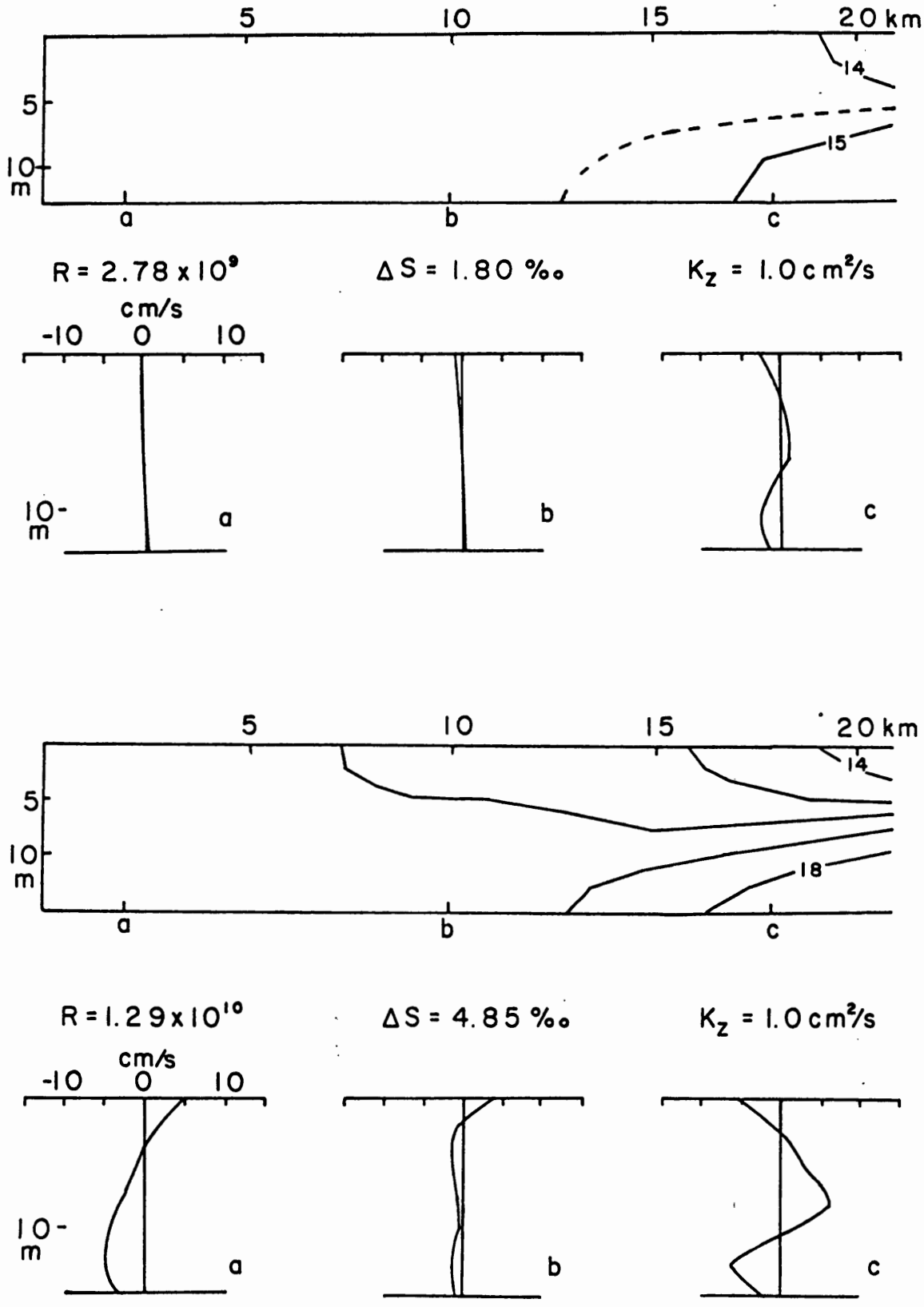

Figure 4: Same as figure 3 except that calculations were made with K_z = 1.0 cm^2/sec.

reduction of 60%. The interior stratification, measured within the Middle Branch, increases by approximately 1 ppt. The horizontal length scale of the three-layer flow, the distance of penetration into the Harbor, is essentially the same for base and plan configurations.

High flow conditions

As the freshwater discharge into the Chesapeake Bay reaches its seasonal high--in late winter and early spring--the average salinity in Baltimore Harbor is lowest and the mouth stratification is at a maximum. Under these conditions the strength of the three-layer flow is also at a maximum.

Results of calculations for both base and plan channel configurations are shown in table 4 and in figures 5 and 6, for high flow conditions. From figure 5 it is apparent that the effect of increasing Chesapeake Bay stratification is to stratify the entire Harbor; this is especially prominent for the planned configuration. For example, within the Middle Branch (point a) a vertical stratification of 2.5 ppt occurs in the base configuration experiment, and a stratification of over 6 ppt occurs in the plan configuration. As can be seen from the velocity profiles, this enhanced stratification results in a stronger three-layered flow throughout the Harbor system, when compared to weak stratification conditions. This is particularly true for the enlarged channel. At high flow, in the base configuration, three-layered flow velocities approach 8 cm/sec at the mouth and drop to less than 2 cm/sec past Key Bridge, while in the plan configuration, at high flow, velocities reach 17 cm/sec at the mouth and remain as large as 7 cm/sec in the inner portion of the Harbor.

As was found for low flow conditions, channel enlargement significantly affects the Harbor response parameters. Table 4 is a summary of these. The outward volume flux within the mixed layer increases from approximately 300 m^3/sec in the base configuration to 500 m^3/sec in the plan configuration. This translates into a decrease in residence time from approximately 14 to 8 days. The stratification near the Harbor head increases by 3-6 ppt as a result of the deepened channel, depending on the strength of turbulent mixing processes. For a vertical mixing coefficient of K_z = 1.0 cm^2/sec (strong mixing), the increase in the stratification in the Harbor interior is approximately 3 ppt, while for K_z = 0.3 cm^2/sec (weak mixing) the increase is by 6 ppt.

Parametric description of three layer flow

In order to make predictions about Harbor response to arbitrary changes in channel depth and Chesapeake Bay salinities, it is desirable to generalize these results in the form of simplified scaling laws. For the three-layered flow in a tidally-averaged steady state, the analysis of Section 2 indicates the critical dimensionless parameter is the Rayleigh number R. To further our understanding of how the three-layered flow depends on R, we have conducted an additional series of numerical experiments using the rectangular version of the Harbor, consisting of a channel 1 km wide, 12 m deep and 21 km long. We have varied both the mouth stratification and the mixing coefficient K_z in order to vary the Rayleigh number between the limits

$$10^8 < R < 3x10^{12}$$

Figures 7-13 show the results of these simulations. The first three figures, 7-9, illustrate the transition in salinity and velocity fields as R is increased. At relatively small values of R (figure 7) mixing is relatively strong while stratification is relatively weak and a significant stratification occurs near the Harbor mouth only. The three layer flow exists only in the outer one-third of the Harbor; internal to that is a weak two layer flow. As the Rayleigh number is increased to moderate values (figure 8) the stratification and the three layer flow fill the outer two-thirds of the Harbor. At large values of the Rayleigh number (figure 9) both the stratification and the three layer flow extend throughout the whole length of the Harbor.

Table 4: Critical Response Parameters, High Flow Conditions

Channel	Residence Time	Outward Flux	Interior Stratification	Length Scale	Mixing Coefficients
Base	15 days	273 m^3/sec	3.3 ppt	7.0 km	0.3 cm^2/sec
	13	324	0.3	4.2	1.0
Plan	8.8	491	9.2	7.0	0.3
	8.1	535	3.0	8.6	1.0

Table 5: Scaling Laws for Three Layered Flow

Parameter	Scaling	Power Law	
		Baltimore Harbor	Rectangular Channel
Mixed Layer (Outward) Flux	BK_z	$Q' = 1.4\times10^{-2}R^{1/2}$	$Q' = 4.3\times10^{-2}R^{1/2}$
Residence Time	D^2/K_z	$T' = 9\times10^{4}R^{-1/2}$	$T' = 3.9\times10^{4}R^{-1/2}$
Interior Stratification	ΔS_m	$\Delta S'_h = 3\times10^{-12}R$	$\Delta S'_h = 6.9\times10^{-12}R$
Length Scale	L	----------	$\delta' = 1.3\times10^{-3}R^{1/4}$

note: $R = \beta g \Delta S_m D^3/K_z N_z$; B = depth averaged width at the mouth.

The curves corresponding to these power law fits are shown in the figures 10-13.

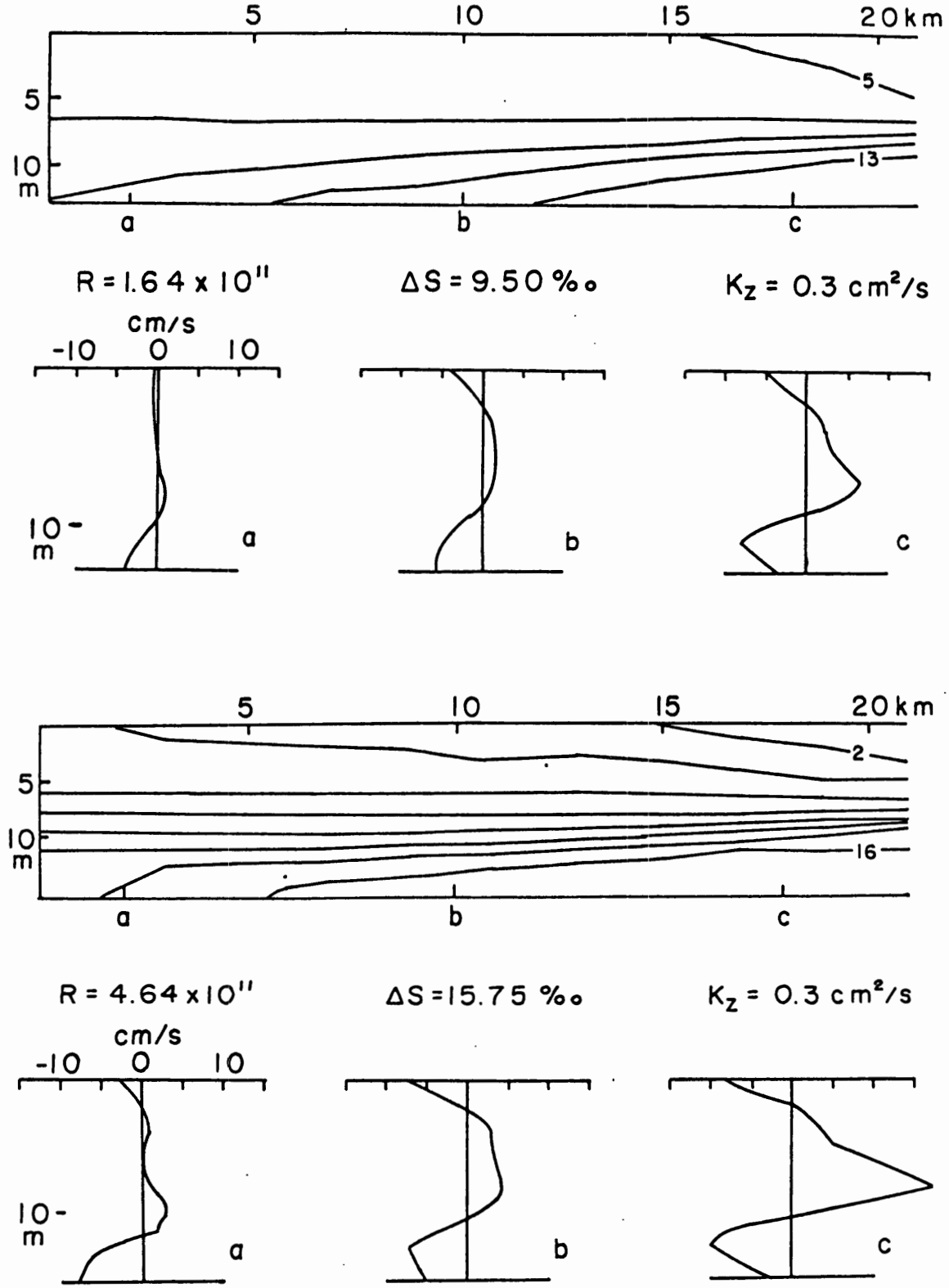

Figure 5: Comparison of base configuration (upper) versus plan configuration (lower) at steady state, under high flow conditions. Calculations made with a vertical mixing coefficient K_z = 0.3cm²/sec.

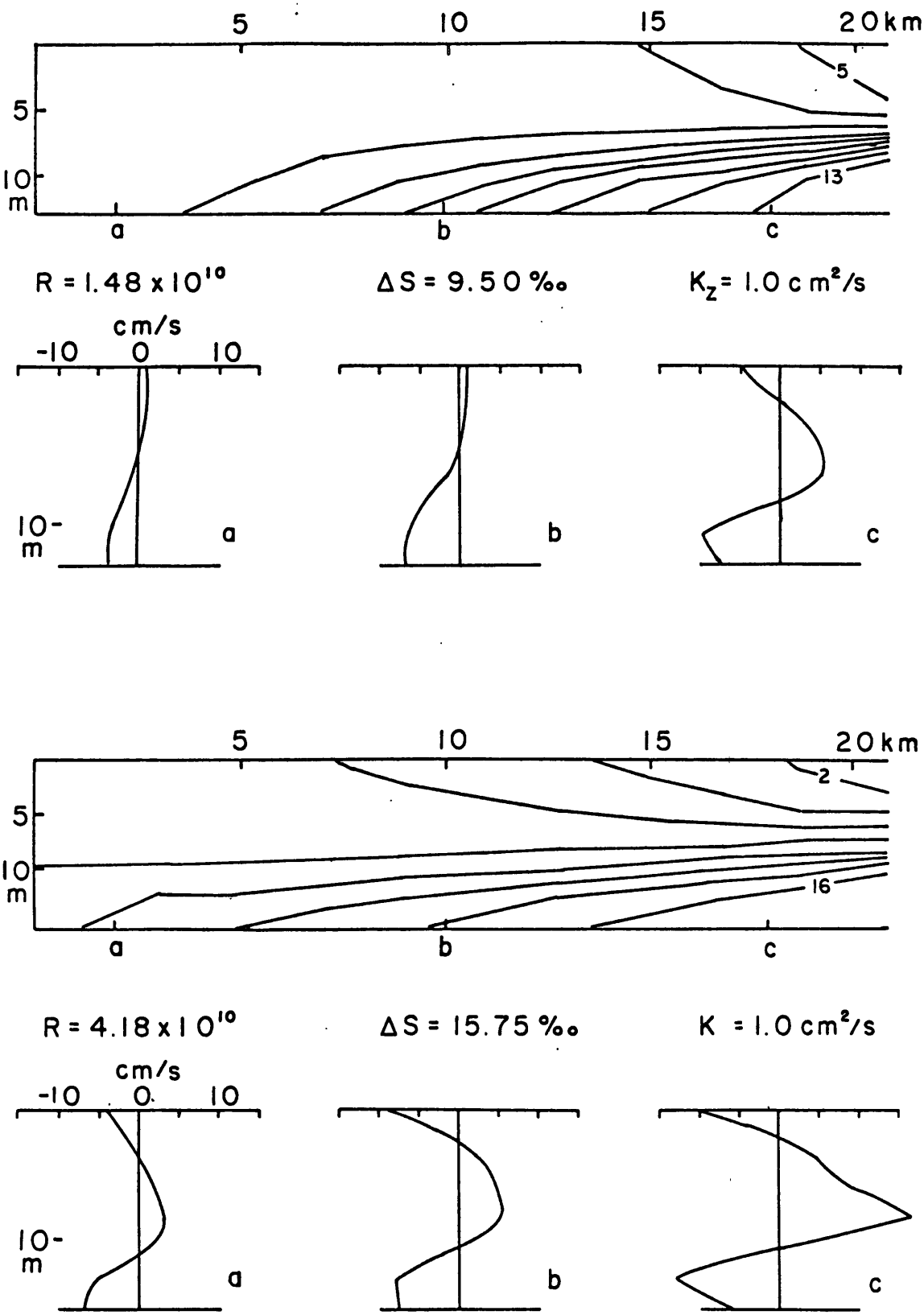

Figure 6: Same as figure 5 except that calculations made with K_z = 1.0 cm^2/sec.

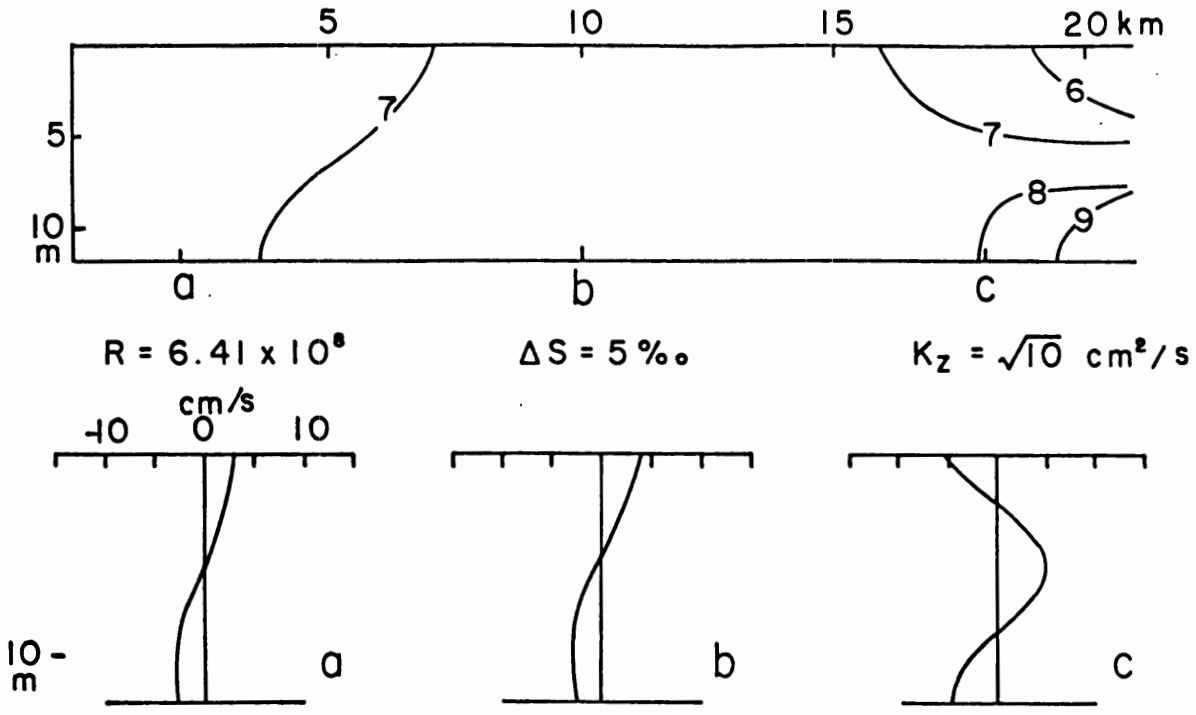

Figure 7: Steady state response of idealized rectangular channel 1 km wide, 12 m deep, 21 km long, to imposed stratification of 5 ppt at R = 6.41×10^8. Shown is the laterally-averaged salinity cross section plus three vertical profiles of horizontal velocity.

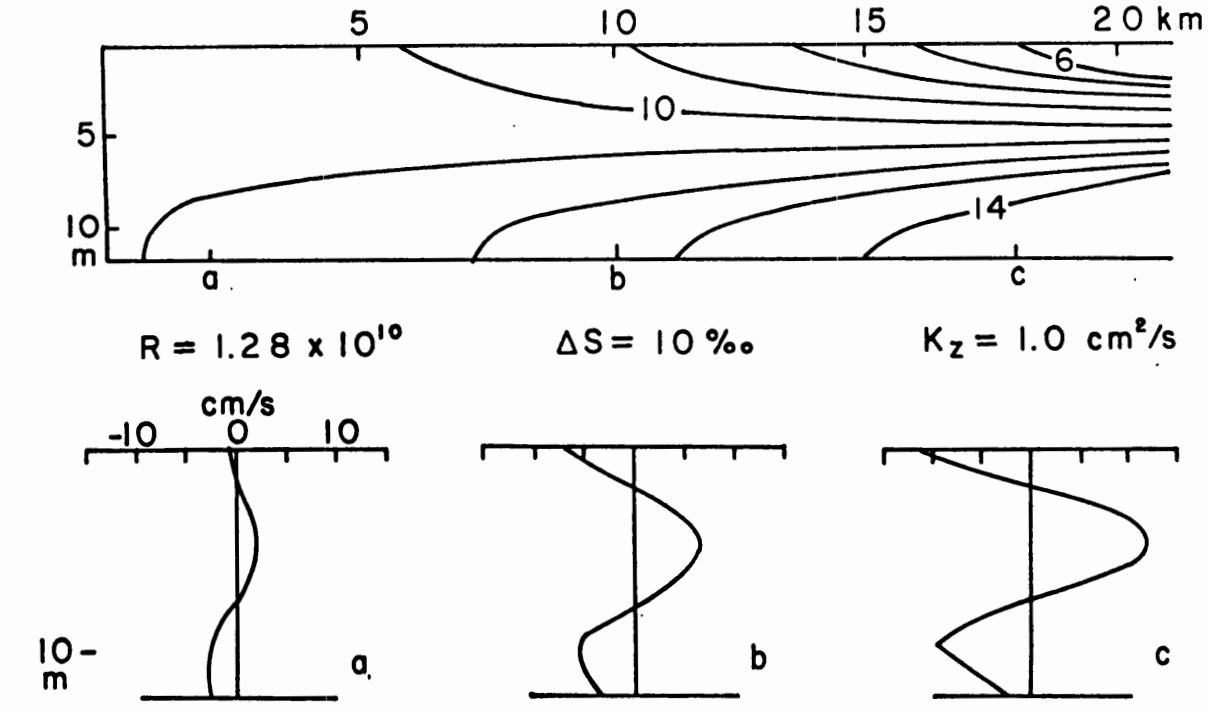

Figure 8: Steady state response of idealized rectangular channel to imposed stratification of 10 ppt at R = 1.28×10^{10}.

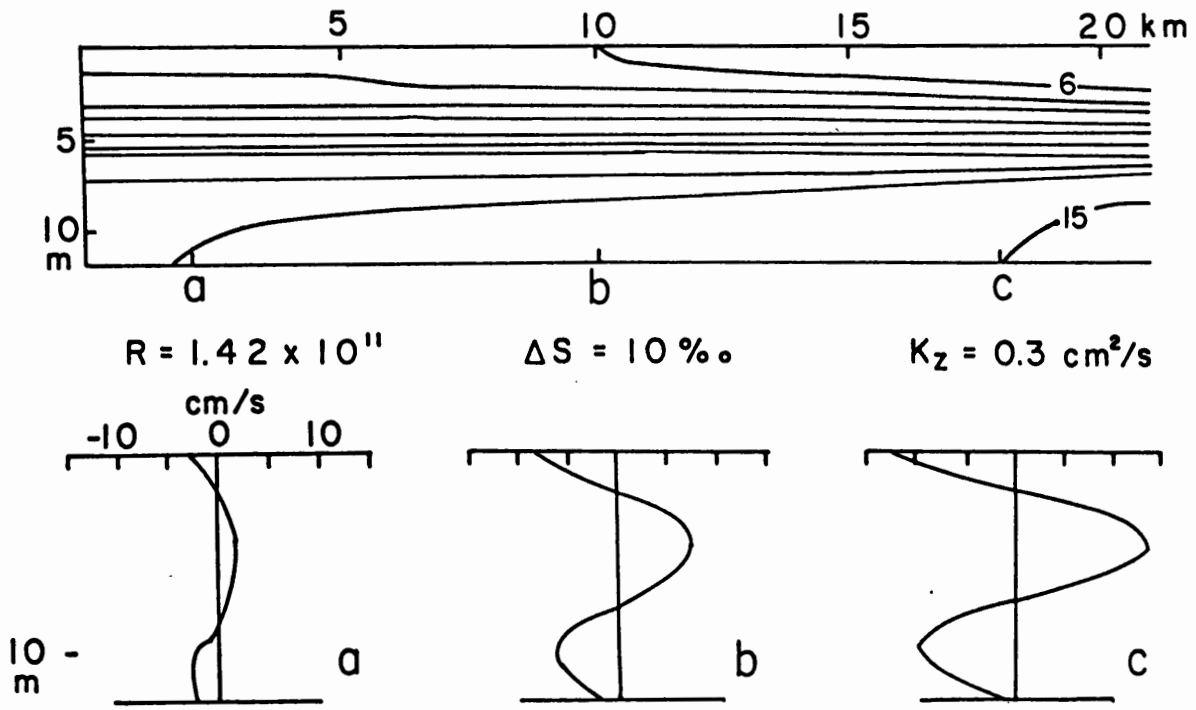

Figure 9: Steady state response of idealized rectangular channel to imposed stratification of 10 ppt at R = 1.42×10^{11}.

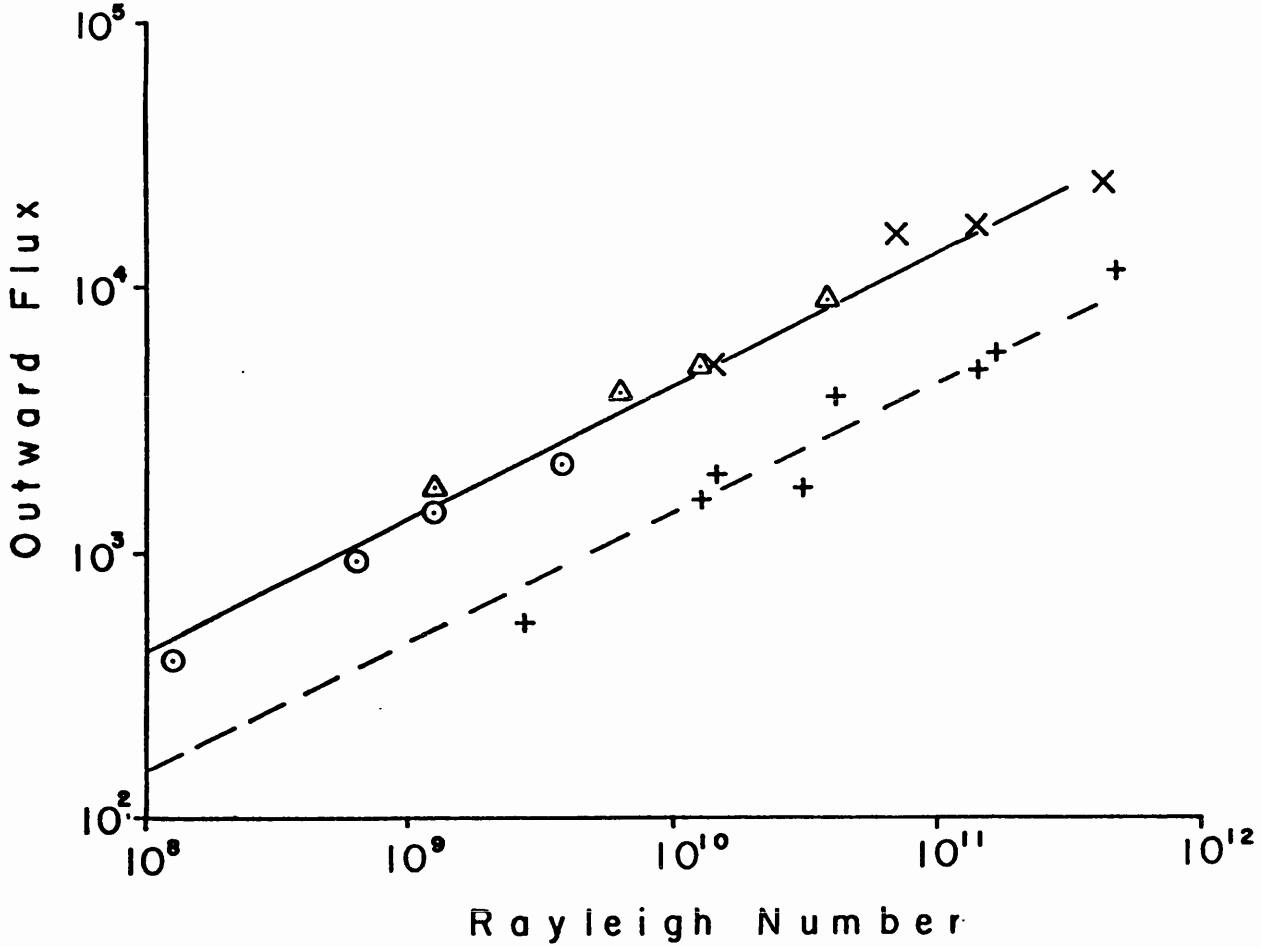

Figure 10: Dependence of the outward volume flux in the mixed layer of the three-layered flow, on the Rayleigh number R for both Baltimore Harbor (+) and the rectangular channel (all other symbols). Power law fits are $Q' = 1.4 \times 10^{-2} R^{1/2}$ for Baltimore Harbor (dashed line) and $Q' = 4.3 \times 10^{-2} R^{1/2}$ for the rectangular channel (solid line).

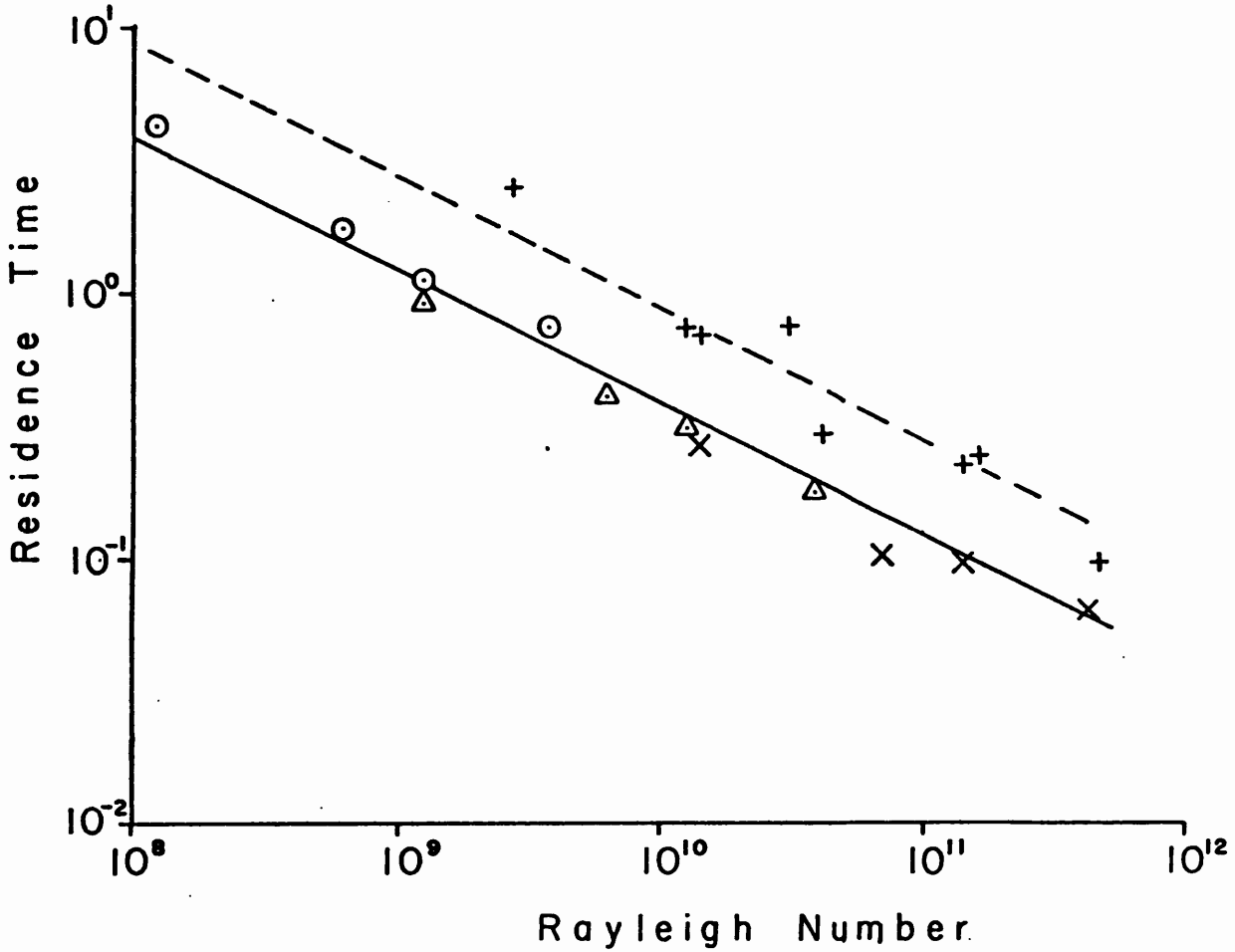

Figure 11: Dependence of residence time for three-layered flow on the Rayleigh number R for Baltimore Harbor (+) and the rectangular channel (all other symbols). Power law fits are $T' = 9 \times 10^{4} R^{-1/2}$ for Baltimore Harbor (dashed line) and $T' = 3.9 \times 10^{4} R^{-1/2}$ for the rectangular channel (solid line).

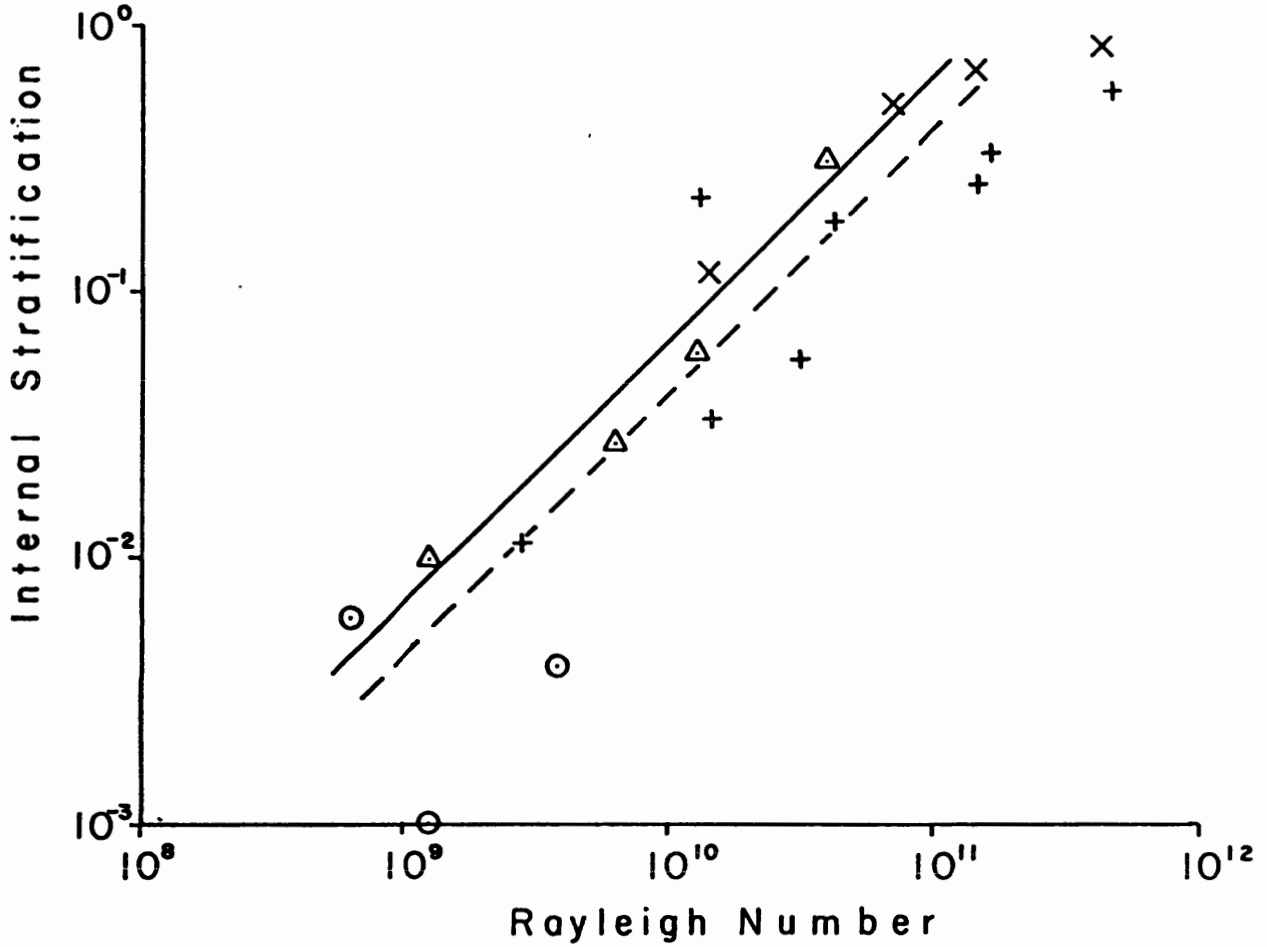

Figure 12: Dependence of the steady state interior Harbor salinity on the Rayleigh number R for Baltimore Harbor (+) and the rectangular channel (all other symbols). Power law fits are $\Delta S'_h = 3\times10^{-12}R$ for Baltimore Harbor (dashed line) and $\Delta S'_h = 6.9\times10^{-12}R$ for the rectangular channel (solid line).

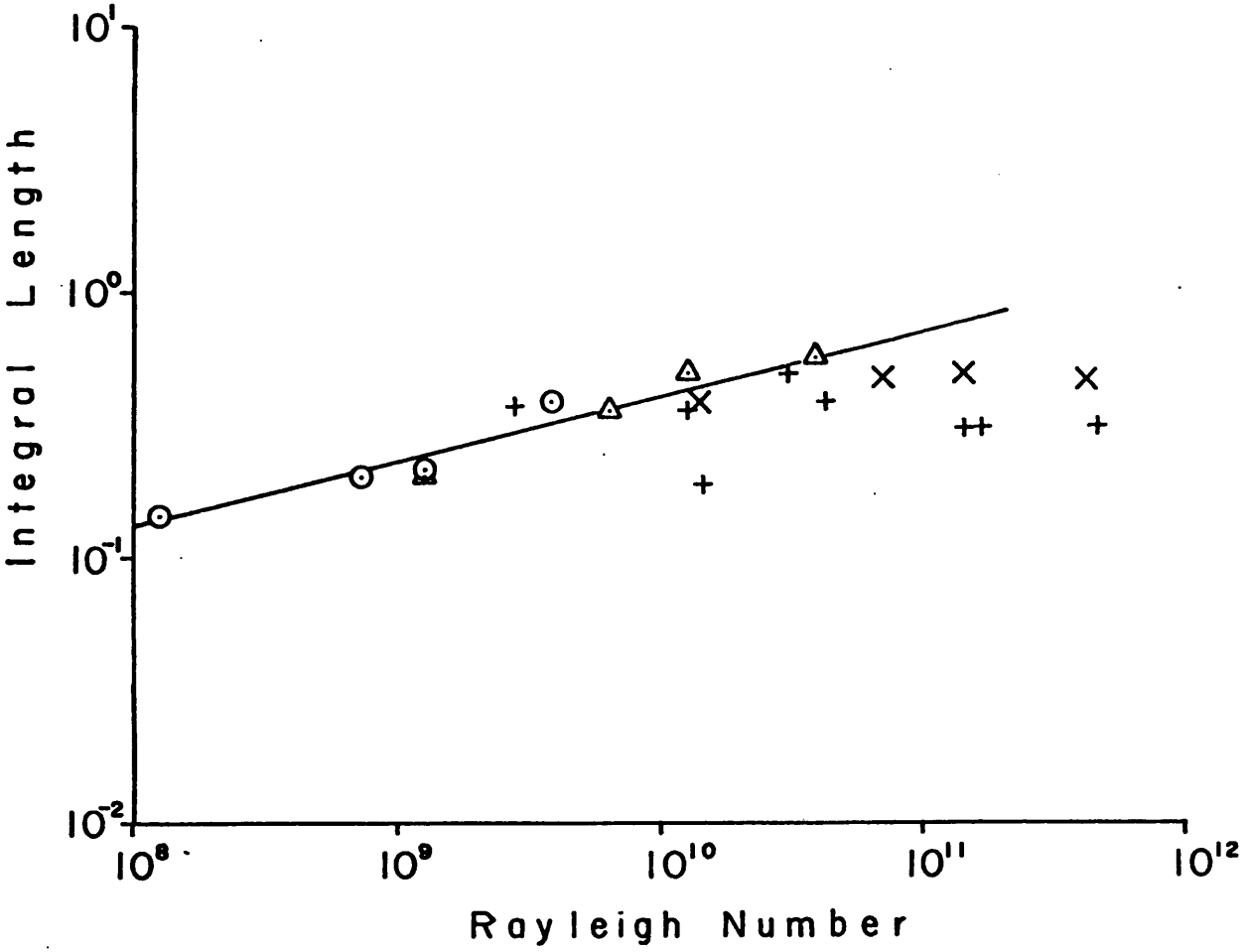

Figure 13: Dependence of the integral length scale (penetration distance) of three-layered flow on the Rayleigh number R for Baltimore Harbor (+) and the rectangular channel (all other symbols). Power law fit is $\delta' = 1.3 \times 10^{-3} R^{1/4}$ for the rectangular channel.

The dependence of the response parameters on R is shown in figures 10-13. Plotted on each figure are the results using the idealized rectangular channel, plus the results from the simulations already described in this section for plan and base configurations of Baltimore Harbor. Although there exists some scatter, it is clear that the response parameters for the three-layered flow can be scaled in terms of simple power law functions of the Rayleigh number, over most of the range of Rayleigh numbers considered. In figures 10-13 we have fit the results to power laws of the type aR^n; the values for a and n are given in Table 5.

As the Rayleigh number approaches and exceeds 10^{11} a transition occurs and the response parameters diverge from simple power law behavior. This transition occurs when the inflowing layers penetrate all the way to the Harbor head without appreciable mixing. In that limit the parameter $\Delta S_h'$ asymptotes to unity, and Q' and T' asymptote to constant values. Thus, the power law regime applies to circumstances in which the stratification at the head is much weaker than at the mouth. All available data indicates that this condition is met in Baltimore Harbor, that the appropriate Rayleigh number is less than 10^{11}, and that the flow lies in the power law regime.

From figures 10 and 11, and table 5, it is evident that the outward flux in the mixed layer of the three-layered flow, at the mouth of Baltimore Harbor, obeys a scaling law of

$$Q' = 1.4 \times 10^{-2} R^{1/2}$$

and similarly the flushing time obeys

$$T' = 9 \times 10^{4} R^{-1/2}$$

Expressed in terms of dimensional quantities the above formulas become

$$Q = 1.410^{-2} (g\beta\Delta S_m D)^{1/2} A$$

$$T = 9 \times 10^{4} (D/g\beta\Delta S_m)^{1/2}$$

Both volume flux and residence time depend only on mouth salinity and channel dimensions. Importantly, they are independent of mixing. The fact that they do not depend on the strength of vertical mixing makes these scaling laws additionally robust.

The scaling law for outward volume flux given in table 3 was first proposed by Long (1977) on the basis of a simplified analytical model. For a rectangular channel, Long obtained

$$Q' = R^{1/2}/8$$

Long's coefficient 1/8 is greater than our result by a factor of about 3, but there is agreement on the Rayleigh number dependence.

4. CALCULATIONS WITH TIME VARIABLE METEOROLOGICAL FORCING

The influence of local and non-local winds on circulation in Chesapeake Bay and its tributaries has been extensively documented (see Wang and Elliott, 1978; Wang, 1979a,b; Grano, 1982). These studies leave little room for doubt that wind strongly affects the circulation within the Chesapeake Bay system over time scales of two to ten days. The experiments performed in this Section are similar to those presented in Section 3, consisting of a direct comparison between the base and plan Harbor channel configurations, both under low flow and high flow conditions. In addition, the experiments presented in this section include the effect of surface wind stress and meteorological tides. In all the simulations presented in this section, a value of the vertical mixing coefficient $K_z = 1.0$ cm^2/sec was used. The duration of each of these experiments was 50 tidal cycles.

The variability in the circulation induced by meteorological forcing is best summarized as time histories of overall response parameters as shown in figures 14 and 15. The first two traces show the variation in the applied wind stress and sea level at the mouth over the last 30 tidal cycles. During low flow conditions (figure 14) the wind is generally weak, and when strong wind events do occur they usually come from the south and southeast. In response to wind events from that direction the Chesapeake Bay tends to fill, resulting in an increase in sea level at the Harbor mouth. Thus there exists a negative corrrelation between sea level at the Harbor mouth and the local direction of the wind stress.

While the wind stress is generally weak during this portion of the hydrological cycle, the stratification within is sufficiently weak also so that the wind driven component of the circulation can dominate flow within the Harbor. The last two traces on figure 14 show the Harbor response, in terms of the horizontal velocity as a function of time, at three specified depths near the Harbor mouth. The upper graph is the response of the base configuration, while the lower graph is response of the plan. It is clear from these traces that the response is basically baroclinic, with the surface layer following the wind and the middle and deep layers opposing the wind. In the base configuration, velocities in the near surface layer show excursions up to 15 cm/sec with an average velocity over the 50 tidal cycles of approximately 5 cm/sec, directed up the estuary. The middle and deep layers in the base configuration show velocity excursions up to 20 cm/sec, and over the 50 tidal cycles the average velocity is approximately 5 cm/sec. In the plan configuration, the near surface layer exhibits transient excursions up to 15 cm/sec and the average velocity in that layer over the 50 tidal cycles simulation is approximately 6 cm/sec directed upestuary. In the middle and deep layers, velocity excursions of up to 25 cm/sec occur, and the average velocity over the duration of the experiment is approximately 7 cm/sec in both bases, directed upestuary in the bottom layer and downestuary in the middle layer.

The applied wind stress and the mouth sea level under high flow conditions as a function of time for 30 tidal cycles are shown at the top two traces in figure 15. During this part of the hydrological cycle wind stress is relatively high compared to its yearly average, and strong wind events generally come from the north. In spite of the fact that wind stress tends to be high, the stratification found in the Harbor is also at a yearly maximum, and is sufficiently strong that the three-layer component of the circulation tends to dominate over the transient components. This dominance is illustrated in the last two traces which show the horizontal velocity at three depths near the Harbor mouth as a function of time in both the base and plan configurations.

In the base configuration, velocity excursions of up to 15 cm/sec occur in the surface layer directed both up and down channel. The 50 tidal cycle average velocity is approximately -5 cm/sec, consistent with that associated with the three-layer circulation. The middle layer also shows excursions up to 25 cm/sec, but the velocity is directed down-channel during nearly all of the 50 tidal cycle experiment. Its average value is approximately 8 cm/sec. In the bottom layer, velocity excursions of up to -15 cm/sec occur, and the average velocity is approximately -9 cm/sec.

The pattern of response in the plan configuration is very similar to the pattern of response in the base configuration, the difference being in the magnitude. In the plan configuration, excursions in the surface layer reach -18 cm/sec and the average velocity is approximately -7 cm/sec. In the middle layer, velocity excursions of up to 25 cm/sec occur and the average velocity is approximately 10 cm/sec. In the deep layer, velocity excursions of up to -22 cm/sec occur and the average velocity over the duration of the experiment there is approximately -15 cm/sec.

5. CONCLUSIONS

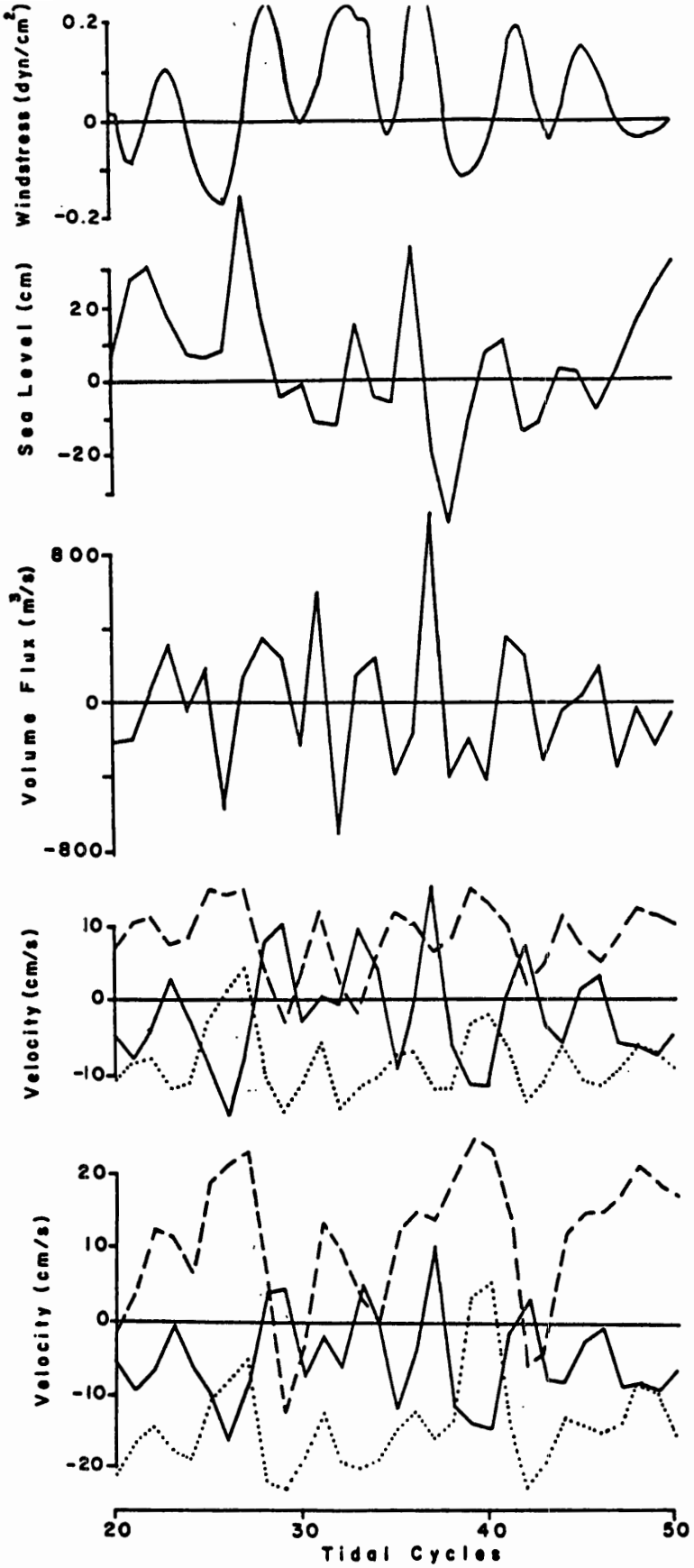

Figure 14: Harbor response to meteorological forcing at low flow conditions. Shown are the following time histories (from top to bottom): (i) longitudinal wind stress component; (ii) meteorological tide at Harbor mouth; (iii) volume flux at mouth; (iv) horizontal velocity at three depths for base configuration (solid=near surface, dash=middle, dots=near bottom); (v) horizontal velocity at three depths, plan configuration. The volume flux was essentially the same for both plan and base configuration.

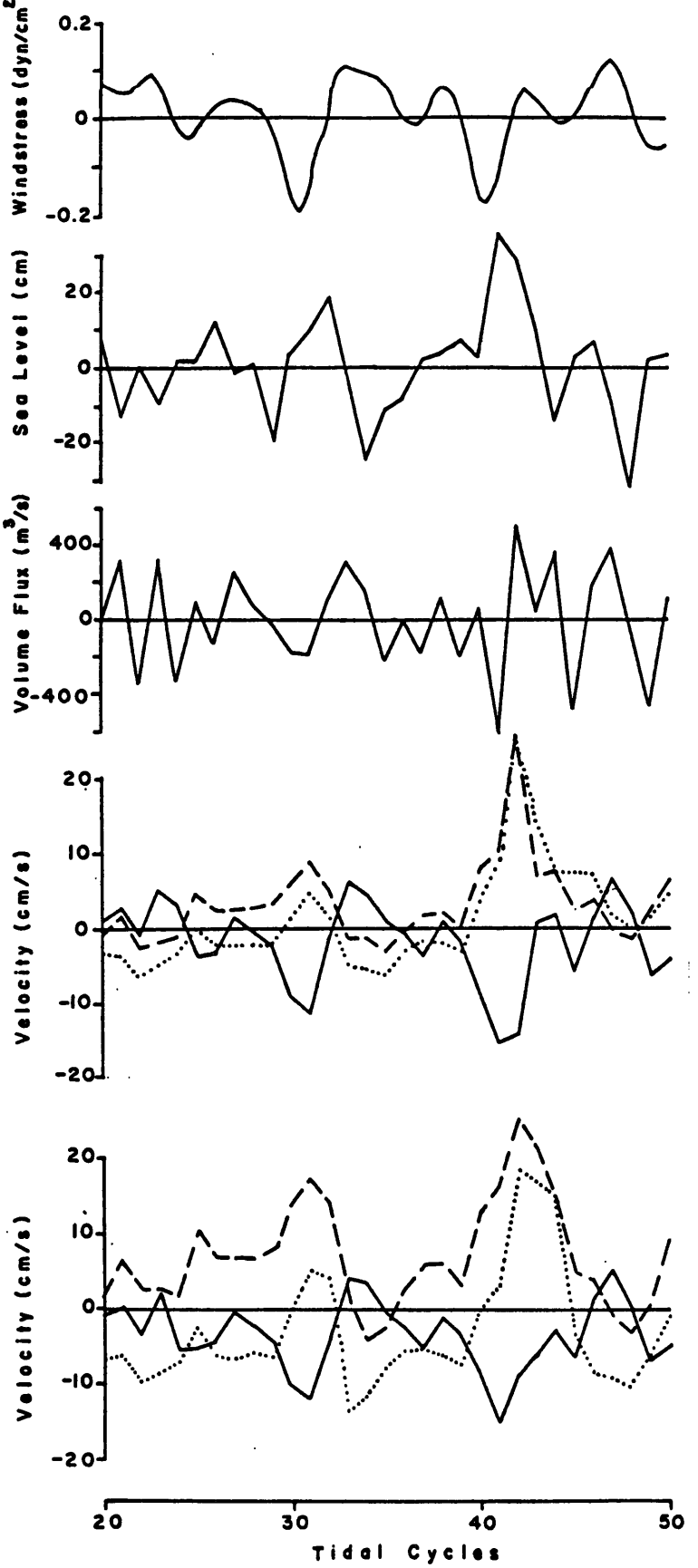

Figure 15: Same as figure 14, for high flow conditions.

The results of numerical modeling presented in Sections 3 and 4 lead us to the following conclusions concerning changes in circulation within the Baltimore Harbor system resulting from channel enlargement.

A change in channel depth from 12.8 m to 15.3 m will increase the magnitude of circulation in Baltimore Harbor but will not change its pattern. This conclusion applies if the channel is maintained at a uniform depth in the longitudinal direction, without sills or barriers.

The three-layered density driven flow, which for timescales longer than 5 to 10 days is a major circulation component within the Baltimore Harbor system, will be enhanced by channel enlargement. The enhancement is due to combined effects of increased water column depth plus the accompanied increase in bottom water salinity which is likely to occur as the channel taps deeper waters in adjacent portions of the Chesapeake Bay. This will result in decreased residence (flushing) times within the main stem of the Harbor. The magnitude of the changes for three-layered flow resulting from increased channel depth are best summarized by two parameters: the volume flux (transport) in the out-flowing, mixed layer Q and the harbor residence time T. During the low run-off portion of the hydrological cycle, projected increases in vertical stratification at the Harbor mouth from 1.0 to 4.85 ppt as a result of increased channel depth will cause an increase in volume flux in the mixed layer from approximately 88 m^3/sec to 210 m^3/sec and a corresponding decrease in residence time from approximately 48 to 20 days. During the high run-off portion of the hydrological cycle, a projected increase in vertical stratification at the harbor mouth from 9.5 to approximately 15.75 ppt as a result of increased channel depth, will cause an increase in volume flux in the mixed layer from approximately 300 m^3/sec to 500 m^3/sec, and a corresponding decrease in the residence time from approximately 15 days to 8 days.

Calculations including with time variable meteorological forcing indicate that the channel enlargement will increase the magnitude of this component of the circulation by a small amount--at the 10% level, approximately. Dredging will influence the magnitude of this component in the circulation only; there is no indication of any change in the qualitative character of the circulation as a result of the channel enlargement.

References

Boicourt, W.C.; Olson, P. A hydrodynamic study of the Baltimore Harbor System; Pt. 1: Observations on the circulation and mixing in Baltimore Harbor. Chesapeake Bay Inst. Tech. Rep.; 1982.

Cronin, W.B.; Pritchard, D.W. Additional statistics on the dimensions of the Chesapeake Bay and its tributaries: cross-section widths and segment volumes per meter depth. Chesapeake Bay Inst., Special Report 42, The Johns Hopkins University; 1975.

Elliott, A.J.; Wang, D-P; Pritchard, D.W. The circulation near the head of Chesapeake Bay. J. Mar. Res. 36: 643-655; 1978.

Garland, C.F.; A study of water quality in Baltimore harbor. Maryland Board of Natural Resources, Rep. 96: 132pp; 1952.

Granat, M.A.; Gulbransen, L.F. Baltimore Harbor and Channels Deepening Study; Chesapeake Bay Hydraulic Model Investigation. U.S. Army Engineer Waters Experiment Station, Technical Report HL-81; 1981.

Grano, Vincent. On the non-tidal circulation of the Upper Chesapeake Bay. Ph.D. Thesis, The Johns Hopkins University; 1982.

Hachey, H.B. Movements resulting from mixing of stratified water. Journal Res. Bd. Canada 1: p. 133; 1934.

Hansen, D.V.; Rattray, M. Jr. Estuarine circulation induced by diffusion. J. Mar. Res. 30: 281; 1972.

Hansen, D.V.; Festa, J.F. Inlet circulation reduced by mixing of stratified water masses. Rapp. Pv. Cons. int. Explor. Mer.: 163-170; 1974.

Long, R.R. Three-layer circulation in estuaries and harbors. J. Phys. Ocean. 7: 415-421; 1977.

Olson, Peter; Boicourt, W.C.; Najarian, T.O. A Hydrodynamic Study of the Baltimore Harbor Circulation. Chesapeake Bay Inst. Tech. Rep.; 1982.

Phillips, O.M.; Shyu, J-H; Salmun, H. An experiment on boundary mixing: mean circulation and transport rates. J. Fluid Mech. 173: 473-499; 1987.

Pritchard, D.W.; Carpenter, J.H. Measurements of turbulent diffusion in estuarine and inshore waters. Bull. Inter. Assoc. Sc. Hydrol. 20: 37-50; 1960.

Stroup, E.D.; Pritchard, D.W.; Carpenter, J.H. Ches. Bay Inst. Tech. Rep. 27, Ref. 61-5: 77 pp.

Wang, D-P. Subtidal sea level variations in the Chesapeake Bay and relations to atmospheric forcing. J. Phys. Ocean. 9: 413-421; 1979a.

Wang, D-P. Wind-driven circulation in the Chesapeake Bay, Winter 1975. J. Phys. Ocean. 9: 564-572; 1979b.

Wang, D-P; Elliott, A.J. Non-tidal variability in the Chesapeake Bay and Potomac River: Evidence for non-local forcing. J. Phys. Ocean. 8: 225-232; 1978.

Wang, D-P.; Kravitz, D.W. A semi-implicit two dimensional model of estuarine circulation. J. Phys. Ocean. 10: 441-454; 1980.

Wilson, R.E. A study of the dispersion and flushing of water-borne materials in the Northwest Branch of Baltimore Harbor, and the relationship between these physical processes and the water quality of the inner harbor. Ches. Bay Inst. Tech. Rep. 64, Ref. 70-3: 101p.; 1970.

Understanding the Estuary: Advances in Chesapeake Bay Research. Proceedings of a Conference. 29-31 March 1988. Baltimore, Maryland. Chesapeake Research Consortium Publication 129. CBP/TRS 24/88.

A Numerical Investigation of Circulation and Salt Distribution in the Patuxent River Estuary

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1. INTRODUCTION

In this study we apply a two dimensional numerical model of laterally averaged circulation and salinity to the Patuxent River estuary, shown in figure 1. Both steady state and time variable simulations are made to determine the interaction between salinity gradients, astronomical and meteorological tides, river discharge and surface wind stress and their effects on the circulation.

In comparison with other Chesapeake Bay tributaries, the Patuxent River estuary exhibits large variations in channel depth. Figure 2 is a longitudinal profile of the estuary from its intersection with the Bay upstream 80 km to Nottingham, MD 65 km from cross section 0. The bathymetry consists of two basins, each bounded on its downstream side by a shallow sill. The sill separating the lower basin from the main stem of Chesapeake Bay is 10 m deep; the average lower basin depth is approximately 18 m and includes a 43 m deep hole. The inner sill separating the upper and lower basins is approximately 3 m deep and the upper basin behind it averages 7 m depth. Circulation and salinity in the Patuxent are strongly influenced by two of the river's bathymetric features: (1) the inner sill, located between Benedict and Lower Marlboro, MD and (2) the 40 m depression located at Pt. Patience. The most striking piece of evidence for topographically controlled hydrodynamics is the presence and persistence of a nearly vertical salt front located about the inner sill. Throughout the lower part of the estuary from the surface to approximately 10 m depth, the isohalines slope upstream in the manner characteristic of two-layer circulation in a partially mixed estuary. But beginning near cross section 20 a pattern of vertical isohalines is evident in the salinity cross sections from all four seasons. The change in salinity across the front varies from about 9 ppt in the fall to about 4 ppt in the spring (figure 3). The width of the front and the fine structure within the frontal region are not well resolved by the Maryland Office of Environmental Programs seasonal surveys (which report salinity profiles at 5 km intervals).

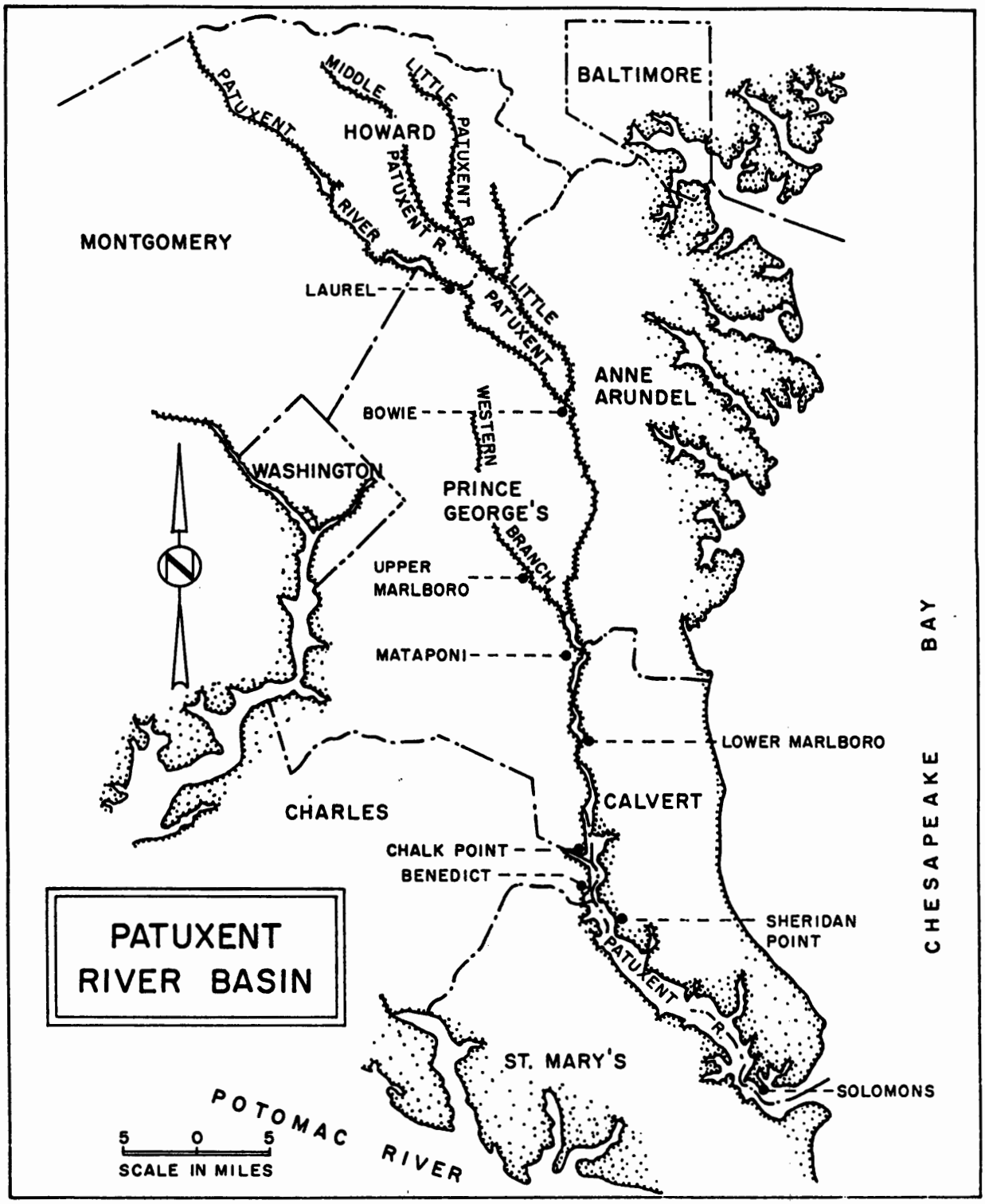

Figure 1: The Patuxent River basin. The estuarine portion begins about 5 km above Mataponi. The tidal limit is a few kilometers below Bowie.

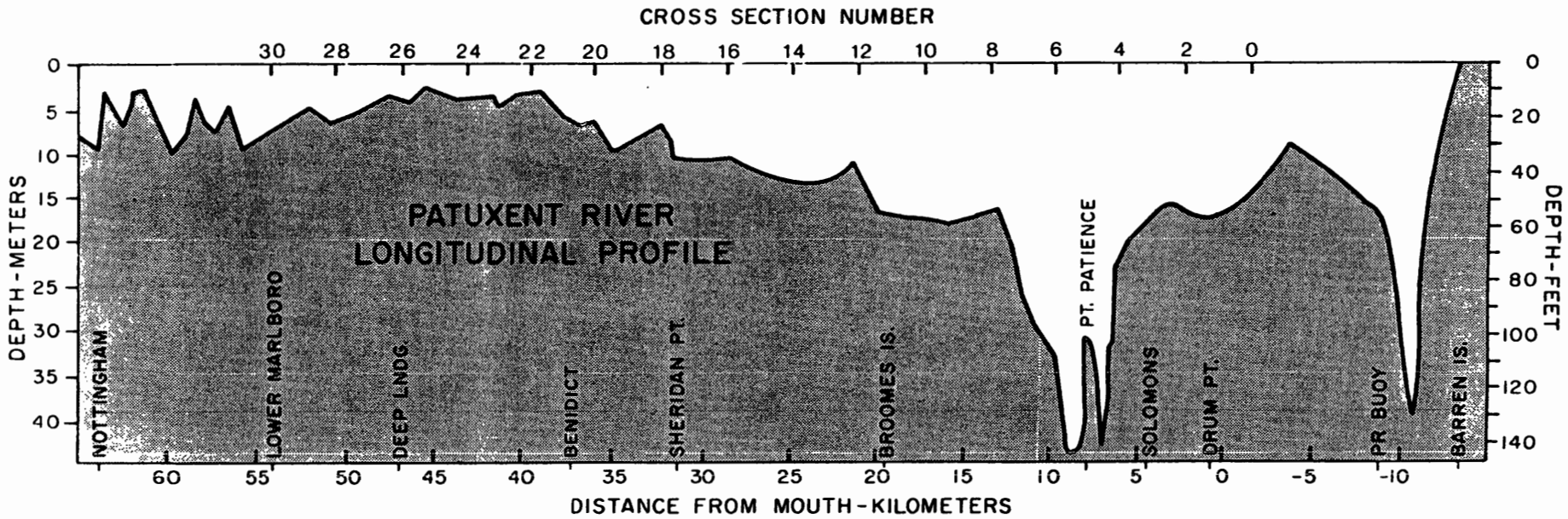

Figure 2: The longitudinal profile of the lower portion of the Patuxent River estuary. Cross section number refers to the tabulation by Cronin and Pritchard (1975). River kilometer scale (lower scale) is used in this report.

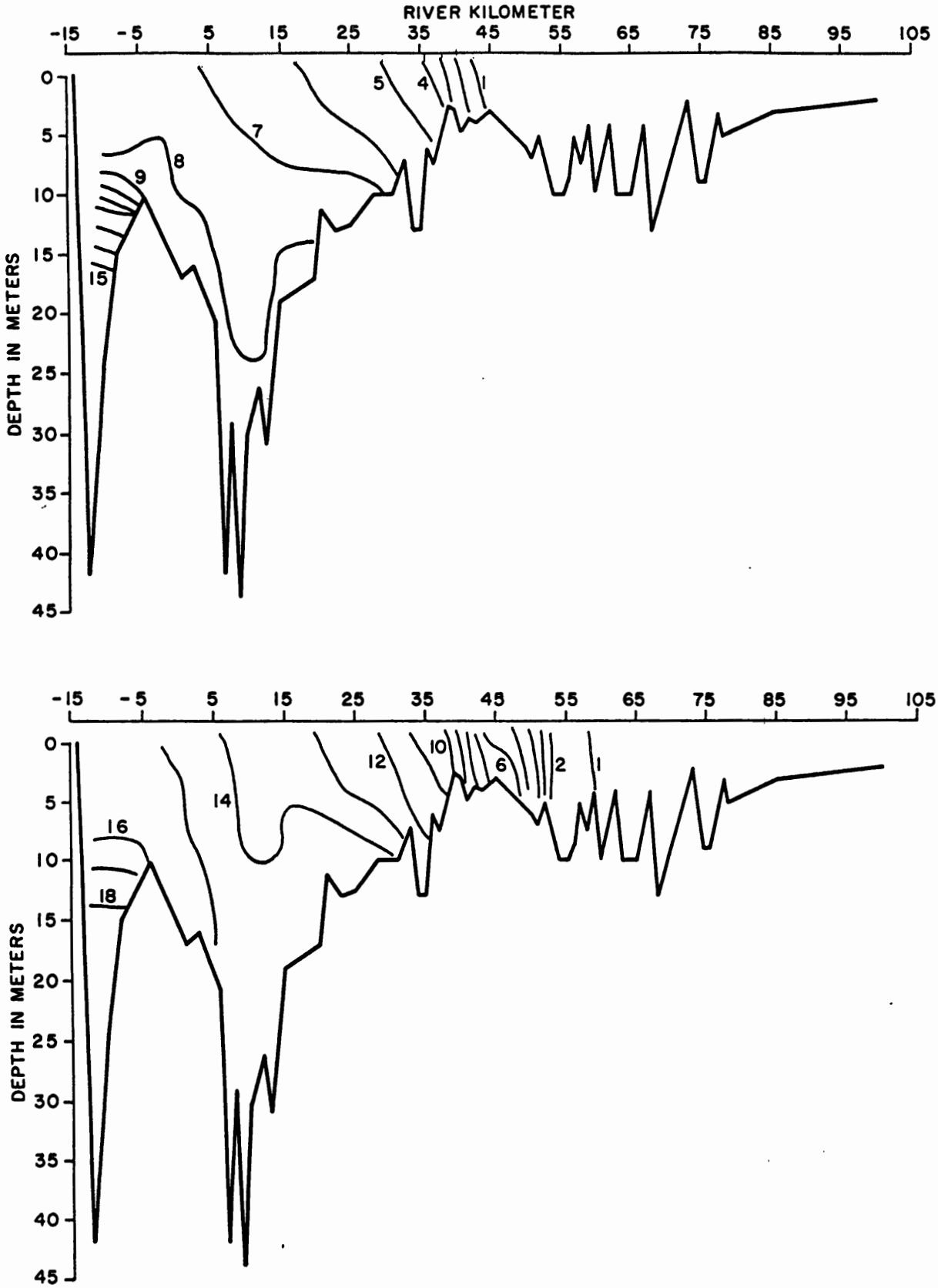

Figure 3: Patuxent River estuary salinity cross sections, (a) seasonally averaged for spring 1984, from OEP Tech Rep. #7, (b) seasonal average for fall 1984, from OEP Tech Rep. #7.

Topographically controlled frontal structures are known to be important in estuaries (Huzzey (1982); Hibiya (1986)). Similarly, permanent small scale fronts are often present on continental shelves, especially near the mouth of large estuaries (Garvine, 1974, 1979). It is certain that the interaction between the upper and lower basins in the Patuxent is strongly affected by the dynamics in the frontal region.

The dynamic conditions within the deep hole at Pt. Patience are also not well understood. On the basis of available data (the maximum depth of OEP surveys is 23 m), one would infer that the circulation within the deep hole has a seasonal variability. In spring and winter and probably during most of the fall, the deep hole is only weakly stratified to 23 m depth. This indicates that there is a substantial circulation to at least 23 m for 8 months of the year or more. In the summer the OEP 1984 (figure 15a) survey (in this case, data from a single cruise, rather than a seasonal average) showed a well-developed halocline in the interval between 5 and 10 m. The halocline intersects the outer sill and the upstream wall of the lower basin, in effect putting a lid on the deep hole. In summer, the deep hole may at times become unventilated and practically stagnant.

A variety of modelling approaches have been used on coastal plain estuaries, from simple one dimensional, semi-analytical models of salt wedge dynamics (Prandle, 1981; Oey, 1984) to laterally-averaged two dimensional models such as used in this study (Festa and Hansen, 1976, 1978; Blumberg, 1978; Wang and Kravitz, 1980) to fully three dimensional models (Oey, Melor and Hires, 1986; Gordon and Spaulding, 1987). At the present time, 3-D models are used in situations where the density-driven component of the circulation is subordinate to wind or tidally induced flow. In those circumstances, the horizontal pattern of depth-averaged flow can be solved without reference to the density distribution, and the vertical structure of the flow can then be determined locally at each grid point. However when density gradients are as important in driving the circulation as tide and winds, as is the case in the Patuxent River estuary, depth averaging does not work. In this case it is more appropriate to lateral average, as we have done. Having computed lateral averages, it is possible to then compute the flow in lateral sections, using the equations of motion for that plane. We have not pursued such an approach for modelling the Patuxent, because we judge lateral flows to be of secondary importance in this case.

Most of the earlier two-dimensional models were designed for steady state conditions, or for simulating slowly evolving flows associated with variations in boundary salinity or river discharge. However, it has been demonstrated throughout the Chesapeake Bay system that the circulation is strongly time variable, not only from astronomical tides, but also in the subtidal frequency band, 0.1-1.0 cpd (Wang, 1979; Elliott, 1978; Wong and Garvine, 1984; Olson, 1986). The subtidal variability is due to the combined effects of meteorological tides (non-local forcing) and locally applied surface wind stress (local forcing). These components often dominate the circulation on timescales shorter than about 5 days. Only when the circulation is averaged over time intervals of 10 days or more does the density driven flow prevail. The measurements reported by Boicourt and Sanford, 1987 confirm this to be true for the Patuxent River estuary as well. It is necessary to resolve frequency components of the circulation ranging from the semi-diurnal tide at the high end, to seasonal variability at the low end, in order to construct an accurate picture of dynamics in the Patuxent.

2. NUMERICAL MODEL

The model geometry has its origin at mean sea level at the head of the estuary, oriented with x,y,z-axes in the downstream, cross stream and vertical directions respectively. The maximum channel depth in any lateral section is denoted by H, and the sea surface elevation, relative to its mean value, is denoted by η. The laterally-averaged velocity components in downstream and vertical directions are u and w, respectively. Channel geometry is specified by the width function

B(x,z). The laterally-averaged shallow water equations of motion are as follows. The momentum balance in the vertical direction is hydrostatic, and the other governing equations are

$$\frac{\partial}{\partial x}(uB)+\frac{\partial}{\partial z}(wB)=0 \tag{1}$$

$$\frac{\partial}{\partial t}(uB)+\frac{\partial}{\partial x}(uuB)+\frac{\partial}{\partial z}(uwB)+gB\frac{\partial \eta}{\partial x}=$$

$$\frac{gB}{\rho_o}\int_o^z \frac{\partial \rho}{\partial x}dz'+\frac{\partial}{\partial x}\left(BN_x\frac{\partial u}{\partial x}\right)+\frac{\partial}{\partial z}\left(BN_z\frac{\partial u}{\partial z}\right)-C_D u|u|\frac{\partial B}{\partial z} \tag{2}$$

$$\frac{\partial}{\partial t}(SB)+\frac{\partial}{\partial x}(SuB)+\frac{\partial}{\partial z}(SwB)=\frac{\partial}{\partial x}\left(BK_x\frac{\partial S}{\partial x}\right)+\frac{\partial}{\partial z}\left(BK_z\frac{\partial S}{\partial z}\right) \tag{3}$$

$$\rho=\rho_o(1+\beta S) \tag{4}$$

representing conservation of mass, horizontal momentum and salinity, plus a linear equation of state. Here t is time, S is salinity, ρ is density and g is gravity. The parameters N_x and N_z are horizontal and vertical turbulent viscosities, and K_x and K_z are turbulent mixing coefficients for salt in the horizontal and vertical directions respectively. The boundary drag coefficient is C_D, and in the equation of state ρ_o denotes fresh water density and β is the coefficient of saline contraction, 7.29×10^{-4}, ppt^{-1}.

The appropriate boundary conditions are the following. On the channel bottom, z=-H, the normal velocity and salinity flux both vanish. In terms of the local channel bottom slope, these are

$$w-\frac{\partial H}{\partial x}u=0, \qquad z=-H \tag{5}$$

and

$$\frac{\partial S}{\partial z}-\frac{K_x}{K_z}\frac{\partial H}{\partial x}\frac{\partial S}{\partial x}=0, \qquad z=-H, \tag{6}$$

accurate to the order of the bottom slope. The bottom stress condition is, to the same order

$$N_z\frac{\partial u}{\partial z}=C_D u|u|, \qquad z=-H. \tag{7}$$

Since the slope of the sea surface is small compared to the bottom slope, it is neglected in applying the upper surface conditions. The surface equivalents of (5) and (6) are then

$$w=\frac{\partial S}{\partial z}=0, \qquad z=\eta \tag{8}$$

and the surface stress condition is

$$N_z\frac{\partial u}{\partial z}=\tau(x,t), \qquad z=0 \tag{9}$$

where τ is the laterally averaged longitudinal component of the wind stress.

Entrance and exit conditions are specified as follows. At the head of the estuary, the Patuxent River freshwater discharge Q_R is specified as a time series and distributed uniformly through the water column as a horizontal current u_R, satisfying the condition

$$\int_{-H}^{\eta} B u_R \, dz = Q_R(t), \qquad x = 0 \tag{10}$$

and the salinity is set to zero

$$S = 0, \qquad x = 0 \tag{11}$$

At the estuary mouth, x=L, where the Patuxent joins the main stem of Chesapeake Bay, the exit velocity condition is just

$$\frac{\partial}{\partial x}(uB) = 0 \qquad x = L \tag{12}$$

while the sea surface conforms to the sea surface of the Bay. This condition is expressed in terms of a time series η_m (t) for the Bay surface elevation

$$\eta = \eta_m \qquad x = L \tag{13}$$

Conditions on salinity at the mouth are constructed to simulate the exchange processes between the Patuxent and the main stem of the Bay. If the transport is out of the estuary, we prescribe a purely advective balance

$$\frac{\partial}{\partial t}(SB) = -\frac{\partial}{\partial x}(uSB), \qquad x = L, if \quad u > 0 \tag{14}$$

When the transport direction changes, we expect there to be a time interval during which water from the preceding outflow is carried back into the Patuxent. As inflow continues, this pool is exhausted and Chesapeake Bay water begins to enter. This sequence of events is represented by a relaxation condition of the form

$$\frac{\partial s}{\partial t} = (S_m - S)/t^*, \qquad S = S_c \quad at \quad t = t_c; \qquad x = L, \tag{15}$$

where t_c is the time of current direction change, t* is the relaxation time constant, an adjustable parameter, S_c is mouth salinity at t_c, and S_m is the Bay salinity. The data required to initialize the calculation are an initial salinity distribution and sea surface elevation pattern (relative to mean water level).

The model contains five friction and mixing parameters, the boundary drag coefficient C_D, plus mixing coefficients for turbulent momentum, N_z and N_x, and salinity S_z and S_x. Estimates of drag coefficient values lie in the range 1-2 x 10^{-3} (Phillips, 1980); and we choose C_D = 1.6 x 10^{-3}.

The largest source of error in estuarine modeling comes from trying to parameterize the turbulent mixing of salt and momentum. The mixing parameters are known to have a functional dependence on the Richardson number

$$Ri = -g\beta \frac{\partial s}{\partial z} \Big/ \left(\frac{\partial u}{\partial z}\right)^2 \tag{16}$$

The exact functional relation, however, between the mixing parameters and the Richardson number is not known. An alternative method, and that used in this study, is to assign values to these parameters based on direct comparison with observed field measurements.

In partially mixed estuaries such as the Patuxent, the most important of the mixing parameters are K_z and N_z, controlling the vertical flux of salt and momentum, respectively. Often in estuaries the horizontal mixing coefficient for salt K_x is unimportant; however in the Patuxent this is not the case, as it controls the structure of the salinity front. By contrast, the horizontal mixing coefficient N_x is dynamically

insignificant; usually it is assigned a nominal value large enough to allow the horizontal dispersion term to contribute a small amount to the momentum balance, for purposes of numerical stability.

The vertical eddy viscosity, N_z=5.0 $cm^2\ s^{-1}$, was selected on the basis of comparisons between observed and calculated velocity profiles and results of a similar study of the Baltimore Harbor. Preliminary solutions with horizontal eddy viscosity, $N_x < 1.5 \times 10^6$ $cm^2\ s^{-1}$ were unstable to numerical oscillations. The value of $N_x = 2 \times 10^6$ $cm^2\ s^{-1}$ was chosen to prevent numerical instabilities from developing.

The mixing parameters for salt, K_x and K_z, were chosen on the basis of a sensitivity analysis. The salt front is a first order feature of the Patuxent River, appearing with only minor deviations in position and gradient in all four of the 1984 OEP seasonally averaged surveys. The position and gradient of the front is one piece of data we require our model to successfully simulate and is the basis of the sensitivity analysis. With an unsmoothed geometry file any value $K_x \leq 1.25 \times 10^6$ $cm^2\ s^{-1}$ proved to be unstable to numerical oscillation. The calculated front, after seven days, with $K_x = 1.25 \times 10^6$ $cm^2\ s^{-1}$, is more diffuse, $\Delta s/\Delta x$ = .27 ppt/km than the observed front, $\Delta s/\Delta x$ = .4 ppt/km. Clearly such large values of K_x do not accurately model the front. In an attempt to stabilize the solution, the geometry file was run through a 3 point smoothing routine. With a smoothed version the lower limit on K_x reduced to $K_x \geq 7.5 \times 10^4$. A constant value of $K_x = 1.25 \times 10^5$ was adopted for the numerical experiments. The value chosen for vertical mixing K_z = 0.2 $cm^2\ s^{-1}$ produces solutions which match observed surface salinity and allow for vertical structure in isohaline patterns.

Equations (1) through (15) are solved on two-dimensional grids using finite differences. The time-stepping procedure is the semi-implicit method used by Wang and Kravitz (1980), in which the sea surface height is advanced with an implicit time step, while the interior variables (salinity, horizontal and vertical velocity) are advanced using an explicit time step. The details of the computational algorithm can be found in Wang and Kravitz, 1980. Diffusion terms are represented by central differences, advection terms by upwind differences. We found that it is necessary to use very small time steps, on the order of 100 s, to insure numerical stability.

In order to include in a realistic way the extreme variations in channel depth, we have used the sloping bottom approximation. Let Δz and Δx denote the grid spacing in the vertical and horizontal directions, respectively. According to this approximation, variations in channel depth are represented by increasing or decreasing the number of grid points by one in neighboring columns. The local channel slope is then either $\pm \Delta z/\Delta x$ or zero. We have constructed a grid with resolutions (grid spacings) of 2 m in the vertical and 640 m in the horizontal, so that the inclined segments have slopes of $\pm$0.003125. This permits us to simulate all the major topographic structures in the Patuxent, including the deep hole at Pt. Patience, where the grid extends to 30 m depth. The slope of the model topography closely approximates the average gradients found in the estuary. In order to correctly apply horizontal differential operators at these points, we have inserted image or "buffer" points onto the grid rows within sloping sections.

The bathymetric grid used in this study consists of 164 columns of w and S points and 103 columns of u points, extending over 104 km (Δx = 638 m). Vertical resolution is 2 m. In the model the estuary head is at river kilometer 100, at the US Rt. 50 bridge crossing, near the tidal limit. The mouth is defined at river kilometer -4, about 4 km downstream from Drum Pt., approximately 8 km downstream from Solomons Is. Major topographic structures incorporated into the model grid include: (1) the outer sill at the model-defined mouth; (2) the deep hole at Pt. Patience (to 30 m); (3) the inner sill above Benedict; (4) the upper basin around Lower Marlboro; and (5) the narrow and shallow upper reach above Jug Bay.

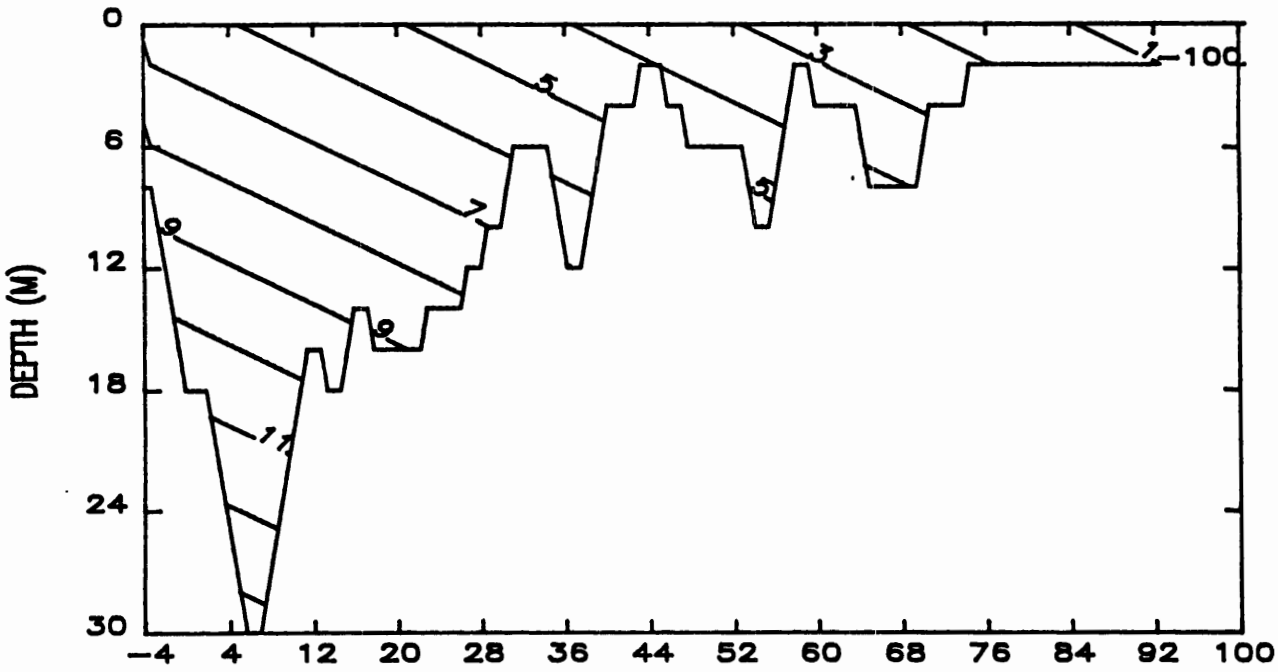

Figure 4a: The initial salinity profile for spring 84 steady state simulation. Distribution is uniformly stratified. The contour interval is 1.0 ppt. Right is estuary head, left is estuary mouth. Positive is left and down.

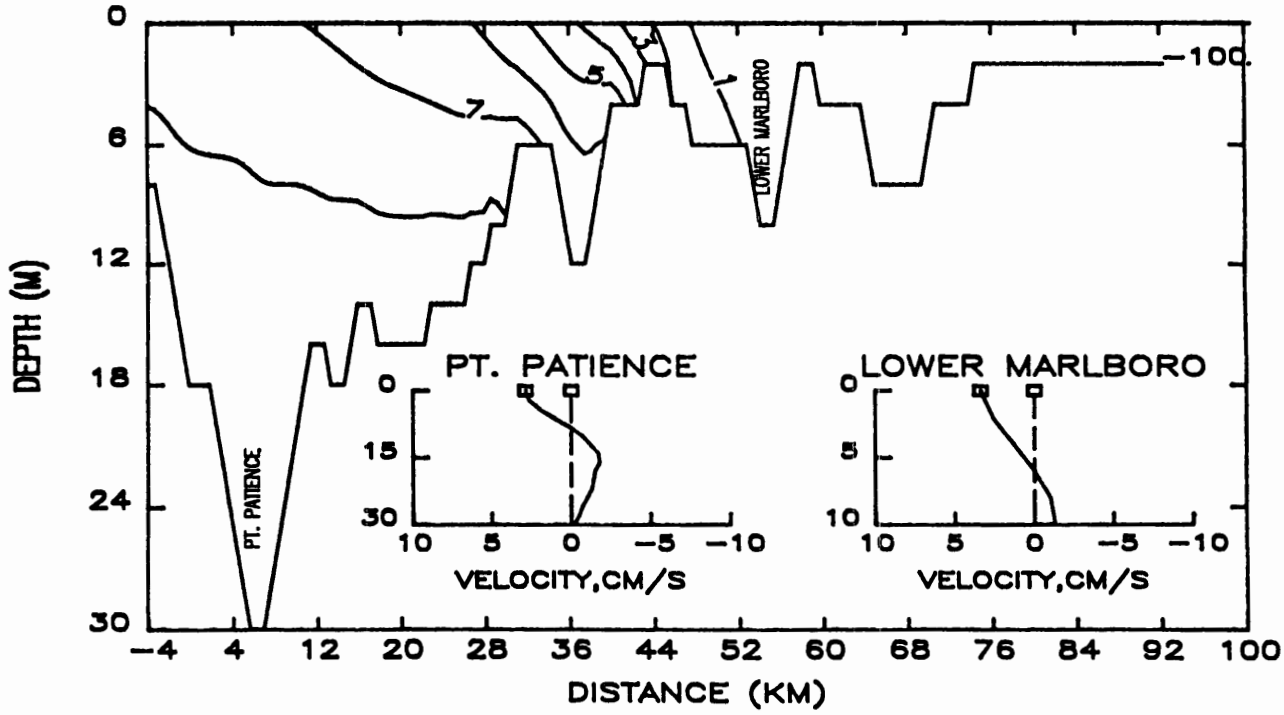

Figure 4b: Contours of salinity and velocity profiles 520 hours into the run. River flux and mouth salinity held constant. No tidal or wind forcing. Coefficients $K_x = 1.25 \times 10^5$, $K_z = .2$ $N_x = 2 \times 10^6$, $N_z = 5.0$.

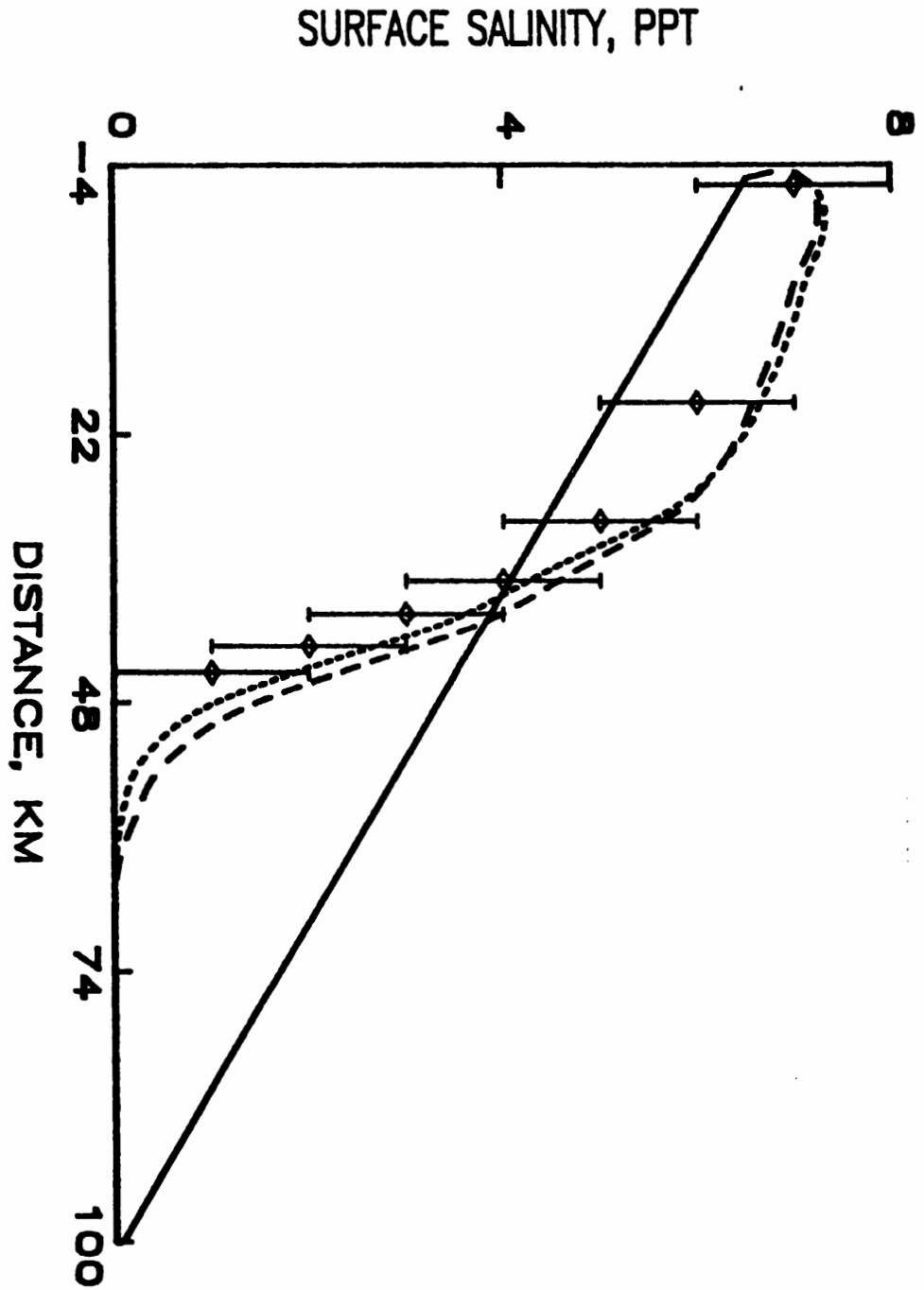

Figure 5: Comparison of observed spring 1984 seasonally averaged surface salinity, plotted diamonds with ± 1 ppt error bars, vs. calculated solutions for spring steady state run. Shown is the progression from initial, linear profile, solid line, to the solution at 400 hours, long dashed line, to the steady state profile at 520 hours, short dashed line.

3. STEADY STATE CALCULATION

A steady state simulation of circulation and salt distribution within the Patuxent River estuary has been run for the purpose of modeling the gross features of the spring 1984 seasonally averaged salinity profile (figure 3a). The most prominent feature of the Patuxent River salinity distribution is the salt front located near the inner sill, between Benedict and Lower Marlboro. The goal of this simulation is to produce the frontal structure from random initial conditions.

To achieve a steady state solution a constant river flux $Q = 20\ m^3/s$ was imposed at the head of the estuary. A salinity profile, increasing linearly from 7 ppt at 1 meter depth to 9 ppt at 9 meters depth was imposed at the mouth of the estuary. The values were taken from spring 1984 seasonally averaged data. Figure 4a shows the uniformly stratified initial salinity distribution. The salinity gradient across the front is .06 ppt/km. Figure 5 illustrates how the model accurately predicts both the position and gradient of the salt from random initial conditions. The solid line shows the linear surface salinity profile at t=0. The 1 ppt isohaline is initially at kilometer 86. After 400 hours the 1 ppt isohaline has moved to 50 km and the frontal gradient is .27 ppt/km. After 520 hours the solution has reached a steady state. The 1 ppt isohaline is at 48 km and the frontal gradient is .29 ppt/km which compares well with the observed value of .3 ppt/km. Calculated surface salinities fall within the ± 1 ppt error bars of all the observed data points. Figure 4b shows the steady state salt distribution and velocity profiles within the upper and lower basins. The velocity profiles at Lower Marlboro and Pt. Patience have remained invariant over the past 100 hours. The steady state profile at Lower Marlboro is 3 cm/s surface outflow balanced by a -1.7 cm/s inflow. At Pt. Patience values for surface outflow and bottom inflow are 3.0 cm/s and -2.0 cm/s respectively.

The calculated steady state salinity distribution for the spring of 1984 compares well with the observed seasonally averaged profile shown in figure 3a. The salt front in both profiles is centered over the inner sill. The observed salinity gradient across the front is $\Delta s/\Delta x \approx .3$ ppt/km which is very close to the calculated value of .29 ppt/km. The steady state simulation accurately models the gross salinity features of the estuary given random initial conditions. The predicted circulation is a two layer shear flow with an average 3.0 cm/s surface velocity and a -2 cm/s bottom velocity.

4. TIME VARIABLE CALCULATIONS

Two intensive survey periods were selected to study time variable circulation under two distinct forcing regimes. The spring survey covers the time interval May 15 - June 15, 1986 when river flow is high and the estuary is less saline. The second, or fall, survey runs between September 3 - September 30, 1986 and is characterized by lower river flow and higher salinities in the adjacent Chesapeake Bay. During each survey period tidal heights were measured from the N.O.S. gauge located at Solomons Is. River flow data was collected from the U.S.G.S. stream gauge located at the Rt. 50 bridge. Time variable mouth salinity was taken from data collected by Boicourt and Sanford, 1987 at station PL 1, located just upriver from Drum Point (see figure 2). Hydrographic surveys and moored current meters provide measurements of salinity distribution and velocity profiles respectively.

The spring period can be divided into 3 regimes based on mouth salinity and river flow. The mouth stratification expressed as $\delta s = s_B - s_S$, where s_B is mouth salinity in ppt at 9 meters depth and s_S is mouth salinity in ppt at the surface, determines the strength of density driven circulation. The first 10 days are characterized by strong stratification, $\delta s \approx 1$ ppt, and weak river flow, the second 10 day period by both weak stratification, $\delta s \approx 0$, and river flow and the last 10 days by strong stratification, $\delta s \approx 2$, and river flow. The initial salinity distribution was derived from the May 9 slack water survey.

The simulation begins at midnight on May 15, 1986 and ends at 744 hours on June 16, 1986. During this period direct comparisons between observed and calculated salinity fields can be made on May 21, June 3 and June 9. Figure 6a is a plot of instantaneous surface salinity at various times within the May 21 tidal cycle. The tidal excursion in figure 6a ranges from a minimum of 5 km near the mouth to a maximum 10 km near the inner sill. When comparing calculated, tidally averaged solutions with observed data, it is important to consider this tidal excursion because a data collection survey may take up to 6 hours to complete or 1/2 the period of an astronomical tide. During this time isohalines are being advected with the tidal flow. If the survey on May 21 began at the mouth, when the tidal height was .23 meters, and ended six hours later when the tidal height was -.15 meters, the calculated solution would exactly match observed data. A second comparison on June 3, 460 hours into the simulation, figure 6b, shows that the tidally averaged solution, though slightly more diffuse than the observed field, lies within the error bars at all but one point. A final comparison is made, figure 6c, on June 9, 20 days into the simulation. Again the solution matches observed values to an acceptable level. The average frontal salinity gradients based on calculated and observed data taken from kilometers 35 and 45 at 5 meters depth are .4 ppt/km and .35 ppt/km. The comparisons indicate the numerical model is accurately predicting the salt balance within the estuary during the month long simulation.

Table 1 lists the range in tidal velocities and the ratio between observed and calculated ranges. It is found that, on average, observed values exceed calculated values by a factor of 1.6. This relationship is expected, however, because current meters, located in the middle of a river measure the maximum of a velocity profile. Calculated velocities are all scaled by estuary width and, therefore, represent lateral averages rather than lateral maximums. Assuming a parabolic distribution, the maximum is 1.5 times the lateral average, which accounts for the difference between calculated and measured values.

Figure 7 is a month long time series of velocity profiles for cross sections at PL1 and PL4 (located just down river from Broomes Is., see figure 2), showing the persistence of a two layer shear flow. The average profile at PL1 is a 5 cm/s surface outflow balanced by a deeper 5 cm/s inflow. Figure 7 also illustrates the dominant driving mechanisms in the estuary by comparing events in the mouth salinity and river flow records against model responses in the PL1 and PL4 velocity profiles. During most of the survey period river flux is small, $\bar{Q}$= 5.0 m^3/s. On June 12, 684 hours into the simulation, river flux increases by more than 100% of the average to 11.2 m^3/s. A pulse in the velocity profile at PL1 is seen after a 12 hour time lag. The response to the flood event is seen throughout the entire PL1 record. Surface velocities increase while deeper return flow stalls.

Comparing time series of mouth salinity and velocity shows that the largest fluctuations in velocity ±4 cm/s, or an 80% fluctuation in mean lower layer velocity, correlate with changes in mouth stratification. Table 2 shows that increases in $\overline{\delta s}$ cause positive deviations in average PL1 velocity, $\bar{v}_{13}$, at 13 meters depth. There is a 36 hour time lag between corresponding peaks in δs and $\bar{v}_{13}$ at station PL1.

The gross circulation patterns within the estuary are best shown using streamfunction contours. Streamfunction distributions, within two of the three δs regimes outlined in Table 2, were calculated by vertically integrating the horizontal velocity field from the expression

$$\psi B = \int_{-H}^{z} UB\,dz' \qquad (17)$$

Figure 8a shows tidally averaged streamfunction on May 20 when $\overline{\delta s}$ = 1.0 ppt and $\bar{Q}$ = 5.0 m^3/s. The most vigorous circulation appears in the lower estuary and is particularly strong near Pt. Patience. The deep hole is being flushed by a deep return flow and is clearly not stagnant.

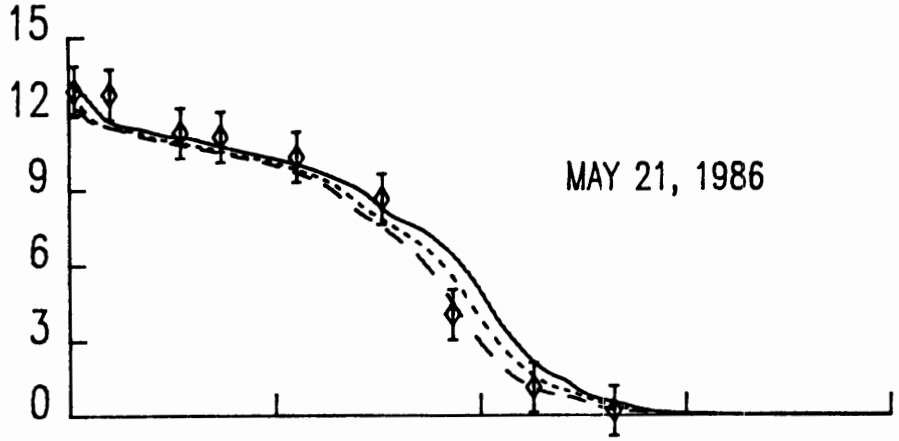

Figure 6a: Plots of instantaneous surface salinity for half a tidal cycle on May 21, 1986 vs. observed data, plotted diamonds with ± 1 ppt error bars. The solutions for this range of tidal heights, .23, -.08 and -.15 meters, plotted as solid, short dashed and long dashed lines respectively, outline the tidal excursion within the estuary. Coefficients same as those in figure 3.

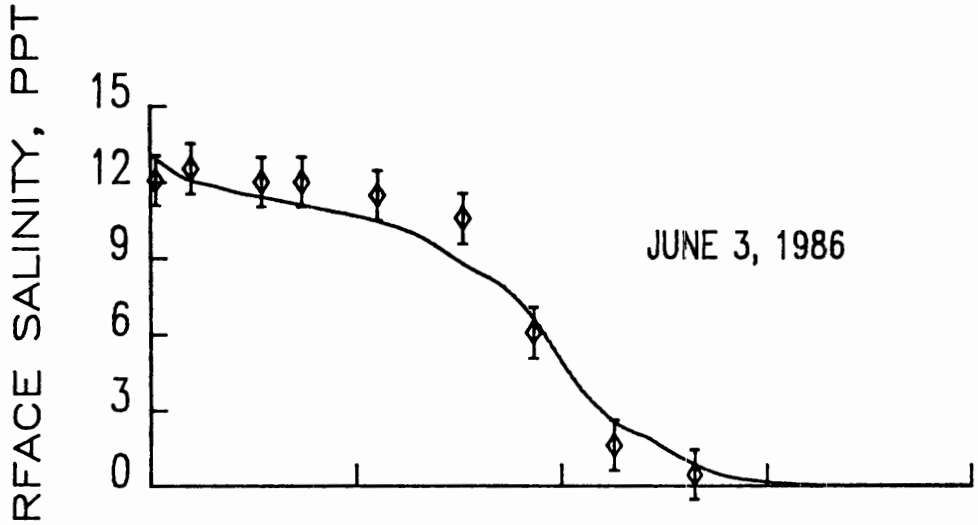

Figure 6b: Tidally averaged surface salinity, solid line, vs. observed data plotted diamonds with ▪ 1 ppt error bars 470 hours into spring run, June 3, 1986.

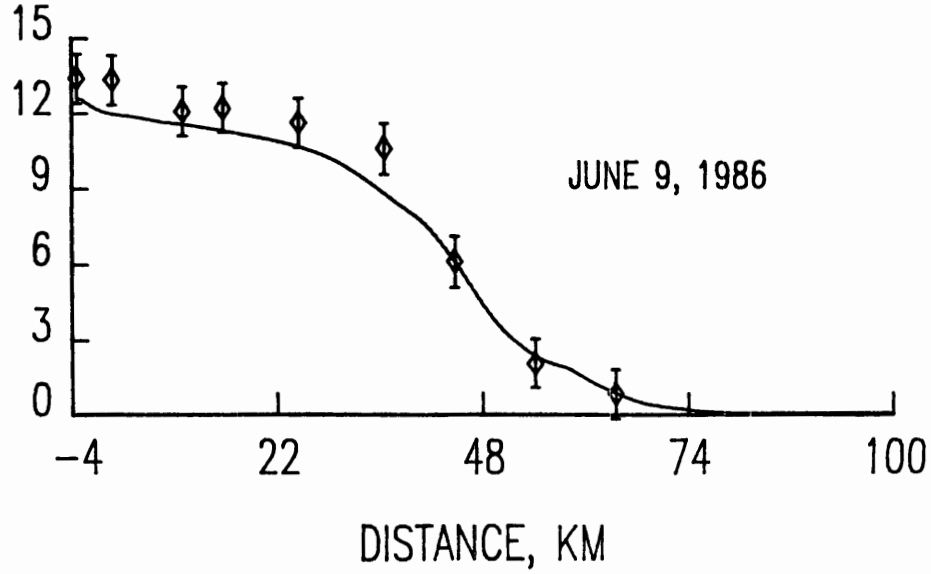

Figure 6c: Tidally averaged surface salinity, solid line, vs. observed data, plotted diamonds with ▪ 1 ppt error bars, on June 9, 613 hours into the spring run.

Table 1 Tidal Velocity Range

Station	Depth (ft)	Observed range (cm/s)	Calculated range (cm/s)	Ratio obs./calc.
PL1	8	43	29	1.5
	45	46	26	1.7
P2	8	26	30	0.8
	25	36	32	1.1
P3	8	67	31	2.1
	80	65	30	1.0
PL4	8	30	30	1.0
	45	28	28	1.6
P5	8	27	24	2.2
P6	28	32	23	1.4
P8	8	60	40	1.5
	22	45	21	1.5
P9	8	92	50	2.1
	15	55	43	1.3

Average correlation factor = 1.58

Table 2 Mouth Salinity Regimes

Time period (hours)	$\overline{\delta s}$ (ppt)	$\bar{v}_{13}$ (cm/s)
0-240	0.96	5.5
240-480	0.1	4.0
480-720	2.0	8.0

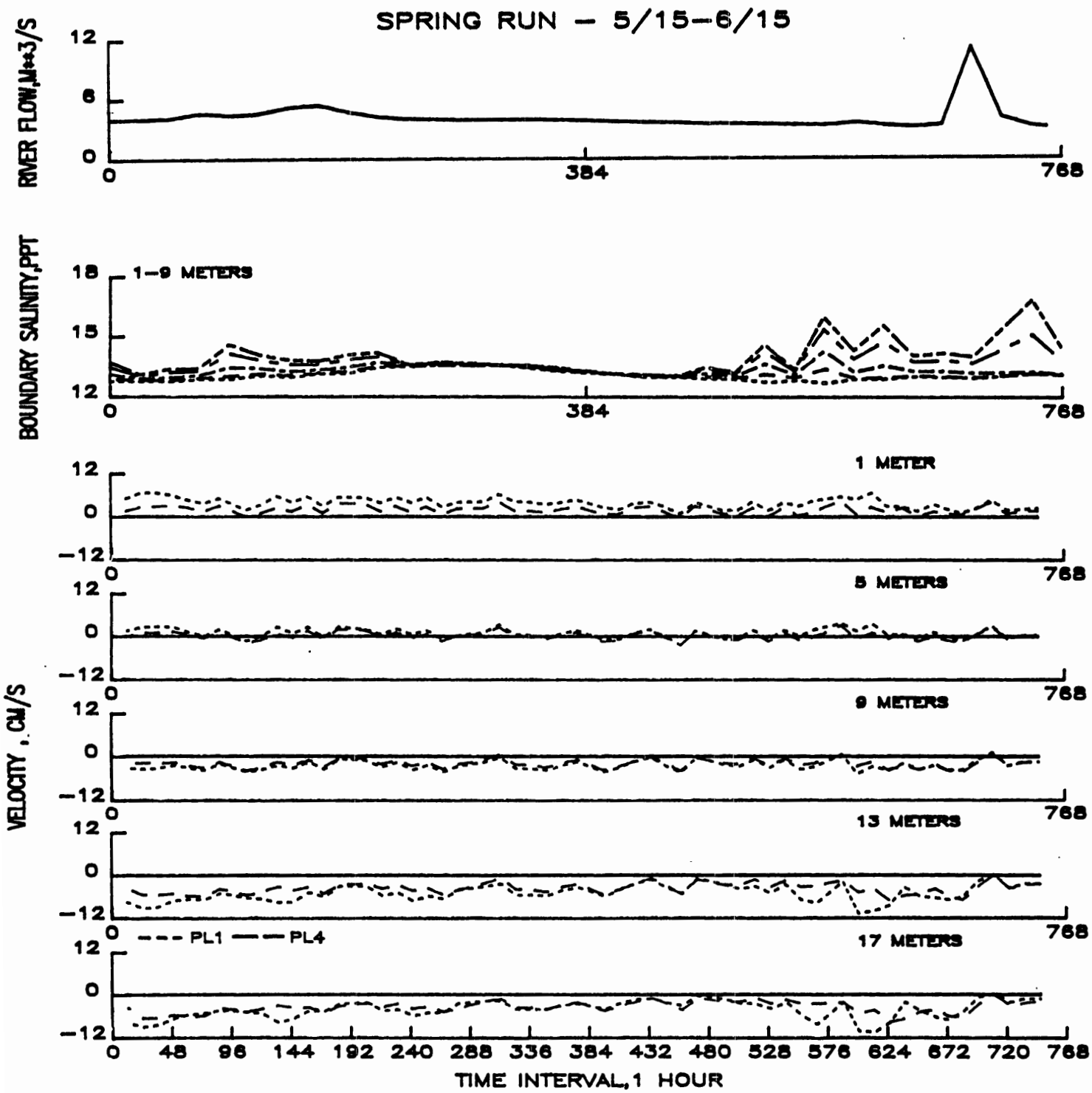

Figure 7: Time series of tidally averaged velocity profile at 1,5,9,13,17 meter depths at stations PL1 and PL4 vs. time series of river flux (m^3/s) and mouth salinity, ppt, for 1,3,5,7, and 9 meters, from spring intensive survey.

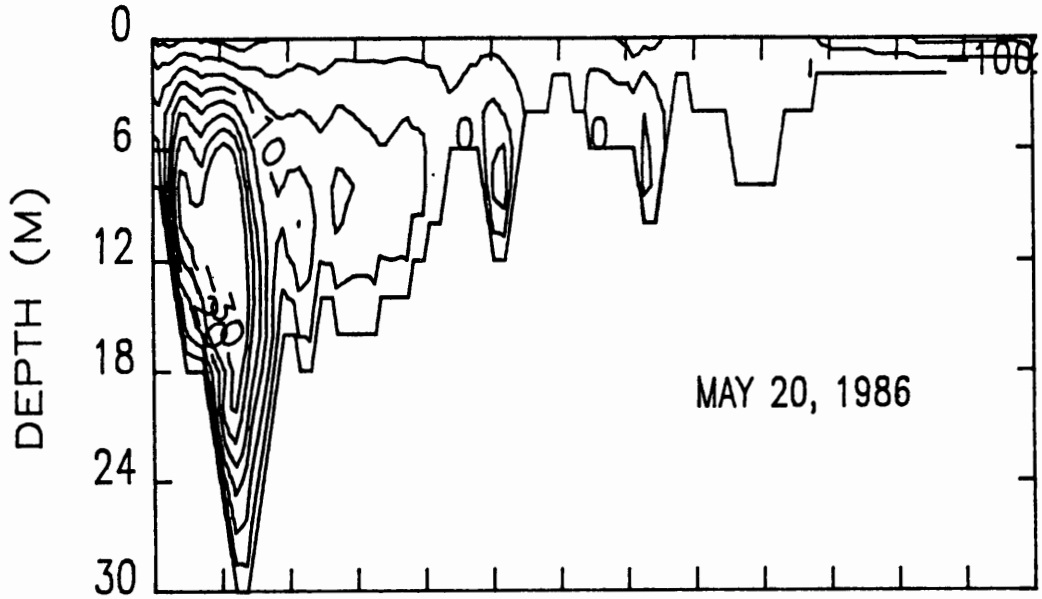

Figure 8a: Contour of tidally averaged streamfunction/1000, cm^2/s, on May 20 during the spring intensive run. Contour interval is 8. Average river flux $\bar{Q}$ = 5 m^3/s and average mouth stratification $\overline{\delta S}$ = 1 ppt.

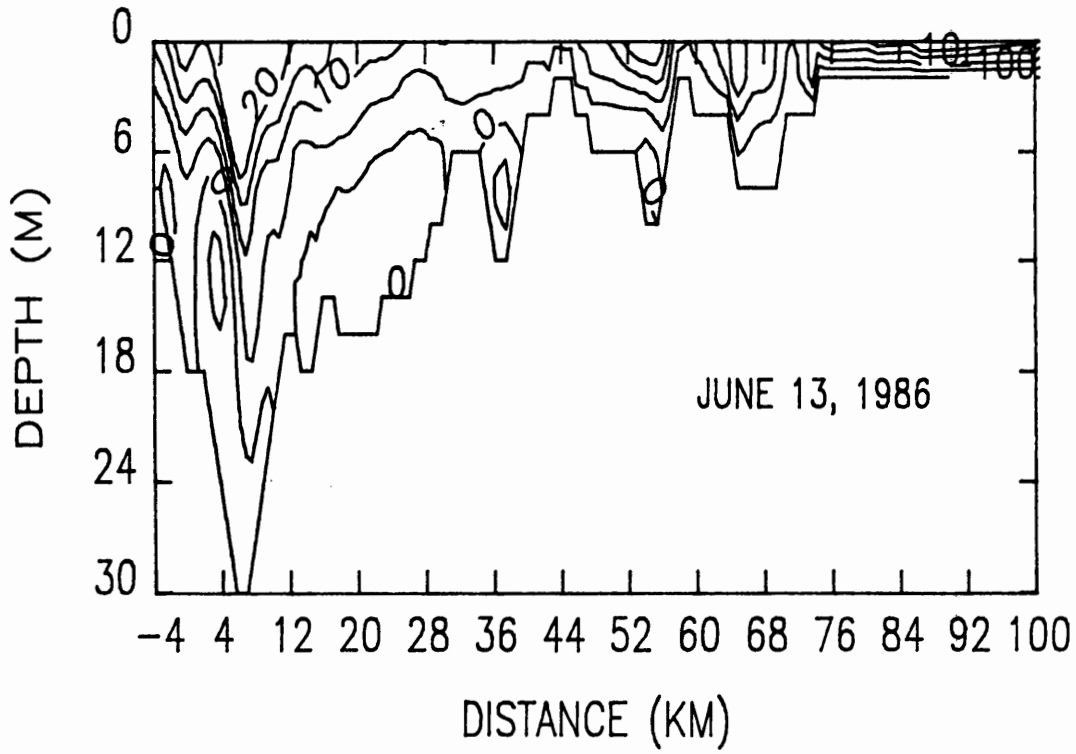

Figure 8b: Contours of tidally averaged streamfunction/1000, cm^2/s, on June 13 during spring intensive run. Average river flux $\bar{Q}$ = 11 m^3/s and average mouth stratification $\overline{\delta S}$ = 2 ppt.

In examining circulation patterns it is interesting to follow the zero streamline near the bottom boundary in the lower estuary. The streamline rises vertically off the bottom near river kilometer 40 and returns, along the surface, to the estuary mouth, indicating the strong density driven circulation is, on average, confined to the lower estuary. Smaller, less vigorous two layer circulation patterns are seen just above the inner sill near Lower Marlboro and further upstream near kilometer 60 where the transition between one and two layer flow occurs.

Figure 8b shows the streamfunction distribution on June 13 during an 11 m^3/s flood event. The density driven flow in the lower estuary is now very weak compared to surface outflow. The circulation is split into two weak cells, one along the upslope between Pt. Patience and Sheridan Pt. and the other in the deep hole. The zero streamline now rises off the bottom at kilometer 10 and forms a closed cell. The density driven circulation in the deep hole does not feel the strong mouth stratification, $\overline{\delta s}$ = 2 ppt, because the river flow is advecting the salt out of the estuary. The strong surface outflow has sealed off the deep hole. Figure 8b and figure 7, at 684 hours, both show that circulation within the deep hole vanishes during the flood event.

The second time variable simulation, the fall run, covers the time period Sept. 3 - Sept. 30, 1986 when, in comparison to spring conditions, river flow is small and Bay salinity is high. The simulation, as in the spring run, is driven by time series of tidal height, mouth salinity and river flow. In addition to these data, a time series of local wind speed was recorded at the Patuxent Naval Air Station near Solomons Is. Hydrographic surveys on Sept. 15, Sept. 22 and Sept. 30, 1986 provide a basis for comparison between measured and calculated salinity distributions. A comparison between the observed velocity profile at Station PL1 (from Boicourt and Sanders, 1987) and the numerical solution at PL1 is made.

The mixing parameters for the fall simulation were the same constant values used in the spring simulation. The initial salinity profile was taken from a digitized version of a Sept. 2, 1986 slack water salinity profile. The wind stress was computed using data from P.N.A.S. and a quadratic law relating wind stress to wind velocity via a friction coefficient, taken here to be 1.6×10^{-3}. Wind speeds measured on land, because of increased frictional resistance, are often less than those acting on the estuary. In light of this observation, two fall runs were made: 1) fall run 4, which uses observed wind speeds and 2) fall run 5, which uses twice the observed wind speed. Results of both simulations will be presented below.

Comparison of observed surface salinity from the first hydrographic survey on Sept. 15 (figure 9a) shows the solution from run 4 to be within the error bars of all but one station, whereas the surface salinity for run 5 fits observed data in the lower estuary but not in the upper estuary. Figures 9b and 9c show comparisons made between calculated and observed surface salinity on Sept. 22 and Sept. 30. Again, the solution from run 4 fits the data throughout the estuary. The calculated gradient, .41 ppt/km, in salinity across the front is close to the observed, .45 ppt/km ± .1 ppt.

Run 5 does not compare well with the observed salinity distribution. The frontal structure in all three comparisons is displaced upriver and the average frontal gradient is .3 ppt/km. Based on comparisons of measured and calculated fields, run 4, with observed wind speeds, gives a more accurate solution for salt distribution within the estuary.

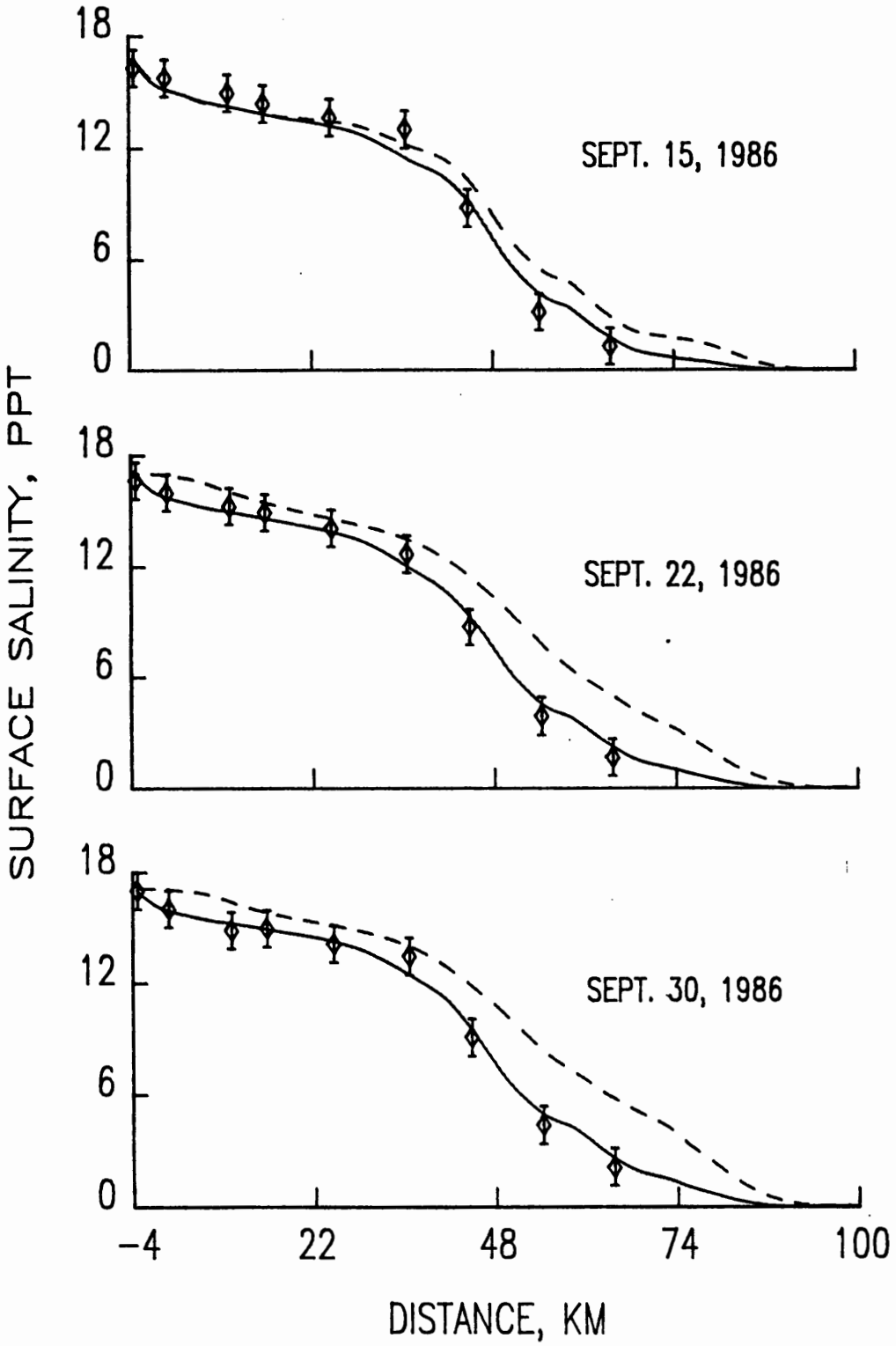

Figure 9: Plot of calculated, tidally averaged surface salinity, ppt, vs. observed data, plotted diamonds with ± 1 ppt error bars on (a) Sept. 15, 1986, 312 hours into the run, (b) Sept. 22, 1986, 470 hours into the run and (c) Sept. 30, 1986, 650 hours into the run. Fall runs have time variable mouth salinity, river flux and tidal heights. Fall run 4, solid line, uses observed time variable wind speed. Fall run 5, dashed line, uses 2x observed wind speed.

Figure 10 shows a comparison of measured and computed average velocity profiles at station PL1. Observed data was taken from Boicourt and Sanders, 1987 and the calculated profile from run 4. The velocity profiles are similar in both flow structure and magnitude. Both profiles are ± 5 cm/s 2-layer shear flows. Values for maximum, observed inflow and outflow velocities are approximately 1.2 times calculated values. The level of no motion for observed and calculated profiles is at 7 and 6 meters respectively.

Figures 11a and 11b are time series of wind speed versus PL1 and PL4 velocity profiles for both fall runs. A comparison of the profiles from the two runs illustrates the effects of increased wind stress on circulation and explains the intrusion of salt into the upper basin in fall run 5. Both runs show the same qualitative response to surface wind stress, but because wind speed is related to wind stress by a quadratic law, the quantitative differences between runs 4 and 5 are significant. During a two day period, September 10-11, an average -2.5m s^{-1} wind in fall 4 causes the surface velocity to go to zero. During the same period, in fall run 5, an average -5 cm^2 s^{-1} wind event causes surface velocity to reverse direction. The resulting -6.5 cm^2 s^{-1} surface flow, a 200% fluctuation in mean velocity, advects salty lower basin water over the sill into the upper basin, resulting in the displaced frontal structure recorded in figure 9.

In general, fall 4, using observed wind speeds, shows a consistent two layer density driven shear flow. Wind driven circulation produces fluctuations in mean flow of ± 100% in the surface layer and ~± 30% in the lower layer. There is one event, in the 13 and 17 meter records, 10 days into the simulation, in which the calculated fluctuation is 100% of the mean flow. This is probably due in part to a 1.4 ppt increase in δs which is coincident with a 4 m/s wind event. An identical 4 m/s wind event at 312 hours produces only a 25% perturbation in the average inflow at 13 meters depth, suggesting that as much as 75% of the fluctuation mentioned above, at 240 hours into the simulation, is attributable to changes in mouth stratification.

Fall run 5, with double the observed wind speed, shows fluctuations of ± 80% of the mean lower layer inflow and as high as ± 200% of the mean surface outflow. Fluctuations of this size compare well with data from Boicourt and Sanders (1987), though the timing of the responses does not exactly coincide. The high values for wind speed necessary to produce such large model fluctuations, however, as shown above, generate salinity fields which do not match observed data.

5. TOPOGRAPHICALLY CONTROLLED CIRCULATION

The influence of variable channel depth on circulation and salt distribution within the Patuxent River, primarily near the inner sill and within the 43 m deep hole, remains unresolved by observation alone. The code used in this study has been modified to handle variable bathymetry in an attempt to further the understanding of topographically controlled circulation. The most direct piece of evidence for topographic control is the persistence of a nearly vertical salt front at the inner sill in OEP surveys for all four seasons.

The dynamic effect of the sill is clearly seen in current meter records. Time series from stations P8 and P9, located to either side of the sill, show a 90° phase difference in maximum tidal velocities. The model successfully matches this phase difference in tidal velocities and predicts a low frequency sill-induced circulation. Figure 12 shows one month averaged streamfunction plots for fall and spring forcing regimes. The two fields, which are clearly similar, show two density driven circulation cells, one in the lower basin and one in the upper basin, divided in the middle by the inner sill.

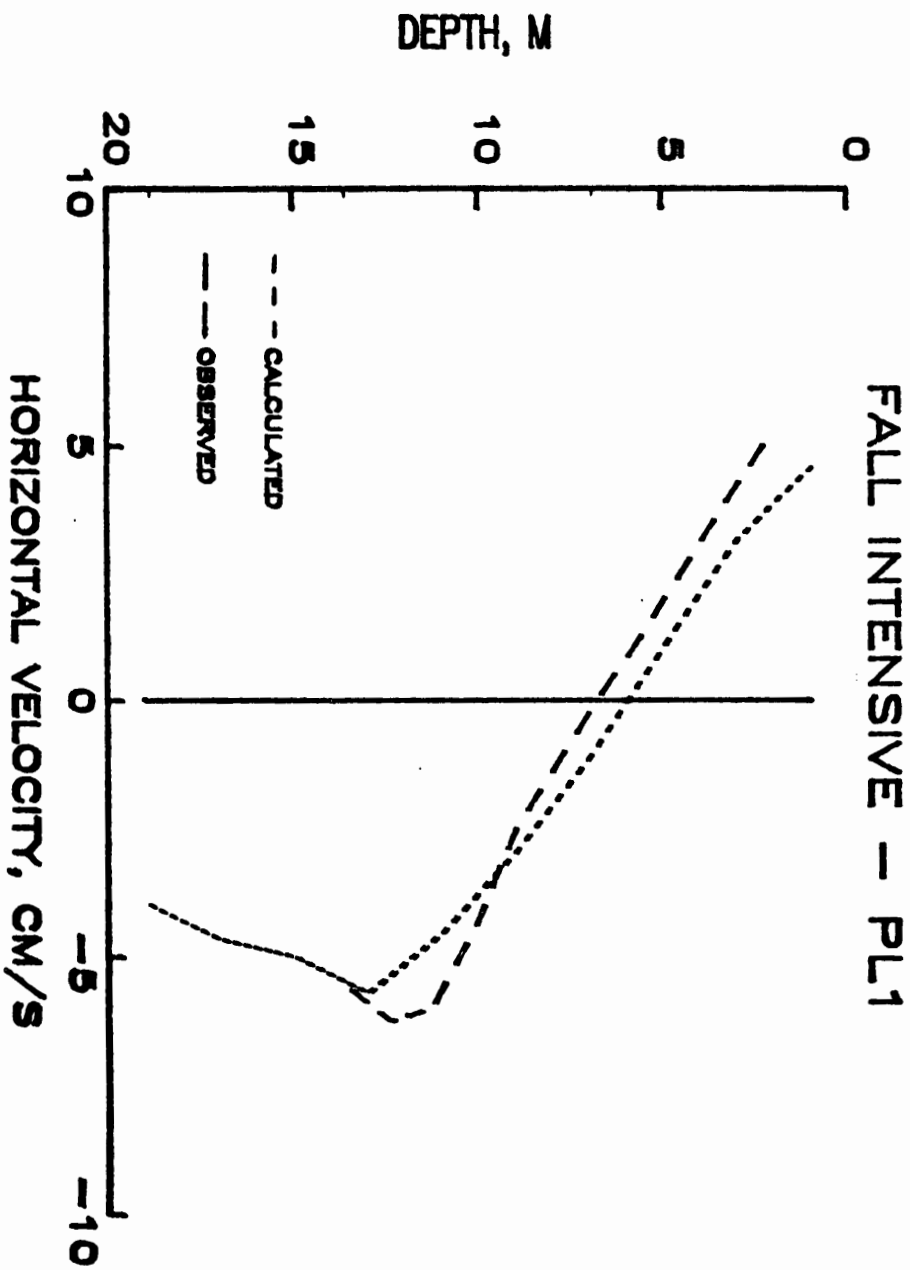

Figure 10: Survey averaged velocity profiles at station PL1. Fall run 4 plotted as a short dashed line. Observed data (from Boicourt and Sanders, 1987) plotted as a dashed line. Both profiles show a 2-layer shear flow.

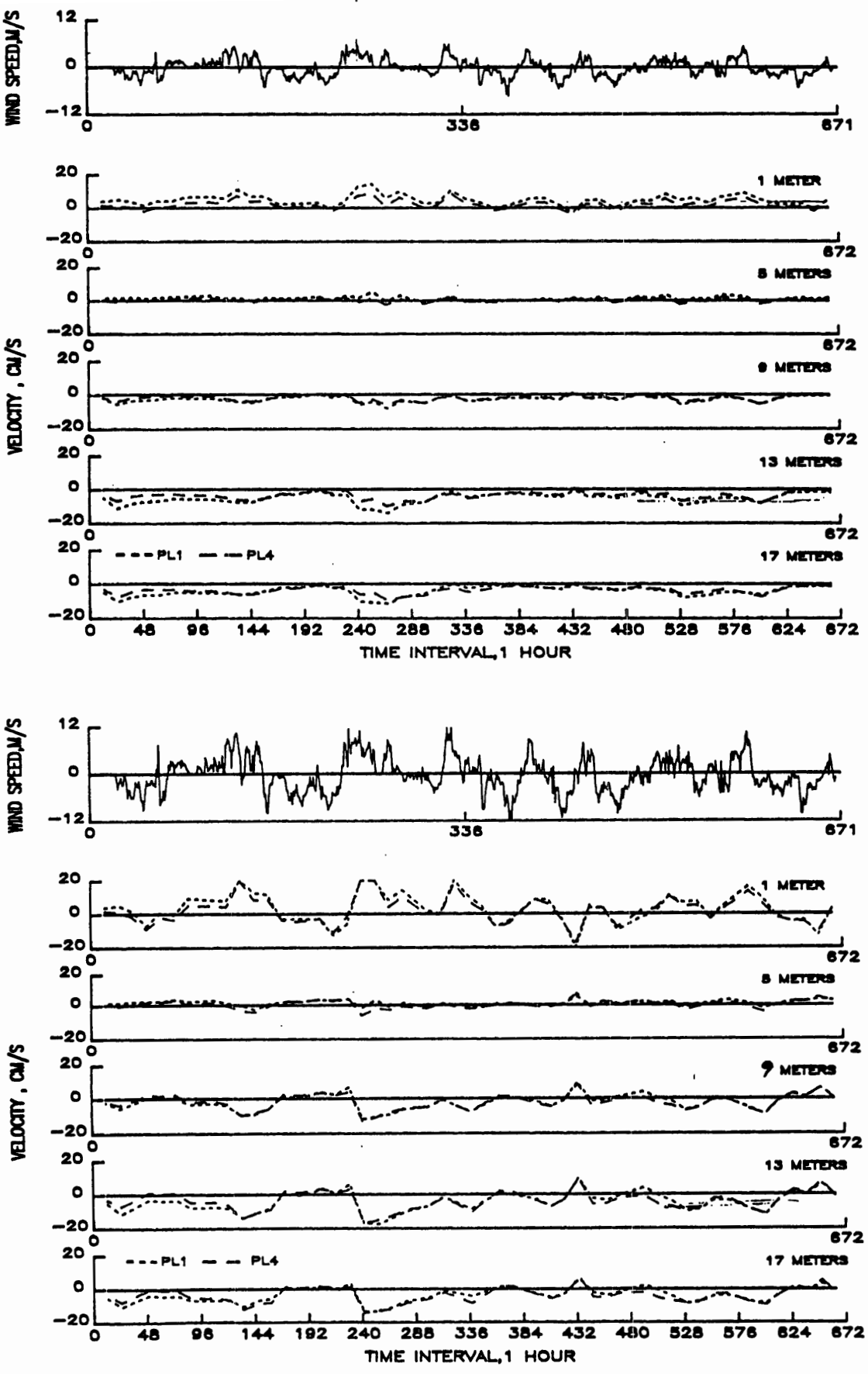

Figure 11: Time series of tidally averaged velocity profile at 1,5,9,13 and 17 meter depths at stations PL1 and PL4 vs. (a) a time series of observed wind speed, fall 4 and (b) a time series of twice the observed wind speed, fall 5.

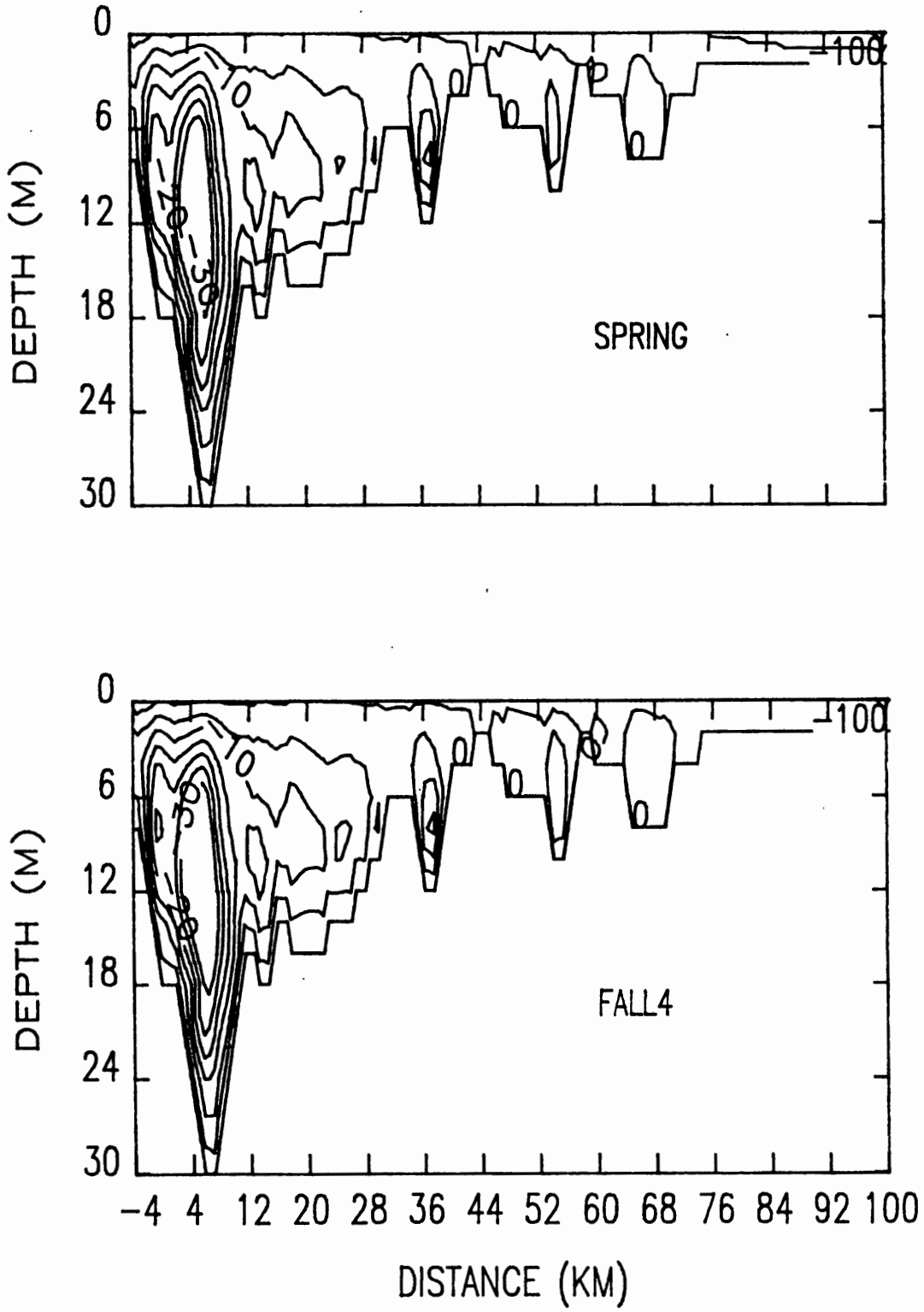

Figure 12: Streamfunction contours calculated from (a) spring survey averaged and (b) fall 4 survey averaged velocity fields, $cm^2/s/1000$. Contour interval is 5.

Figure 13 shows the large distortions in instantaneous surface elevation which occur in the vicinity of the sill, where both width and depth decrease. The surface slope changes sign to either side of the sill. The effect of this sign reversal on velocity is illustrated in the profiles at kilometer 32 and 54 located on either side of the sill. The phase difference in tidal velocity records at P8 and P9 indicates the phase of the tide changes near km 44, because of the constriction.

Figure 14 shows contours of salinity variance (σ^2) calculated for both transient runs from the expression

$$\sigma^2(x,y) = \frac{1}{N-1}\sum_{1}^{N}\left(s_i(x,y) - \bar{s}(x,y)\right)^2 \qquad (18)$$

where N is the total number of entries at point x,y in the time domain. The time average salinity at a point is expressed by

$$\bar{s}(x,y) = \frac{1}{N}\sum_{2}^{N} s_i(x,y)$$

The largest variance for the spring survey occurs between the inner sill and Lower Marlboro, in association with a local maximum in tidal excursion. Of more interest is the absence of a minimum variance within the deep hole near Pt. Patience. The values of $\sigma^2 = .3$ and $\sigma^2 = .5$ for spring and fall runs suggest that, on average, the deep hole is not stagnant. During both steady state and transient simulations the hole remains well flushed with average velocities of 4 cm s^{-1}. The hole may, however, be subject to transient short term periods of stagnation.

Salinity surveys made in the lower basin have documented that well developed halocline structures are sometimes present over the deep hole section, particularly during summer (low flow) conditions. An example, from summer 1984, is shown in figure 15a. The processes leading to halocline formation are not fully understood, nor do we know how frequently such events occur and how long they last. These may well be very significant events, however, because they may produce stagnant conditions in the deep hole region and in smaller holes throughout the lower estuary. From figure 15a it appears the halocline in the Patuxent forms in response to increased stratification in adjacent parts of Chesapeake Bay. This stratification could result from either a decrease in surface salinity, an increase in bottom water salinity or both. The pattern of isohalines near the Patuxent mouth in figure 15a indicates at least the first alternative occurred.

In order to determine whether this scenario can account for halocline formation, and also to determine the consequences for circulation in the deep hole region, we conducted a numerical experiment of the model's response to increased stratification at the mouth. The initial conditions corresponded to the final state of the spring intensive simulation. On that state we have imposed a pattern of decreasing surface salinity at the mouth, producing an increased mouth stratification.

The deep hole response to several days of increased stratification is shown in figures 15b. The salinity pattern is very similar to the data in figure 15a. Low salinity surface water from the Bay has intruded about 5 km into the Patuxent, and between 4 and 10 m depth the isohalines throughout the lower estuary are nearly horizontal. The topographically controlled front separating upper and lower basins has an enhanced gradient. Velocities throughout the water column in the Pt. Patience deep are uniformly small, less than .2 cm^2 s^{-1}. The velocity profile at Lower Marlboro remains a normal two layer flow while circulation within the deep hole has nearly vanished. It is clear that low salinity, surface water intrusions from the Bay can produce transient salinity patterns like that seen in figure 15a.

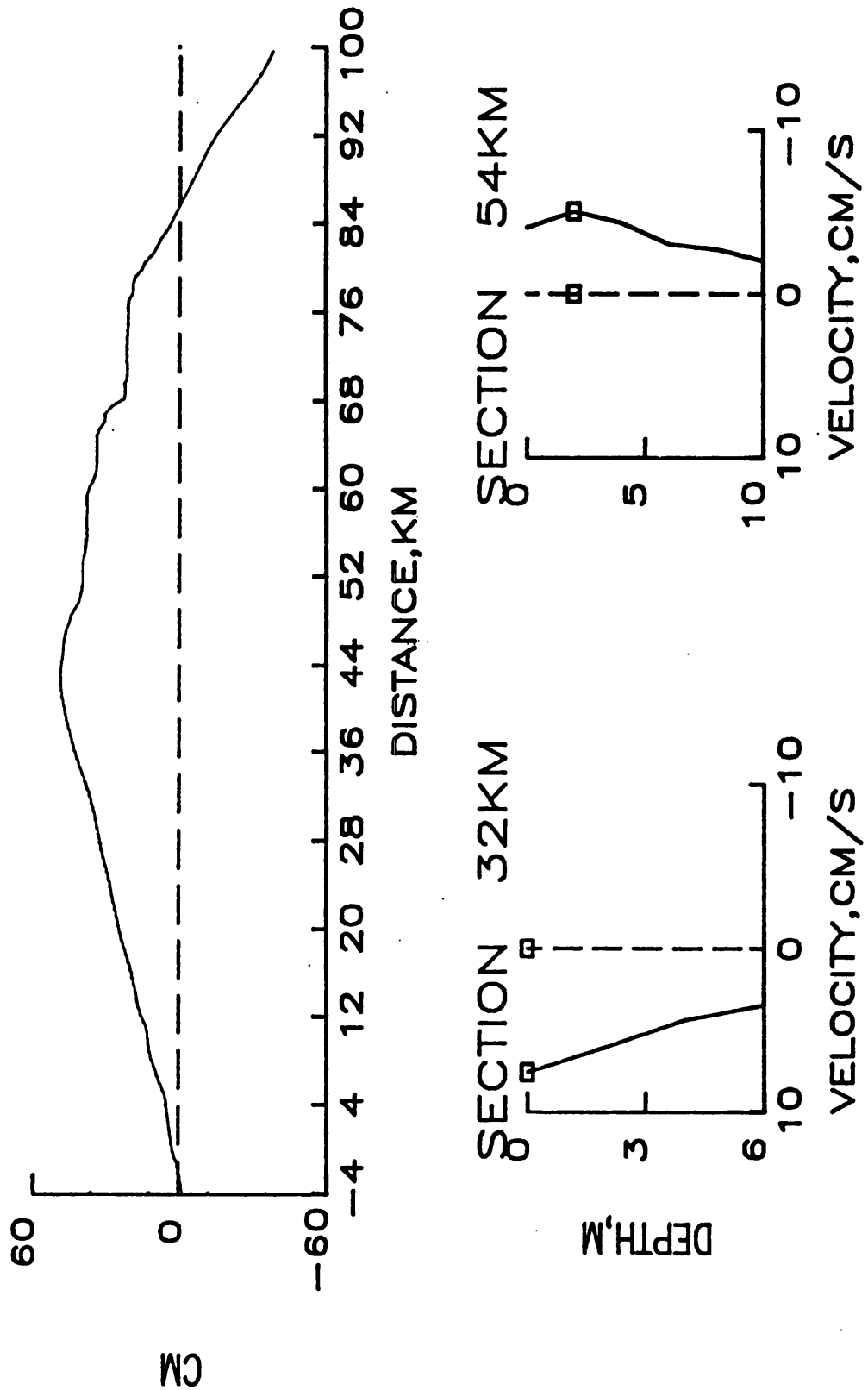

Figure 13: Plot of instantaneous surface elevation and velocity profiles at 159 hours, during the May 21 tidal oscillation, for the spring intensive run. Plot shows the abrupt sign reversal in surface slope which occurs across the inner sill at 44 kilometers and the phase difference in tidal velocity between profiles at 32 and 54 km.

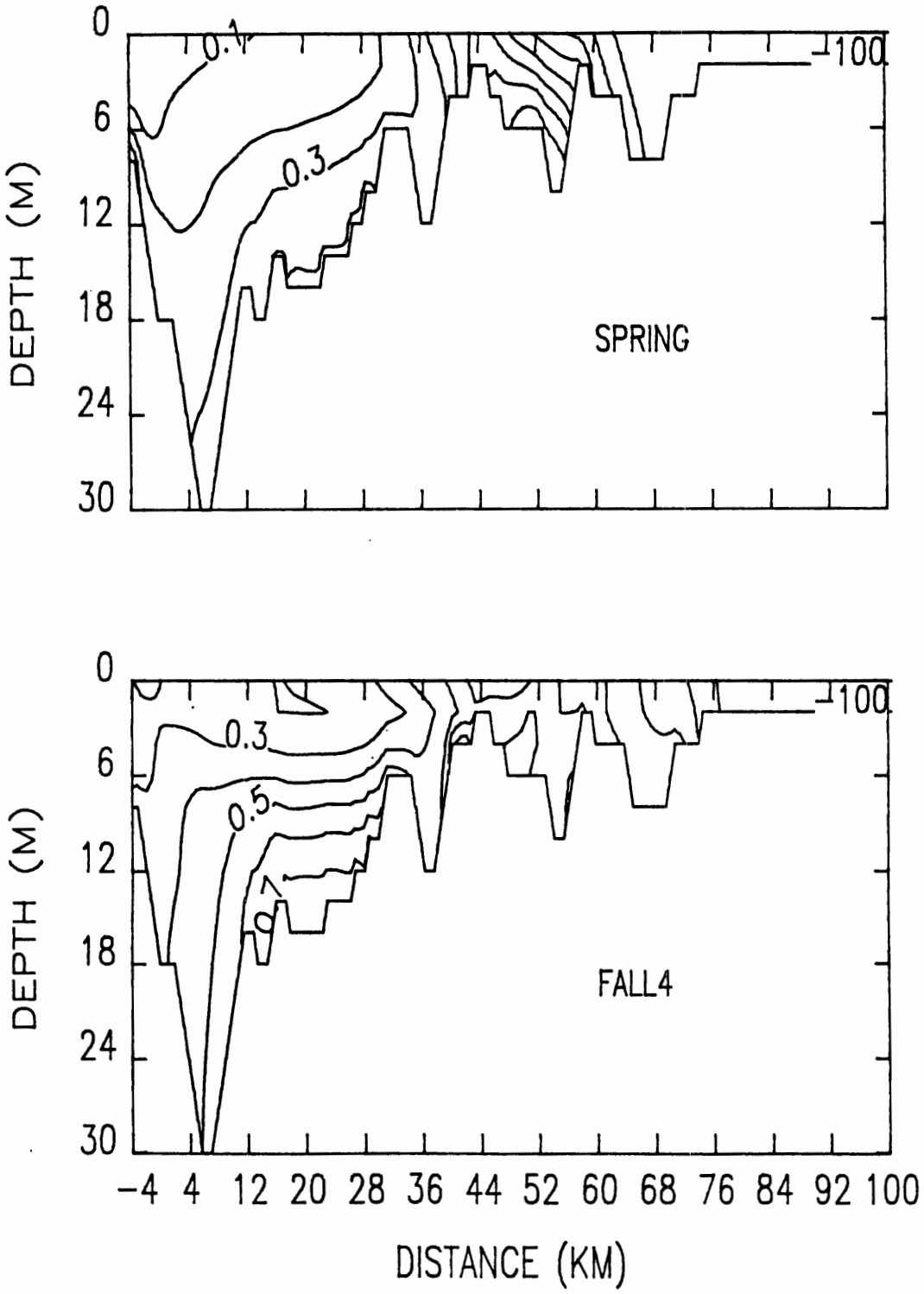

Figure 14: Contoured salinity variance calculated over (a) entire spring intensive and (b) entire fall run 4. Contour interval .1.

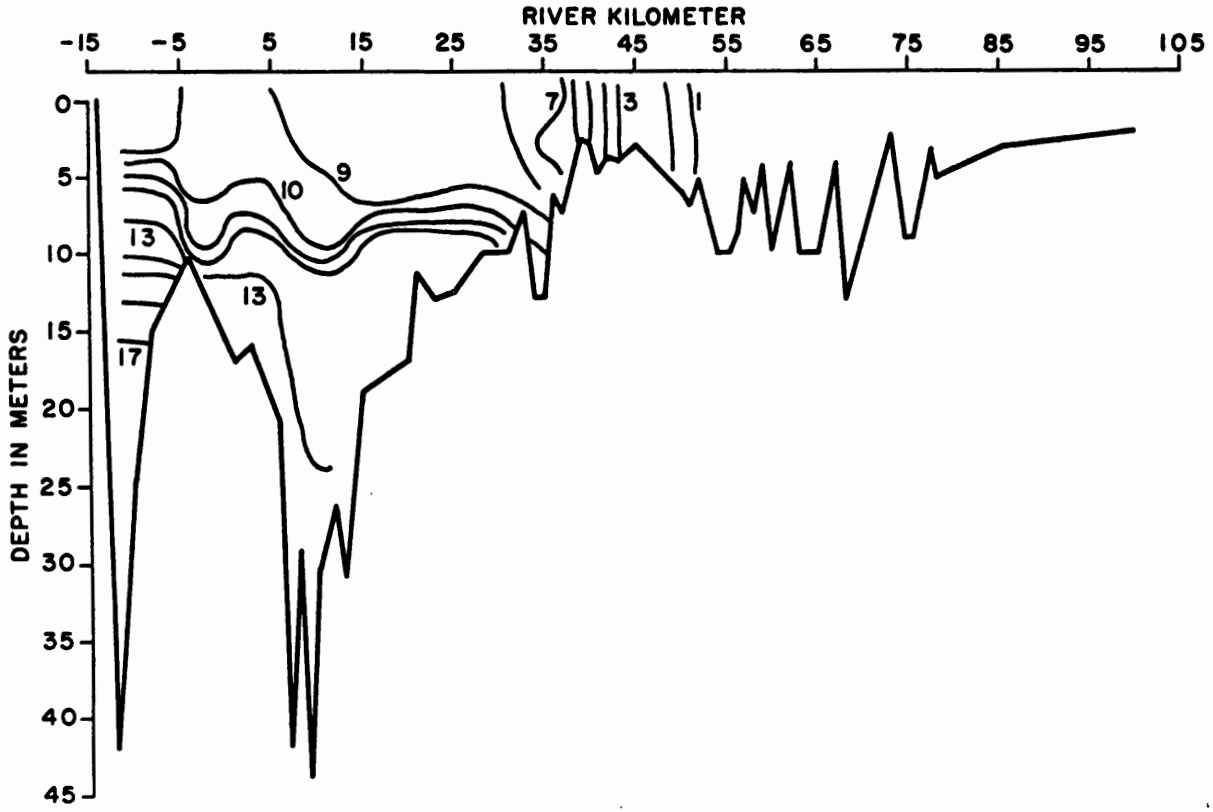

Figure 15a: Patuxent River estuary salinity cross section, from a single slack water cruise, summer 1984, from OEP Tech Rep. #7.

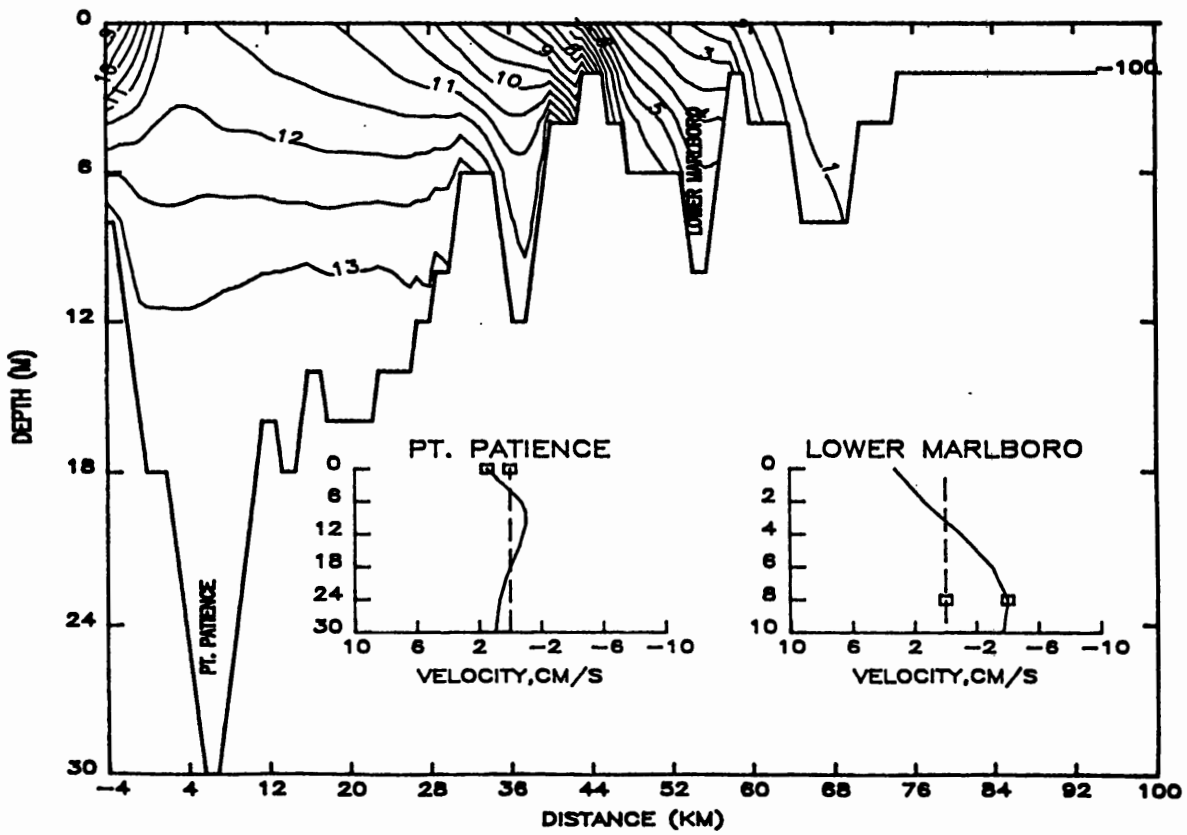

Figure 15b: Salinity pattern and velocity profiles over seventh tidal cycle of summer intrusion experiment. Contours are ppt.

CONCLUSIONS

Steady state simulations indicate that the seasonal average flow in the lower basin and much of the upper basin of the Patuxent Estuary is a two layer density driven flow, with velocities of ± 3 cm/s in the lower basin. A stationary, nearly vertical salinity front is present in all model calculations, centered on the inner sill separating the two basins. The front forms regardless of initial conditions. Its location is dictated by topography and its width is governed by the balance between river discharge, the strength of lower basin density driven circulation, and the magnitude of the horizontal turbulent diffusivity. Transition to unidirectional (one layer) flow occurs well upstream from the front, in the neighborhood of river kilometer 55. The best fit to the observed seasonal average salinity distribution, based on a comparison with data from spring 1984 (high discharge conditions), is found for eddy mixing and viscosity coefficient values K_z = 0.2 and N_z = 5.0 $cm^2 s^{-1}$.

Comparison between field data taken during the spring 1986 intensive survey and the model calculation for the same period shows that the model accounts for all of the major features seen in the data. Calculated surface salinities are generally within the error estimates of observed values throughout the whole estuary. The optimal value of the horizontal mixing coefficient K_x is near 1.25×10^5 cm^2/s. This value produces a salinity front on the inner sill above Benedict with a 0.35 ppt/km gradient, compared with a mean value of 0.4±.1 ppt/km estimated from the data. Comparison between measured and computed tidal velocities shows generally good correlation throughout the estuary, with an average observed/calculated ratio of 1.6. This indicates the lateral distribution of tidal velocities is nearly parabolic. The range of calculated tidal velocities is 20-50 cm/s. The estuary is dominated by a 2-layer ±5 cm/s shear flow. On average, the transition from one to two layer flow occurs near river kilometer 60 but is very sensitive to freshwater runoff. Three layer flows are not seen. Low frequency variability, ±50% of fluctuation of mean surface velocities and ± 80% mean lower layer velocities, during this period is due primarily to variations in boundary salinity.

Our comparison of the observations made during the fall intensive survey reinforces the conclusions drawn from the spring survey comparison, and in addition, highlights the wind driven component of the circulation. Calculations using values of wind speed measured remotely match the observed surface salinities whereas the calculation using twice the measured winds does not.

As was found for the spring period, the two layer density driven flow predominates in long term (>10 day) average. Average two layer flow velocities at the mouth are calculated to be +5 and -6.25 cm/s, with the level of no motion at 6-7 m depth. These agree with average values at station PL1, corrected to give zero net flux. Three layer flows occur only as brief transients, the result of combined wind and density forcing. Wind driven fluctuations of ±5-6 cm s^{-1} are seen within the surface layer. Fluctuations in mean lower layer velocity are due to both wind events, ±2 cm s^{-1}, and variations in mouth salinity, as much as ±4 cm/s. The magnitude of the wind driven component in the calculations agrees with the observed magnitude. However, the phase of wind driven fluctuations near the estuary mouth does not agree well with the data from station PL1, as presented by Boicourt and Sanford (1987).

The deep hole and basins at 36 and 54 kilometers contain recirculatory circulation cells which persist through both spring and fall simulations. The basin at 68 kilometers maintains two layer circulation during fall or low flow conditions but is flushed during spring or high river flow conditions. There is no minimum in salinity variance within the hole as would be expected were it stagnant. The deep hole is, on average, well flushed by a vigorous two layer circulation to at least 30 m depth. The hole is subject to brief periods of stagnation in association with halocline development in the lower estuary and or high runoff events. Simulations of summertime conditions indicate that the

deep hole at Pt. Patience can stratify and become stagnant in response to sharp increases in Chesapeake Bay stratification. During these episodes, a sub horizontal halocline develops over the deep hole, and circulation velocities below the halocline nearly vanish.

References

Boicourt, W.C.; Sanford, L.P. A hydrodynamic study of the Patuxent River estuary. Part A: Observations of current, temperature, salinity, tidal height and wind. Part B: Hydrography of the Patuxent River estuary. October 1985 to October 1986 (preprint).

Blumberg, F.A. The influence of density variations on estuarine tides and circulations. Estuarine Coastal Mar. Sci., 6: 209-215; 1978.

Elliott, A.J. Observations of the meteorologically induced circulation in the Potomac estuary. Estuarine Coastal Mar. Sci. 6: 285-299; 1978.

Festa, J.F.; Hansen, D.V. A two-dimensional numerical model of estuarine circulation: the effects of altering depth and river discharge. Estuarine Coastal Mar. Sci. 4: 309-323; 1976.

Festa, J.F.; Hansen, D.V. Turbidity maxima in partially mixed estuaries: a two-dimensional numerical model. Estuarine Coastal Mar. Sci. 7: 347-359; 1978.

Garvine, R.W. Dynamics of small-scale oceanic fronts. J. Phys. Ocean. 4: 557-569; 1974.

Garvine, R.W. An integral hydrodynamic model of upper ocean frontal dynamics: Part I. Development and analysis. J. Phys. Ocean. 9: 1-18; 1979.

Gordon, R.B.; Spaulding, M.L. Numerical simulations of the tidal- and wind-driven circulation in Narragansett Bay. Estuarine Coastal Shelf Sci. 24: 611-636; 1987.

Hibiya, T. Generation mechanism of internal waves by tidal flow over a sill. J. Geophys. Res 91: 7697-7708; 1986.

Huzzey, L.M. The dynamics of a bathymetrically arrested estuarine front. Estuarine, Coastal and Shelf Science 15: 537-552; 1982.

Oey, L.-Y. On steady salinity distribution and circulation in partially mixed and well mixed estuaries. J. Phys. Ocean. 14: 629-645; 1984.

Oey, L.-Y.; Melor, G.L.; Hires, R.J. A three-dimensional simulation of the Hudson-Raritan Estuary. Part I: Description of the Model and Model Simulations. J. Phys. Ocean. 15: 1676; 1985.

Olson, P. The spectrum of subtidal variability in Chesapeake Bay circulation. Estuarine, Coastal Shelf Sci. 23: 527-550; 1986.

Patuxent Estuary Water Quality Survey: 1984 Data Summary. MD Office of Environmental Programs Tech Rep #7, 1985.

Phillips, O.M. The Dynamics of the Upper Ocean. Cambridge: Cambridge University Press; 1980.

Prandle, D. Salinity intrusion in estuaries. J. Phys. Ocean. 11: 1311-1324; 1981.

Wang, D.P. Wind-driven circulation in the Chesapeake Bay, winter 1975. J. Phys. Ocean. 9: 564-572; 1979.

Wang, D.P.; Kravitz, D.W. A semi-implicit two-dimensional model of estuarine circulation. J. Phys. Ocean. 10: 441-454; 1980.

Wong, K-C; Garvine, R.W. Observations of wind-induced subtidal variability in the Delaware estuary. J. Geophys. Res. 89: 10589-10597; 1984.

Understanding the Estuary: Advances in Chesapeake Bay Research. Proceedings of a Conference. 29-31 March 1988. Baltimore, Maryland. Chesapeake Research Consortium Publication 129. CBP/TRS 24/88.

Lagrangian Drift Model of Suspended Sediment Transport in Chesapeake Bay

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INTRODUCTION

Suspended sediment transport in Chesapeake Bay is simulated with a numerical model of circulation and Lagrangian drift, and modeled distributions are compared to in situ and remotely sensed data. Suspended sediment, which is important in itself for transporting chemicals and for limiting photosynthesis by blocking light, also acts as a tracer for circulation patterns and can be used in understanding transport mechanisms. More importantly, suspended sediment is a variable which can potentially be used to calibrate and verify circulation models, and unlike water currents, it can be remotely sensed, making data acquisition rapid and inexpensive. Model results also offer the possibility of interpolating conditions in space and time when only limited observational data is available.

The circulation model used in this study has been used to hindcast tidal, wind-driven, and density currents in the Bay and the local continental shelf (Hess 1985, 1986). The Lagrangian drift model (drifters move at the speed of the surrounding water) was originally developed to simulate the motion of free-moving larval stages of marine organisms (Johnson 1987) and was applied to blue crab larval drift (Johnson et al. 1986, 1987; Hess and Johnson 1988). That drift model has been modified to include settling under the force of gravity and the multiple injection of drifters to

simulate a continuous source. A settling velocity which represents suspended sediment particles in Chesapeake Bay (Schubel 1972) was chosen and applied uniformly to all drifters. When settling takes the particle to the bottom, the particle is removed from further advection. Identification of areas of deposition can also be made.

A convenient test case for the simulation is the high turbidity occurrence in the upper Bay during the high-precipitation event in March 1979. Remotely-sensed (Stumpf 1988) and in situ (Cronin 1982) observations of water turbidity in the Bay are compared to model simulation results.

THE NUMERICAL CIRCULATION MODEL

Hess (1985, 1986) has developed a general three-dimensional, free-surface numerical circulation model, called MECCA (Model for Estuarine and Coastal Circulation Assessment). MECCA uses finite difference approximations to the momentum, continuity, and temperature and salinity equations to simulate time-varying water currents, salinities, and temperatures in a shallow water domain at time scales from a few minutes to several months, and space scales from a few kilometers to a few hundred kilometers. The model is designed to simulate circulation driven by tides, winds, water density gradients, and atmospheric pressure gradients. A fully three-dimensional, time-variable approach was decided upon because of the importance of knowing surface currents (as opposed to layer-averaged currents), and because of the rapid changes in currents expected to occur when tides and winds are important.

The major feature of the circulation model is the use of split-mode velocity equations. The external, or barotropic, velocity mode (vertically-averaged velocity) is subtracted from the total velocity to get the internal, or baroclinic, velocity mode. Because the internal mode stability requirement in the numerical solution scheme is less stringent than that for the external mode, the internal mode can be updated less often, resulting in significant savings in computer time. For these simulations, the external-mode time step used was 6 minutes, and the internal-mode time step was 30 minutes. The model uses a dimensionless vertical coordinate, also known as a sigma coordinate. This terrain-following coordinate improves the representation of the bathymetry.

All modeled mass and momentum diffusivities are functions of local velocity gradients, and therefore may vary over time and space. In addition, vertical diffusivities are reduced in the presence of vertical density gradients as a function of the local Richardson number.

The differential equations are approximated by finite differences on a grid mesh of 434 square and triangular elements 11.2 kilometers on a side (Fig. 1). Placement of the variables in the horizontal plane is staggered to prevent spurious numerical solutions. Vertical staggering at the 10 model levels allows for better resolution of vertical gradients. Semi-implicit grid-row manipulations are employed in both the vertical and horizontal to augment computational stability. The grid mesh allows for the resolution of narrow inlets in barriers and for variable-width channels. These channels allow for rivers, which generally have widths much narrower than a typical grid cell, to be explicitly included in the numerical grid scheme.

Tidal forcing at the deep-water boundaries was represented by the sum of constituent tides with amplitudes and phases adjusted to reproduce the tides at the Hampton Roads gage and the currents at the entrance to Chesapeake Bay. Twenty constituents are included and the offshore amplitude is updated each 6 minutes.

Daily river flows at five major rivers (Susquehanna, Potomac, Rappahannock, York, and James) are supplied at grid cells representing the approximate location of the fall lines. These values are assumed to represent midday conditions; quadratic interpolation is used to estimate flows at other times.

Numerous model calibration and verification studies have been carried out over the past few years (Hess 1986). Validation studies have shown that tides and tidal currents in the mouth are simulated to the acceptable level of accuracy of 10 to 20 cm/s, or 15 percent of full scale (Fig. 2), and that tidal currents throughout the Bay as a whole are of similar accuracy (Fig. 3). Further tests (Hess 1985) indicate that wind-driven and density-driven currents are modeled to a similar level of accuracy. In sum, MECCA produces reasonable approximations to the actual currents when historical input data are used. More importantly, it is a useful tool for the comparison of scenarios covering long periods, as in high runoff events modeled here.

Several simplifications of the model were made for these simulations. No winds were applied, a linearized bottom friction was assumed, and water density was assumed to vary only with salinity. A single-component tide with an amplitude of 0.4 meters and a period of 12 hours was used to drive the currents. Constant horizontal and vertical viscosities were used.

THE LAGRANGIAN DRIFT MODEL

Johnson (1987) has developed a Lagrangian drift model called LARTREK. LARTREK simulates the movements and distribution

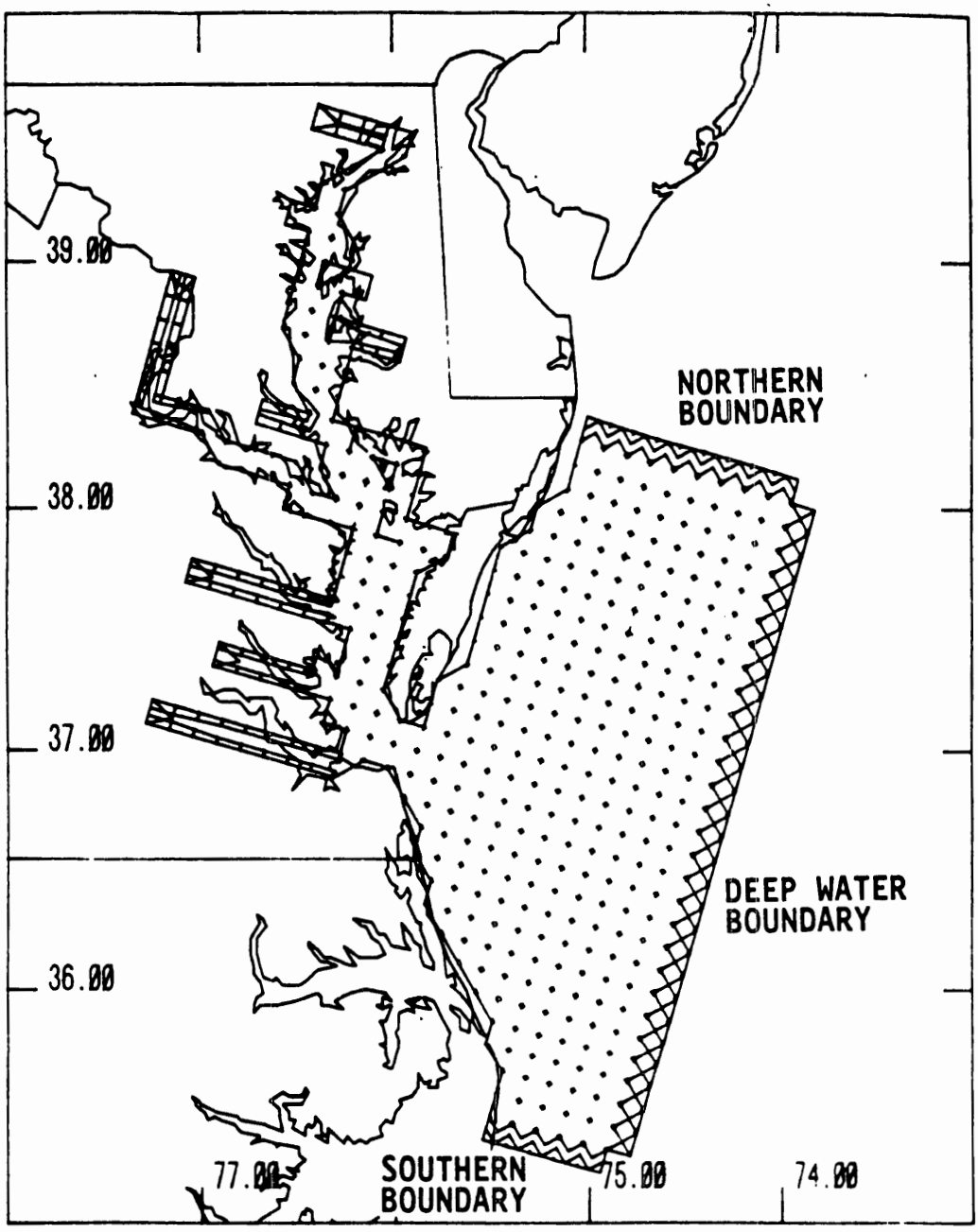


Figure 1. The circulation model grid mesh for Chesapeake Bay and the adjacent coastal waters showing the location of oceanic boundaries, and river inputs. Grid cells measure 11.2 x 11.2 km. Cells marked with an "X" along the deep-water boundary require input water level data. Cells along the northern and southern boundaries marked with a "<<" use a radiation outflow boundary condition. Cells marked with a ">" at the heads of rivers require flowrate boundary conditions.

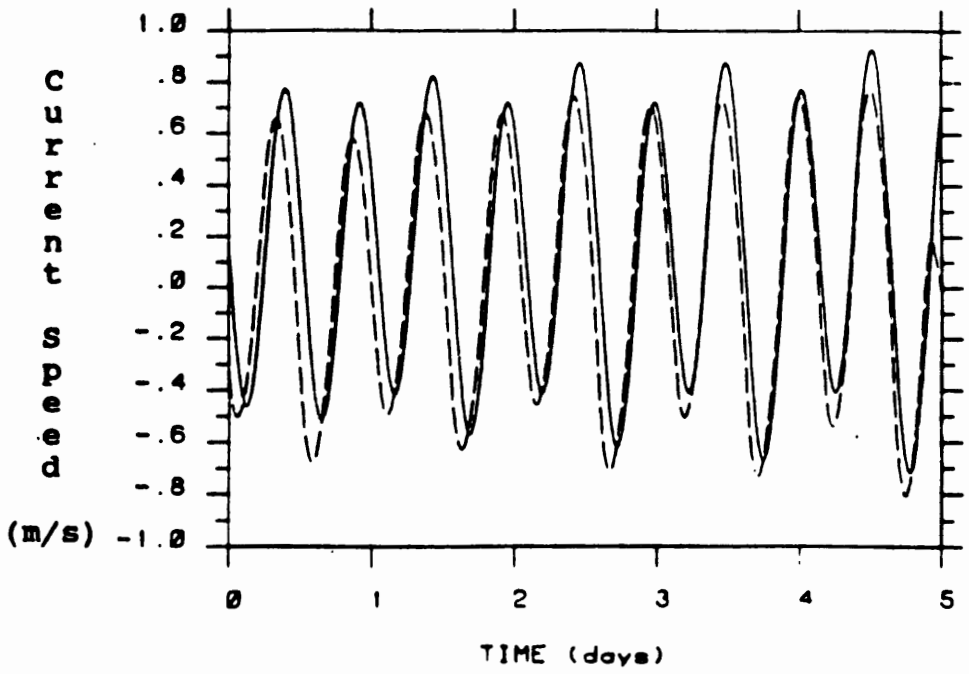


Figure 2. Modeled and NOS-predicted tides and currents near the mouth of Chesapeake Bay. Flood currents are positive.

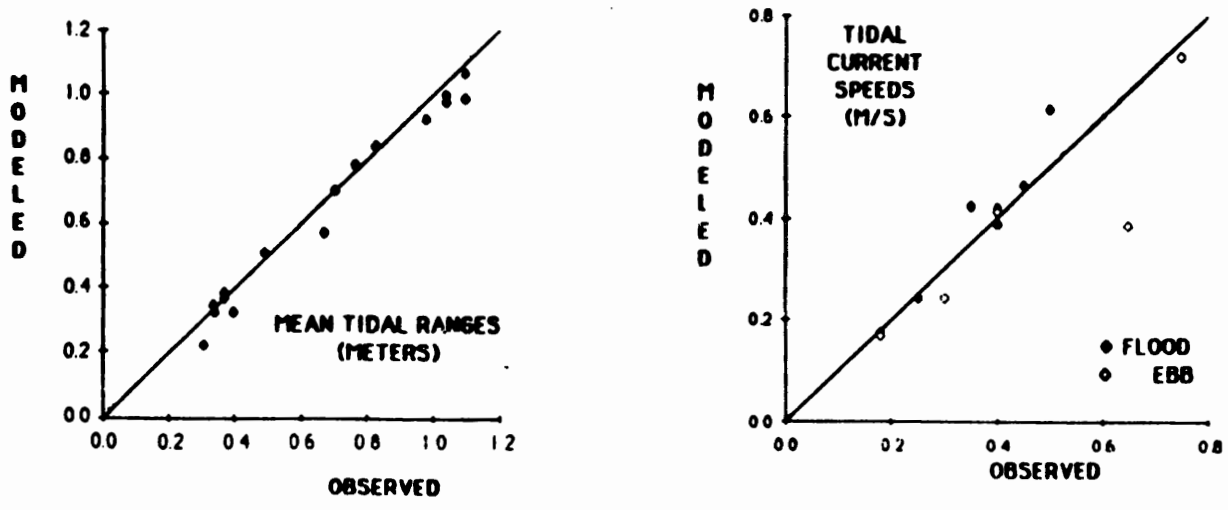


Figure 3. Modeled and NOS-observed (a) mean tide ranges and (b) current amplitudes at several locations in Chesapeake Bay.

of Lagrangian drifters using the three dimensional velocities calculated by MECCA, and algorithms to simulate beaching and deposition. LARTREK is run concurrently with MECCA.

The simulation of suspended sediment transport with discrete Lagrangian drifters offers several advantages over the solution of the advective-diffusive equation. Large concentration gradients at a sub-grid scale can be produced with Lagrangian drifters. Identification of individual drifters allows for explicit representation of the time the sediment has been water-borne or deposited, and for determination of the point of origin. An inhomogeneous sediment can be represented by multiple settling velocities. The Lagrangian approach also offers advantages when applied to biological organisms which have swimming or migratory behaviors.

Horizontal advection

After each external-mode time step of MECCA (6 minutes of simulated time), LARTREK calculates new positions for each drifter by three-dimensionally interpolating the u, v and w components of velocity from MECCA. Because the horizontal velocity field, v(x), may vary over the distance a drifter moves within a time step, the drifter's position is advanced in the horizontal plane by a predictor-corrector method. For a drifter initially at point x_o, the first estimate of the spatially-averaged velocity, v', is $v(x_o)$. Then an estimate of the updated position, x_p, is computed by

$$x_p = x_o + v'\Delta t \qquad (1)$$

where Δt is the circulation model's external mode timestep. An improved estimate of the spatially-averaged velocity over the path is then computed by

$$v' = [v(x_o) + v(x_p)]/2 \qquad (2)$$

Another estimated end position is then recomputed from (1); four iterations of this procedure have been shown to be sufficient (Johnson et al. 1986).

A small random drift velocity is computed using the relationship

$$v'' = [2A_h/\Delta t]^{1/2} \qquad (3)$$

where A_h is the local horizontal turbulent eddy viscosity (Csanady 1973). The velocity v" is added to the spatially-averaged velocity at each timestep in a randomly chosen direction.

In this numerical drift model, the interaction of a water-borne drifter with the shoreline is carefully controlled. In the present version of the model, drifters are reset to a

distance of 100 meters from land if they encounter a land boundary. Drift parallel to the boundary is not restricted. In some areas of the MECCA grid system, the land-water boundary occurs adjacent to a triangular grid. Because of the differences in geometry between square and triangular grids, separate but analogous land interaction schemes and velocity interpolation schemes are invoked for the triangular grid cells.

Application to Suspended Sediment

A settling velocity is added to the drifter's vertical velocity to simulate sediment settlement and deposition. A velocity of 10^{-5} m/s (approximately 1 meter per day) was used in all simulations (Schubel 1972). When a drifter touches bottom it is deposited and removed from further motion; there is no resuspension. Diffusion in the vertical direction is approximated by introducing a random motion.

Suspended sediment entering the Bay through a river is simulated by the simultaneous release of five Lagrangian drifters in the MECCA cell which represents the river's uppermost reach. The drifters have the same horizontal coordinates and are spaced uniformly over the vertical, starting from the surface. Injection occurs at specified time intervals following a hydrodynamic spinup period.

To insure a constant concentration, an injection rate which depends on the river flow is used. Given a reference flow (Q_r) and time interval (t_r), the injection time interval (t_i) for any other flow (Q) is

$$t_i = (Q_r/Q)t_r \qquad (4)$$

Here Q_r=2000 m^3/s and t_r=6 hours. Although there is no convenient way to accurately estimate concentration from the distribution of drifters, one could simply divide the number of drifters in each MECCA cell by the cell's volume to get a relative concentration.

RESULTS OF SIMULATIONS

General sensitivity tests

Several scenarios were run to test the relative importance of different factors. Early tests showed that the non-linear advective terms have little effect in determining sediment distribution.

A series of runs was made to determine the time required to come to at state of equilibrium (i.e. when the rate of injection equals the rate of removal by deposition) for suspended sediment entering the Bay through the Susquehanna River (Fig. 4a). For a Susquehanna flow of 4000 m^3/s, the time was 7.5 days (Fig. 4b); for a flow of 8000 m^3/s, the time was 11 days (Fig. 4c); for a flow of 12,000, the time was 12 days (Fig. 4d). The difference in the required times

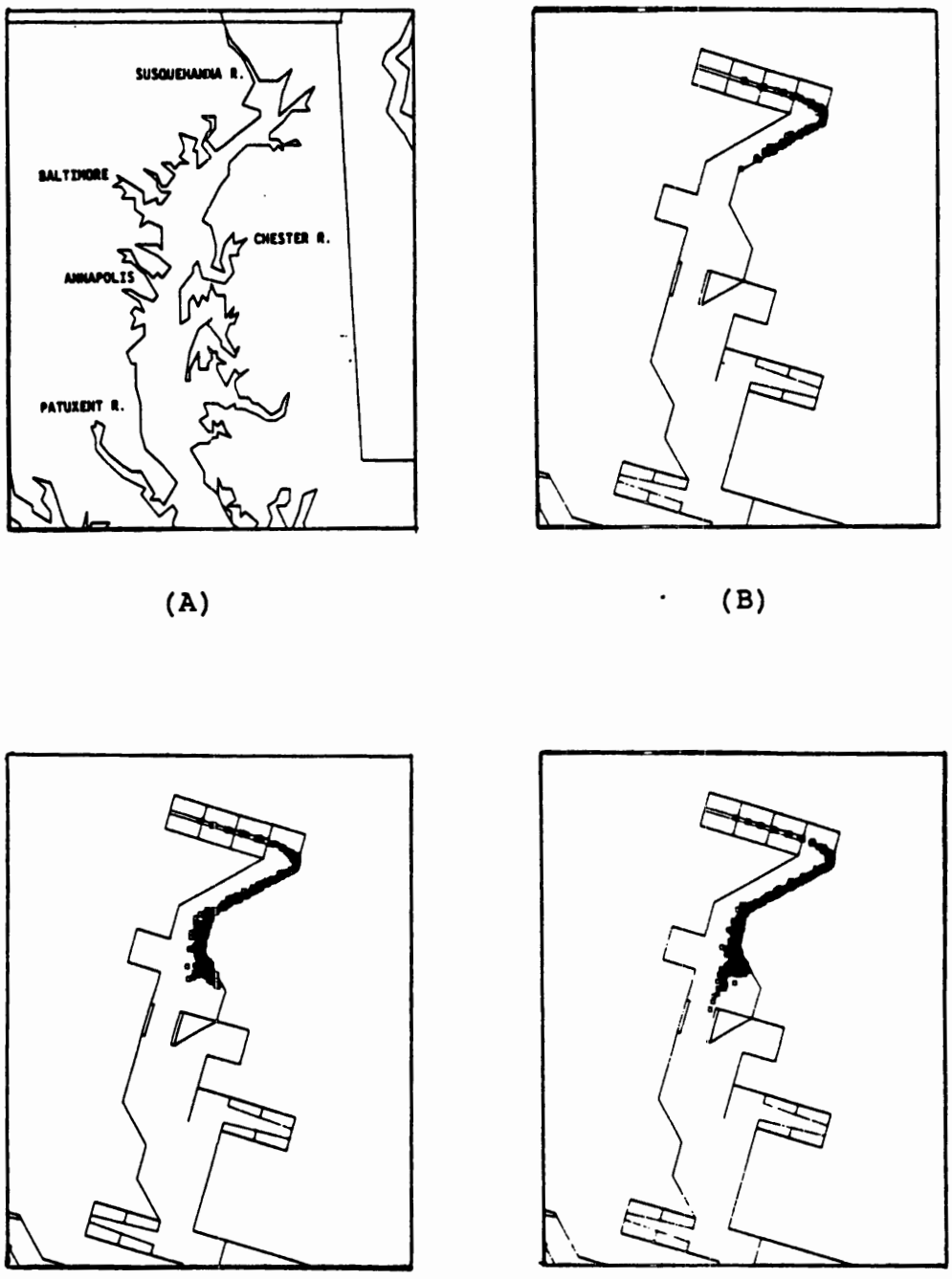


Figure 4. Simulated equilibrium positions of Lagrangian drifters in the upper Chesapeake Bay for various Susquehanna River (A) flows. Modeled position of drifters (shown as a small square drawn at the drifter location) in the uppermost 4 meters at equilibrium for flows of (B) 4,000 m^3/s, (C) 8,000 m^3/s and (D) 12,000 m^3/s.

is due to the advection of drifters into deeper water when the flow is increased.

Tests were made to determine the effect of the vertical random velocity. Computing a value from (3), but with the vertical diffusivity (A_v=0.003 m^2/s) rather than the horizontal diffusivity (A_h), gave the diffusive velocity $w'=4 \times 10^{-3}$ m/s. This speed caused rapid deposition of drifters and was not suitable for representing diffusion. An alternative diffusive velocity, 10^{-4} m^2/s, was used instead.

March 1979 flood simulations

The high flow event in March 1979 is an ideal test case for the simulation model because shipboard and satellite data on suspended sediment distributions are available. Daily mean river flowrate data for the Susquehanna River (Fig. 5) were taken from U.S. Geological Survey records (USGS 1979) for the period of the simulation (1 February - 31 March, 1979). Annual mean river flowrates were used for the Potomac, Rappahannock, York, and James Rivers.

Observed suspended sediment concentrations in the Susquehanna River (Lang 1982) varied with flow, increasing to a peak on 7 March 1979, and diminishing thereafter; there is no data before 5 March (Fig. 5). The modeled concentration was assumed to be zero until 5 March, then to vary directly with the flow until 13 March, when it again becomes zero. The injection rate which approximately fits this assumption is

$$t_i = (Q_r/Q)^2 t_r \qquad (5)$$

Observations of surface concentrations are available for four March dates. Landsat images showing Bay-wide surface water turbidity distribution with a resolution of less than a kilometer are available for 9 and 17 March (Stumpf 1988). Ship cruise data is available for 12-13 and 26-27 March (Cronin et al. 1982).

The observed and simulated distribution for 9 March 1979 is shown in Fig. 6. The remotely-sensed area of maximum distribution extends from the Susquehanna River entrance southward to latitude 39.00 N. The modeled region of maximum sediment in the upper 4 meters does not extend as far south.

The observed and simulated distribution for 12-13 March 1979 is shown in Fig. 7. The shipboard data's area of maximum distribution extends from the Susquehanna River entrance southward to about 20 kilometers below its previous (9 March) position. The modeled region of maximum sediment in the upper 4 meters extends only as far south as the northern entrance of the Chester River.

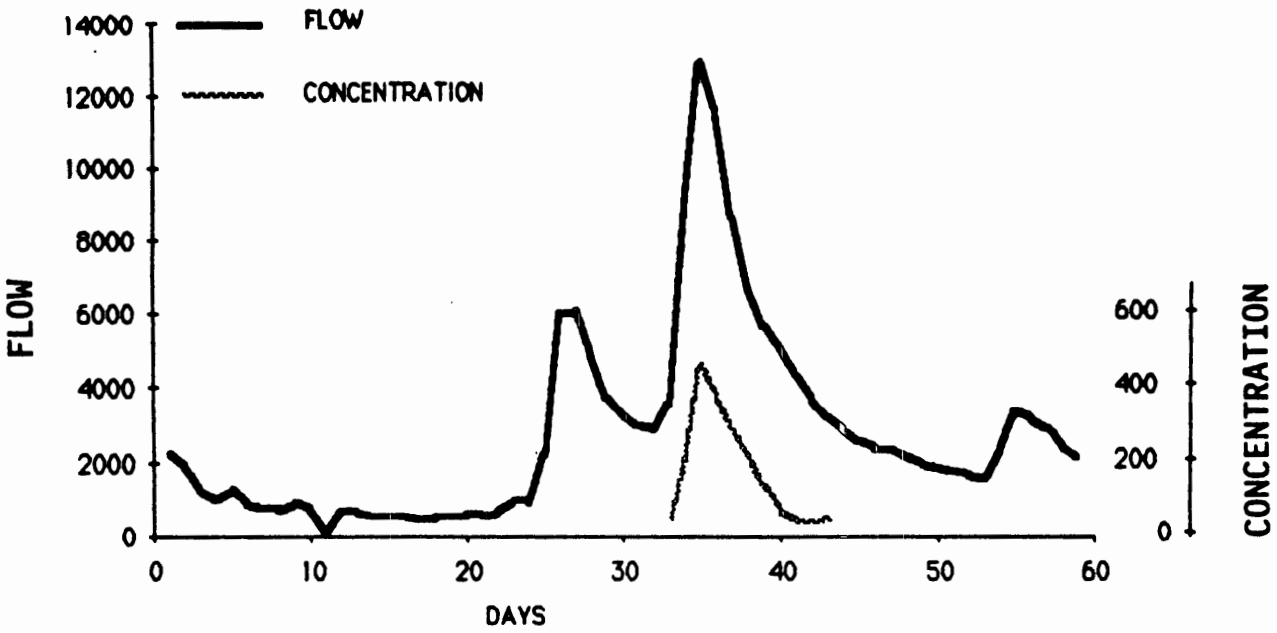
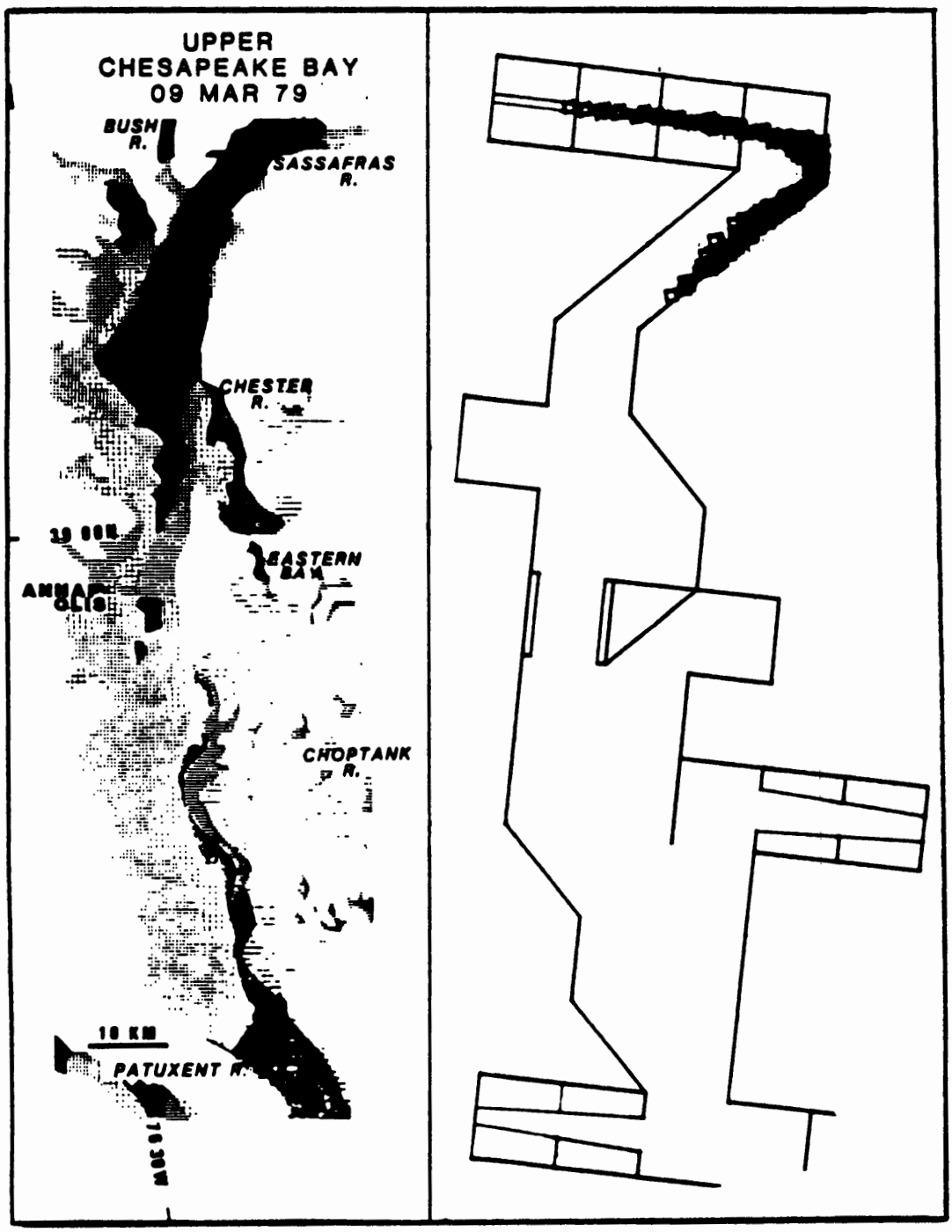


Figure 5. Daily flow and suspended sediment concentration in the Susquehanna River for 1 February - 31 March, 1979. Flow is shown in the units m^3/s and concentration is in units mg/l.



(A) (B)

Figure 6. (A) Observed (satellite) distribution of surface suspended sediments (dark areas indicate high concentration), and (B) simulated positions of Lagrangian drifters (squares) in the upper 4 meters in the upper Chesapeake Bay model grid for 9 March 1979.

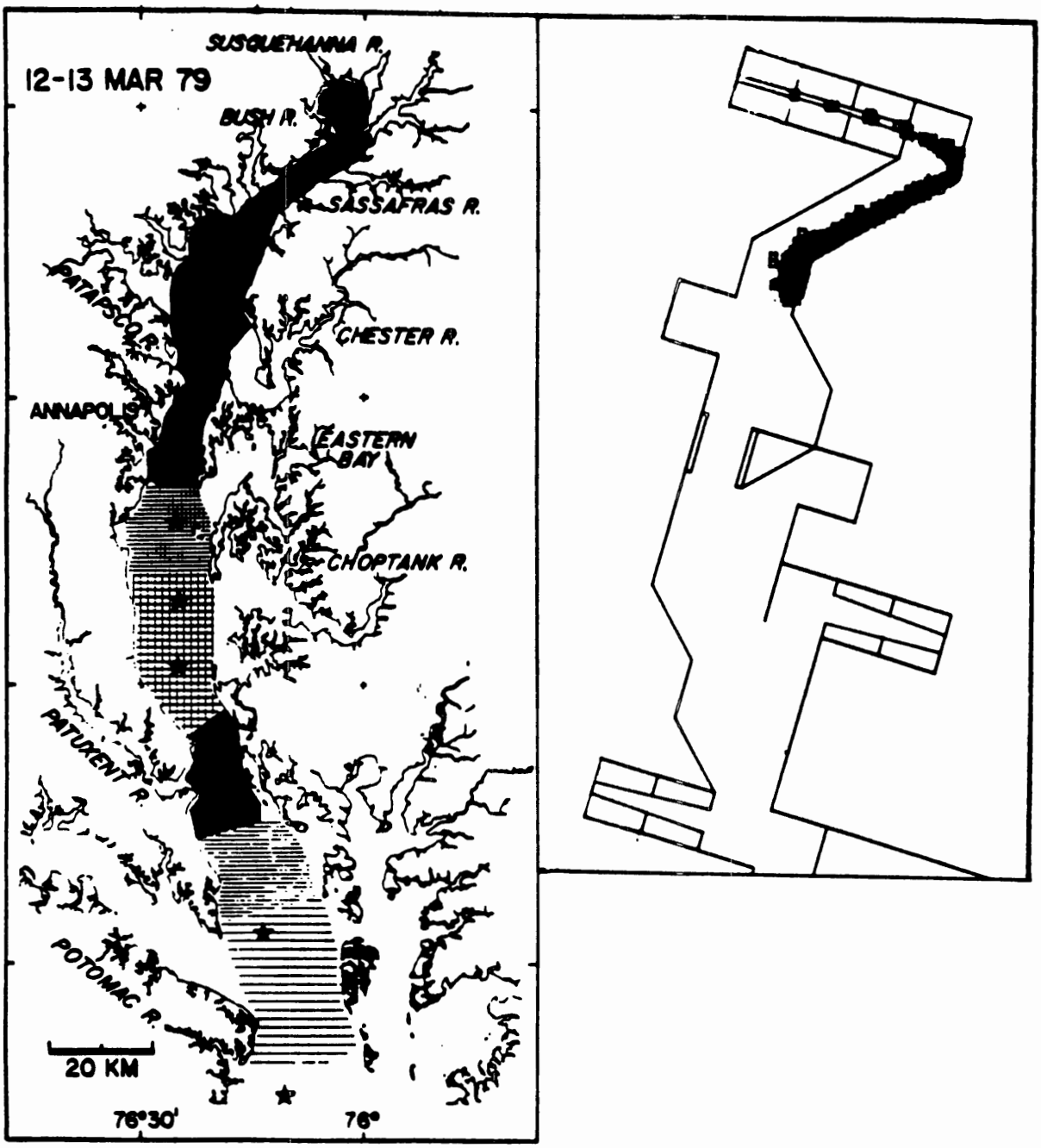


Figure 7. (A) Observed (shipboard measurement) distribution of surface suspended sediments (dark areas indicate high concentration), and (B) simulated positions of Lagrangian drifters (squares) in the upper 4 meters in the upper Chesapeake Bay model grid for 12-13 March 1979.

The observed and simulated distribution for 17 March 1979 is shown in Fig. 8. The remotely-sensed area of maximum distribution extends from the Sassafras River entrance southward to about 10 kilometers south of latitude 39.00 N. The modeled region of maximum sediment in the upper 4 meters does not extend as far south.

The observed and simulated distribution for 26-27 March 1979 is shown in Fig. 9. The shipboard data's shows two areas of high concentrations, one near the Bush River and the other just north of the Patapsco River. The modeled region of maximum sediment in the upper 4 meters shows some patchiness; areas of high concentration are located near Annapolis and the entrance to Chester River.

Many of the simulations showed that modeled sediment was not advected as far to the south as the observations would seem to indicate. To test the influence of the settling velocity on the distribution, another simulation of the March flood was carried out with a settling rate of 0.5×10^{-5} m/s, half the previously used value. There was only a small increase in the area of maximum near-surface concentration.

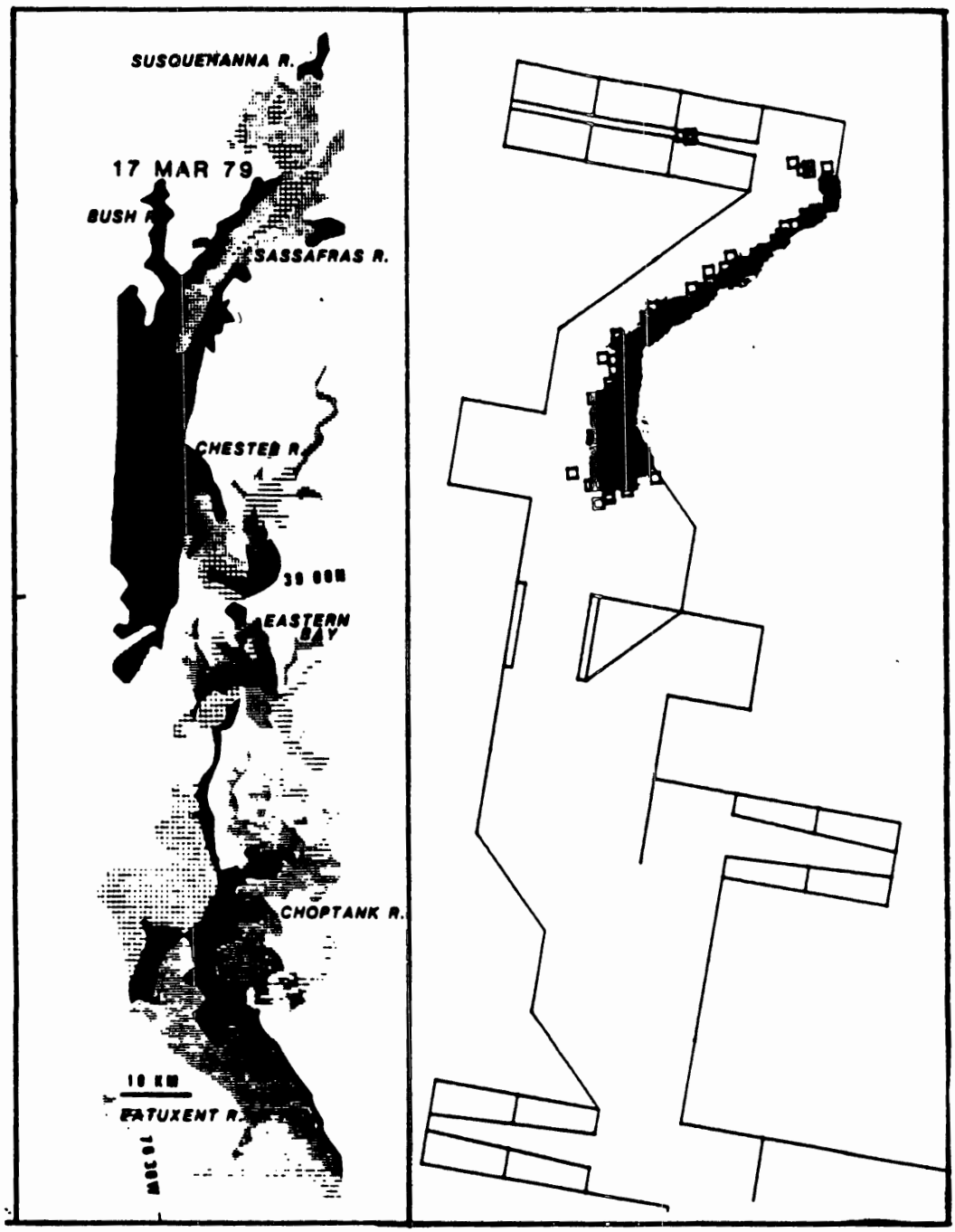
CONCLUSIONS

These results of preliminary computer simulations of suspended sediment transport demonstrate the potential usefulness of the technique. Although the sediment generally followed the expected patterns, in some cases modeled suspended sediment did not move a far south in the Bay as observations would indicate. The simulations may not be unrealistic, however, given the assumptions of no wind and no sediment release before 5 March.

Further investigations should focus on the effects of grid resolution on the currents, influence of density gradients, the representation of vertical diffusion, and the role and modeling of resuspension.

ACKNOWLEDGEMENT

The author would like to thank his colleague Dr. Richard Stumpf for supplying data for this study and for his valuable discussions about remote sensing, modeling, and suspended sediment transport.



(A) (B)

Figure 8. (A) Observed (satellite) distribution of surface suspended sediments (dark areas indicate high concentration), and (B) simulated positions of Lagrangian drifters (squares) in the upper 4 meters in the upper Chesapeake Bay model grid for 17 March 1979.

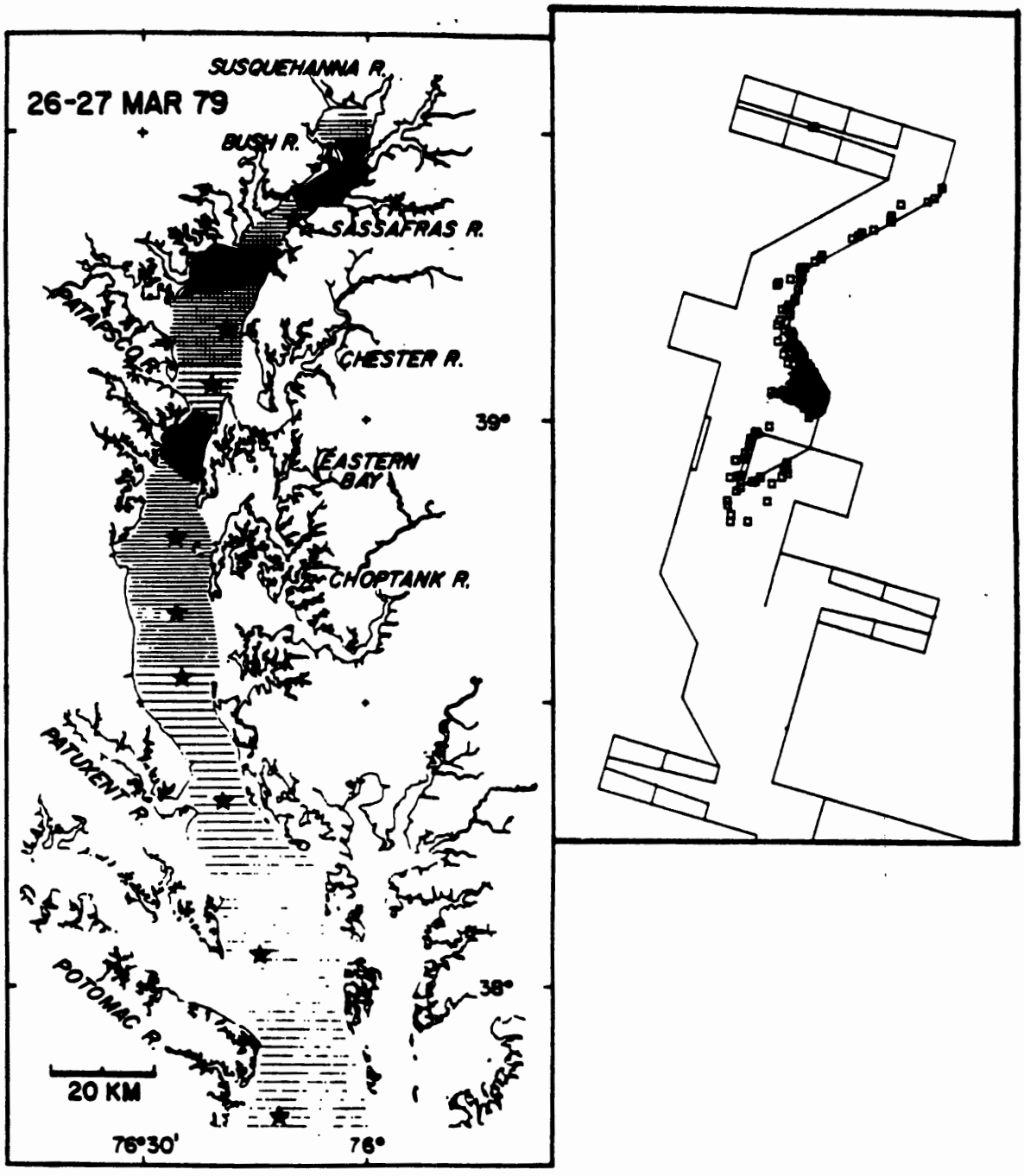


Figure 9. (A) Observed (shipboard measurement) distribution of surface suspended sediments (dark areas indicate high concentration), and (B) simulated positions of Lagrangian drifters in the upper 4 meters in the upper Chesapeake Bay model grid for 26-27 March 1979.

REFERENCES

1. Cronin, W. B.; Taylor, W. R.; Mallonee, M. Suspended sediment distribution and related data, September 1978 - October 1980, Part I. Chesapeake Bay Institute, Johns Hopkins University, Baltimore , Md., CBI Open File Report no. 27; 1982.

2. Csanady, G. T. Turbulent diffusion in the Environment. Reidel, Boston; 1973.

3. Hess, K. W. Assessment model for estuarine circulation and salinity. NOAA Technical Memorandum NESDIS AISC 3, Washington, DC: U.S. Department of Commerce, National Oceanic and Atmospheric Administration; 1985; 39 p. Available from: Marine Environmental Assessment Division, 1825 Conn. Ave., NW, Washington DC 20233.

4. _____. Numerical model of circulation in Chesapeake Bay and the continental shelf. NOAA Technical Memorandum NESDIS AISC 6, Washington, DC: U.S. Department of Commerce, National Oceanic and Atmospheric Administration; 1986; 47 p. Available from: Marine Environmental Assessment Division, 1825 Conn. Ave., NW, Washington DC 20233.

5. Hess, K. W.; Johnson, D. F. Numerical simulations of blue crab larval drift and larval recruitment to Chesapeake Bay. Marine Environmental Assessment Division, Washington, DC: U.S. Department of Commerce, National Oceanic and Atmospheric Administration; 1988; 34 p. Available from: Marine Environmental Assessment Division, 1825 Conn. Ave., NW, Washington DC 20233.

6. Johnson, D. F. A model for the simulation of larval drift. NOAA Technical Memorandum NESDIS AISC 8, Washington, DC: U.S. Department of Commerce, National Oceanic and Atmospheric Administration; 1987; 21 p. Available from: Marine Environmental Assessment Division, 1825 Conn. Ave., NW, Washington DC 20233.

7. Johnson, D. F.; Hess, K. W.; Pytlowany, P. J. Interdisciplinary synoptic assessment of Chesapeake Bay and the adjacent shelf. NOAA Technical Memorandum NESDIS AISC 5, Washington, DC: U.S. Department of Commerce, National Oceanic and Atmospheric Administration; 1986; 90 p. Available from: Marine Environmental Assessment Division, 1825 Conn. Ave., NW, Washington DC 20233.

8. _____. Circulation modeling as an aid to management of the blue crab fishery in Chesapeake Bay. Proceedings of the Tenth National Conference of the Coastal Society, 12-15 October 1986, New Orleans, LA. 1987: 111-118.

9. Lang, D.J. Water quality of the three major tributaries to the Chesapeake Bay: the Susquehanna, Potomac, and James Rivers, January 1979-April 1982. USGS Water Resources Investigation 82-32.

10. Schubel, J. R. Distribution and transportation od suspended sediment in upper Chesapeake Bay. Geological Society of America, Memoir 133, 151-167; 1972

11. Stumpf, R. P. Sediment transport in Chesapeake Bay during floods: analysis using satellite and surface observations. Jol. Coastal Res., 4, 1-15; 1988.

12. United States Geological Survey, 1979. Water Resources Data, Maryland and Delaware, Water Year 1979, Water Data Report MD-DE-79.

Understanding the Estuary: Advances in Chesapeake Bay Research. Proceedings of a Conference. 29-31 March 1988. Baltimore, Maryland. Chesapeake Research Consortium Publication 129. CBP/TRS 24/88.

How to Estimate the Thickness of Benthic Boundary Layers in Estuaries With and Without Tides

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The benthic boundary layer (BBL) is the fluid layer adjacent to the sea bottom. It is usually characterized by strong shear and intensive vertical mixing. In the past the BBL has been modelled as an Ekman layer. However, as was recently shown by the author, in presence of imposed background stable stratification the Ekman boundary layer model failed and the height of BBL is determined by the scale $L_N = u_* / N$, where u_* is the friction velocity based on the bottom stress and N is the buoyancy frequency, which is chosen to characterize the background stratification (can be considered also as initial stratification). It is interesting to apply this idea to the vertical mixing in estuarine systems in presence of strong tides. Some estimates relevant to the description of the variability of heights of BBL in such conditions will be presented.

The purpose of this short paper is to show that the benthic boundary layer (BBL) generated by tidal waves can be strongly influenced by the presence of imposed background stable stratification. since the thickness of BBL is much less than the wavelength, the Prandtl model for laminar BL can be described by the equation

$$\frac{\partial u}{\partial t} - \nu \frac{\partial^2 u}{\partial z^2} = \frac{\partial u_\infty}{\partial t} \qquad (1)$$

where u_∞ is the free stream velocity which in the case of tidal wave on finite depth can be presented as

$$u_\infty = V_0 \cos(kx - \sigma t) \qquad (2)$$

where $V_0 = a\sigma/shkD$, a is wave amplitude, D is depth, k and σ are correspondingly wave number and frequency. The thickness of laminar benthic boundary layer δ in such case is given by the usual expression

$$\delta = \left(\frac{2\nu}{\sigma}\right)^{1/2} = \left(\frac{\nu T}{\pi}\right)^{1/2} \tag{3}$$

where ν is kinematic viscosity, T is wave period. It is well known since the work of Collins (1963) that in natural conditions the BBL is always turbulent even if wave amplitude a doesn't exceed a few centimeters. Since the period of tidal wave T and the characteristic length $\lambda = 2\pi/k$ exceeds the typical time and length scales of turbulence in the BBL, the tidal motion can be considered as a continuous sequence of stationary realizations for the BBL, or at least the parameters of BBL can be treated as slowly varying functions of time and position (compare with T and $\lambda = 2\pi/k$). This justifies the approach when to estimate the height h of a turbulent benthic boundary layer we can introduce an effective value for eddy viscosity K_h in such a way that instead of (3) we can write

$$h = \left(\frac{2K_h}{\sigma}\right)^{1/2} = \left(\frac{K_h T}{\pi}\right)^{1/2} \tag{4a,b}$$

For effective turbulent viscosity in shear driven BBL we can use the simplest expression

$$K_h \approx u_* h \tag{5}$$

Here we avoid a numerical constant of proportionality being of order one. More precisely the expression (5) can be written as

$$K_h \approx \kappa u_* h \tag{6}$$

where $\kappa \approx 0.4$. With (6) the expression (4a) can be rewritten as

$$h = 2\kappa L_\sigma \tag{7}$$

where the scale L_σ is defined as

$$L_\sigma = \frac{u_*}{\sigma} \tag{8}$$

The scaling length L_σ corresponds to the scale of the well mixed region in neutrally stratified tidal BBL and is a fundamental estimate for the thickness of turbulent BL generated by periodic tidal motion. the application of the scale L_σ to the estimates of the thicknesses of benthic boundary layers need first of all its comparison with internal Ekman scale $L_e = u_*/\Omega$ (Ω-Coriolis parameter, which is a good measure of the height of stationary turbulent boundary layer. Since as a rule

$$\frac{L_e}{L_\sigma} = \frac{\sigma}{\Omega} \cong 1 \tag{9}$$

we can conclude that BBL generated by tidal motion in neutrally stratified environment can easily reach Ekman height L_e. A typical range of values of u_* in tidal flows with different bottom roughness conditions are 0.7-4 cm/sec (see Kagan, 1968). For this range of values of u_* the semidiurnal tide will produce BBL of the thicknesses which are comparable with the Ekman height L_e. However some recent observations of the turbulence close to the sea bottom (see for example Ozmidov, 1987) and also the numerical simulation of the formation of BBL in initially continuously stratified fluid subject to a suddenly imposed barotropic pressure gradient demonstrate that the thicknesses of the

well mixed turbulent region close to the bottom of the sea are sufficiently less than the scales L_e and L_σ. Following recent work by the author (S.A. Kitaigorodskii, 1988) it can be explained by taking into account the effect of initial stable stratification N(z,x), where N is Brunt-Vaisals frequency (can be considered as slowly varying function of time and position). In this case the growth of BBL is limited, since the flux Richardson number Rf_h in the vicinity of the entrainment zone must be less than its limiting value Rf_{lim}. The basis for the estimate of flux Richardson number Rf_h is the following simple formula for the entrainment buoyancy flux Q_h

$$Q_h = -K_h N^2 \tag{10}$$

The above mentioned condition for Rf_h can be written as

$$Rf_h = \frac{-Q_h}{u_*^2 du/dz} \le Rf_{lim} \tag{11}$$

There are two ways to proceed further. First is based on the assumption that the major contribution to the du/dz comes from the constant flux region of BBL, which is not yet influenced by rotation. Then we can write that

$$\frac{du}{dz} = \frac{u_*^2}{K_h} \tag{12}$$

which permits to rewrite (11) using again (5), (6) and (10) as

$$h = h_{max} = bL_N \quad ; \quad b = \frac{Rf_{lim}}{\varkappa} \quad ; \quad L_N = \frac{u_*}{N} \tag{13}$$

where the scaling length L_N corresponds to the scale of the well mixed region developed by upward diffusion of turbulent energy in the presence of a background stable stratification and b is a numerical coefficient. Its value can be estimated by taking $Rf_{lim} \approx 0.10\text{-}0.15$ and $\varkappa \approx 0.4$ as 2-3, which is not too far from empirical findings, based on the data from stratified atmospheric boundary layers (Kitaigorodskii, 1988). The application of the scale L_N to the estimates of the heights of BBL generated by tidal waves need first of all its comparison with the scale $L_\sigma = u_*/\sigma$. The expression (13) can be rewritten as

$$h_{max} = bL_\sigma \left(\frac{\sigma}{N}\right) \tag{14}$$

and since as a rule for semidiurnal and diurnal tides $\sigma/N \ll 1$, we can conclude that in the presence of imposed stable stratification the BBL generated by tidal motions never reaches equilibrium on the typical Ekman scale $L_* \sim L_\sigma$, but rather achieves a quasi-equilibrium thickness characterized by the plate $h_N = b\, u_*/N$. If instead of the assumption (13) we will estimate shear as $du/dz \sim u_*/h$, then instead of (14) we can write (by using (2))

$$h = bL_N \left(\frac{a\sigma}{u_* shkD}\right)^{1/2} = bL_\sigma \left(\frac{\sigma}{N}\right)\left(\frac{a\sigma}{u_* shkD}\right)^{1/2} \tag{15}$$

It means that the growth of tidal BBL is limited not only by the scale L_N (in the presence of imposed stable stratification) but also by the magnitudes of tidal motions. For finite depth D and given kD (let us say of order one) (17) can be rewritten as

$$h = bL_N \left(\frac{a}{L_\sigma}\right)^{1/2} \tag{16}$$

which demonstrates that if the tidal wave amplitudes are very small the BBL can become very thin compared both with Ekman height L_e and height $h_N = b\,L_N$. The detailization of (17) can be done by assuming some form for the bottom stress u_*^2. If we'll use the classical quadratic law $u_*^2 \sim U_1^2 \sim a^2\sigma^2$ then

$$\frac{a}{L_\sigma} = \frac{a\sigma}{u_*} = const \qquad (17)$$

Even though $a\sigma/u_* \sim 10^2$ the square root of it in (17) permits still to write that

$$h \approx L_N = L_\sigma \frac{\sigma}{N} \qquad (18)$$

Therefore in the case of imposed background stratification we can simply write that the thickness of equlibrium turbulent boundary layer are defined only by two scales L_N and L_e, so that

$$h = L_N Y(\mu_\sigma) \qquad (20)$$

where

$$\mu_\sigma = \frac{L_N}{L_\sigma} = \frac{\sigma}{N} \qquad (21)$$

The two asymptotic expressions for Y (μ_σ) will correspond to $\mu_\sigma \to \infty$ (neutral BBL) when $h \sim L_\sigma$ and $\mu_\sigma \to 0$ when $h \sim L_N$.

References

Collins, J. I. Inception of turbulence at the bed under periodic gravity waves. J. Geophys. Res. 68: No. 21; 1963.

Kagan, B. A. Fluid dynamical models of tidal motions in the sea. Leningrad: Hydrometeorological Publishing House; 1968.

Kitaigorodskii, S. A. A note on similarity theory for atmospheric boundary layers in presence of background stable stratification. Tellus: [1988 in press].

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Abstract only

High-Resolution Thermistor Chain Observations in the Upper Chesapeake Bay

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Selected results from four years (1984-1987) of thermistor chain data and coincident CTD, current meter and acoustic backscatter measurements are presented. The data shown exemplify a number of super-tidal features which were ubiquitous during the measurement periods. The features are subsurface intrusions with mixing and high-frequency internal waves on their surfaces, estuarine surface fronts, monochromatic, high-amplitude internal wave trains, breaking internal waves and broader-band internal wavefields. These features are discussed in light of their effects on mixing and transport, and their implications for monitoring sampling strategies and interpretation of results.

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Satellite Observations of Turbidity Variations in Chesapeake Bay, Spring-Summer 1987

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INTRODUCTION

The transport and distribution of suspended sediments can have a strong influence on the living resources of Chesapeake Bay through their effect on turbidity and siltation. The suspended matter shows considerable spatial and temporal variability--particularly with changes in river discharge. As these variations may occur within a few days within portions of the Bay system; frequent, synoptic observations are needed to evaluate the changing conditions.

Satellite sensors may provide routine synoptic coverage of the turbidity or sediment loads in the surface waters of the Bay. Although satellites have been used as a means of routine monitoring of chlorophyll and temperature in the ocean (e.g. Brown et al. 1985), they have not been used for routine studies of estuaries. The experimental Coastal Zone Color Scanner (CZCS) had some potential for this purpose, however some of the bands did not have the dynamic range to handle turbid water. (It was also shut down in June, 1986.) Landsat and SPOT sensors provide only 1-2 images per month, therefore the frequency of usable (cloud-free) images is too low to monitor changes caused by episodic events. In contrast to these other sensors, the Advanced Very High Resolution Radiometer (AVHRR) can provide daily images with usable radiometric information; therefore it appears to be the best sensor presently available to investigate estuaries.

Currently, this sensor can provide reliable estimates of sea surface temperatures (Strong and McClain 1984). It has been used to detect chlorophyll blooms in turbid water (Stumpf and Tyler 1988), and through

the calculation of water-reflectance from the sensor data, Stumpf (1987, 1988a, 1988b) has shown the ability of this sensor (and others) to provide estimates of suspended sediments (seston) or turbidity measures such as the Secchi depth or attenuation coefficient. This paper will show the utility of this sensor in monitoring short-term and seasonal changes in turbidity, as defined by reflectance, in Chesapeake Bay using data from 1987.

METHODS

Satellite Characteristics

The AVHRR is onboard the NOAA TIROS-N (Television and Infrared Observation Satellite) platform, which is polar-orbiting and sun-synchronous. NOAA-6, 8, and 10 overpasses occur at about 0800 and 2000 local time. NOAA-7 and NOAA-9 overflights occur about 1430 and 0230 local time. During 1987, the NOAA-9 and NOAA-10 satellites were operational. The satellites have approximately a 9-day cycle and track from west to east, producing about 6-7 potentially usable daytime scenes during each cycle.

The AVHRR scans orthogonally to the direction of travel over an angle of $\pm 55.4^{\circ}$ from nadir. The usable scene width is about 2000 km. The sensor has a field of view (pixel size) of 1.4 milliradians, which corresponds to a ground diameter of 1.1 km at nadir. The satellite collects 360 scanlines per minute.

The AVHRR has either 4 or 5 bands as shown in Table 1. Channels 3, 4, and 5 are used to calculate sea surface temperature. Channels 1 and 2 are used here in the calculation of water reflectances. The specifications of the AVHRR are described in detail in Kidwell (1985) and Lauritson et al. (1979). Stumpf (1987) and Everdale (1986) present example treatment and applications of the data to coastal oceanography.

Table 1. AVHRR Spectral Bands

channel	1	2	3	4	5*
wavelength (μm)	.58-.68	.72-1.0	3.5-3.9	10.5-11.3	11.5-12.5
description	red	near-IR	thermal - infrared		

*Available only on NOAA-7 and NOAA-9.

Processing

The images were obtained as Level 1B digital data tapes from NOAA's Satellite Data Services Division. The images were mapped to a Mercator projection with a pixel size of 1.18 km at 38°N using a nearest neighbor routine, and missing data points were filled with a 3x3 average. An additional linear correction produced a positioning accuracy of 1 pixel

for the shoreline. Both reflectances and SST's were calculated for each image. The temperatures, not presented here, were calculated using the equations in Kidwell (1985).

The reflectance is the ratio of irradiance leaving the water to the irradiance entering the water. This is estimated using

$$R(\lambda) = \frac{\pi\, L_w(\lambda)}{E_o(\lambda) \cos\theta_o \exp(-[\tau_r(\lambda)/2 + \tau_{oz}(\lambda)]/\cos\theta_o)} \quad (1)$$

where

$$L_w(\lambda) = [L_*(\lambda) - L_A(\lambda)] \,/\, T_1(\lambda) \quad (2)$$

R is the above water reflectance, λ is the spectral band, L_w is the radiance from the water, L_* is radiance at the sensor, L_A is the atmospheric path radiance, T_1 is the transmission coefficient from the earth to the sensor, E_o is the solar irradiance outside the atmosphere, θ_o is the solar zenith angle, τ_r is the Rayleigh optical depth, τ_{oz} is the gaseous (e.g., ozone) absorption optical depth (Stumpf 1988b). Equation (2) is an atmospheric correction assuming uniform atmosphere over the scene area. This acts as a bias correction for both Rayleigh scattering radiance and minimum aerosol radiance (Stumpf 1988b).

An additional correction is performed to remove variable haze and clouds (Stumpf 1988b). This correction is

$$R_D = R(1) - YR(2) \quad (3)$$

where R_D is the cloud-corrected reflectance; R(1) and R(2) are determined from (1); and Y is a constant determined by the aerosol characteristics. This correction has the advantage of removing atmospheric contamination pixel-by-pixel. Here, Y is set equal to 1.0, which applies to aerosols having relatively large particles, such as clouds, that produce achromatic haze. Heavy overcast and some (colored) haze types cannot be removed and generally produce elevated reflectances. Overcast was masked in white using a combination of temperature and reflectance data.

R_D has a precision within about ± 0.004 depending on the presence of uncorrectable haze. The accuracy cannot be readily determined, owing to uncertainties in the sensor calibration. There is evidence that the NOAA-9 AVHRR channel 1 and channel 2 detectors have shown a decrease in sensitivity since launch, however the sensor has no onboard method of calibrating this change. Currently, the calibration has not been corrected for changes in sensitivity, therefore reflectances shown in this report may be somewhat less than the true reflectances (or those observed when the NOAA-9 was first launched in 1985). Within the 7-month period investigated here, changes in the sensor should remain slight.

Stumpf (1987) has shown that the reflectance can be related to the

suspended sediment concentration through the relationship

$$R = \frac{.33\ b_{bs}^*}{S^* + a_x/n_s} \qquad (4)$$

where n_s is the sediment concentration; b_{bs}^* is the specific backscatter coefficient for the sediment; $S^* = a_s^* + b_{bs}^*$, where a_s^* is the specific absorption coefficient of materials related to the sediments; and a_x is the absorption coefficient for water, dissolved and algal pigments. Constant values of b_{bs}^* and S^* can be used to describe an estuary over a range of data. Variations in a_x, such as would occur in strong blooms, may affect the relationship, and may reduce the value of R_D somewhat below the true value (Stumpf 1988b). The form of this relationship causes R to vary approximately with the logarithm of n_s.

Monthly mean reflectances were determined using four weekly scenes per month. The spatial means were found using all the pixels within a geographic area. The upper Chesapeake extends from 39°27'N to 39°02'N, the middle Bay from 39°02'N to 38°22'N, the lower Potomac from 77°01'W (Morgantown) to the entrance, and the lower James from 76°41'W to the entrance.

River flow data was obtained from USGS gaging station data. The stations are at Conowingo Dam, Maryland for the Susquehanna, near Chain Bridge at Washington, DC for the Potomac, and Cartersville--about 60 km above the fall line at Richmond--for the James.

RESULTS

River Flow

The three major tributaries to the Bay, the Susquehanna, Potomac, and James Rivers, showed their highest flows of the year in March and April 1987 (Figure 1). Peaks occurred about March 3 in the Potomac and James. The Susquehanna had high flow about March 11. In April, the Susquehanna and Potomac Rivers had high flow events somewhat early in the month, both rivers peaked about April 6 and the Potomac had its maximum peak for the year about April 18. The James peaked twice later in April, with flood conditions about April 18 and high flow on April 26. Flows from June to September corresponded to those in May, with no high flow events through the summer.

The satellite imagery shows variations in reflectance consistent with the expected influx of sediments during these events. As an example, Figures 2 and 3 show reflectances on April 1 and April 10. The Chesapeake and lower Potomac have low to moderate reflectances (<0.025) on April 1, with the James showing moderately high reflectances (0.03). On April 10, after the Potomac and Susquehanna had flow peaks about April 6, their estuaries showed high reflectances. The Susquehanna plume extended into the middle Bay below Annapolis (marked A in the figures). The turbidity plume from the Potomac extended into the main Bay, with reflectances of double those of April 01, even though the flow peak was not exceptionally large for this river. In contrast the James estuary had maintained approximately the same reflectance, consistent

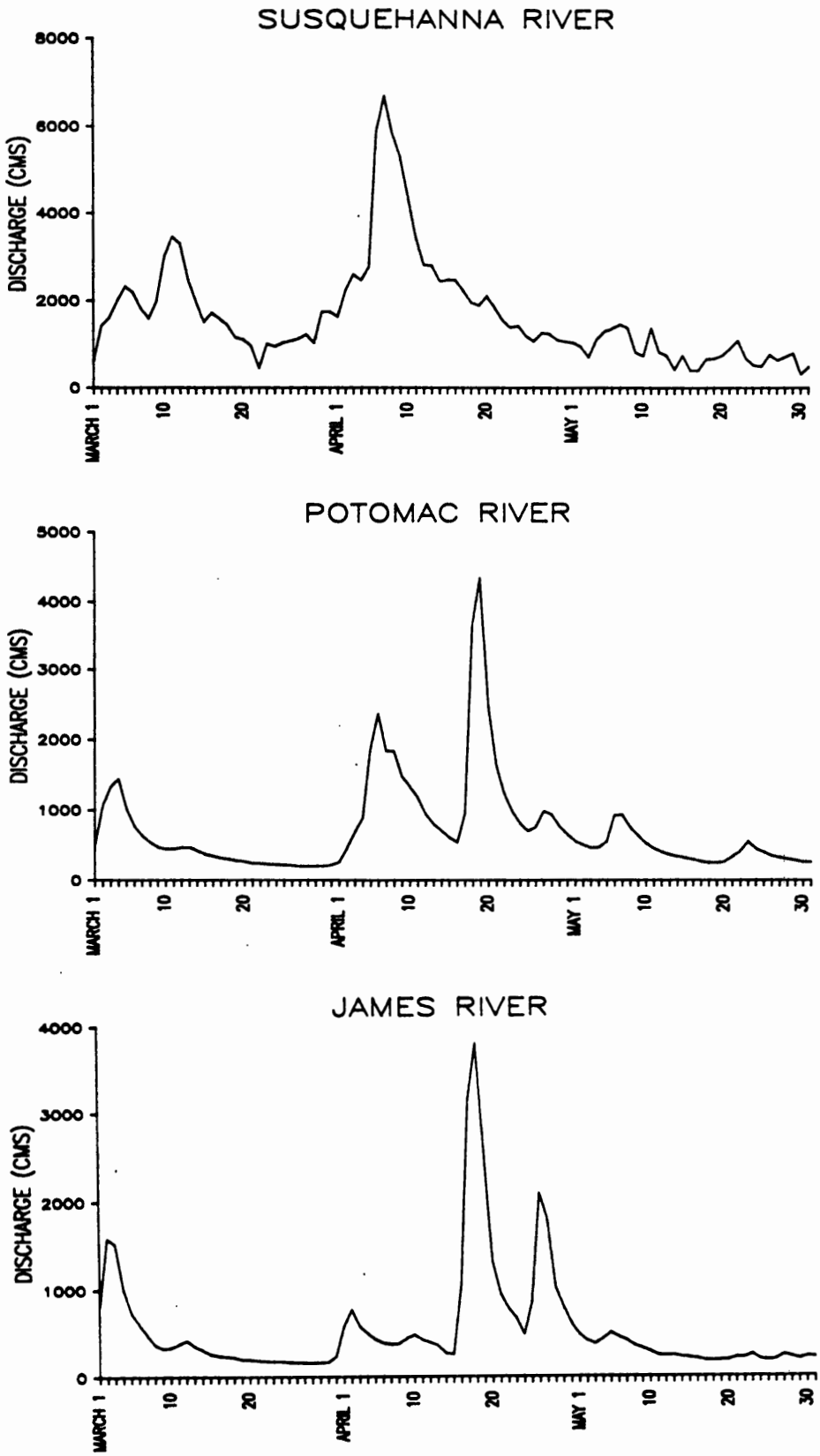

Figure 1. River discharge for the Susquehanna, Potomac, and James Rivers, Spring of 1987.

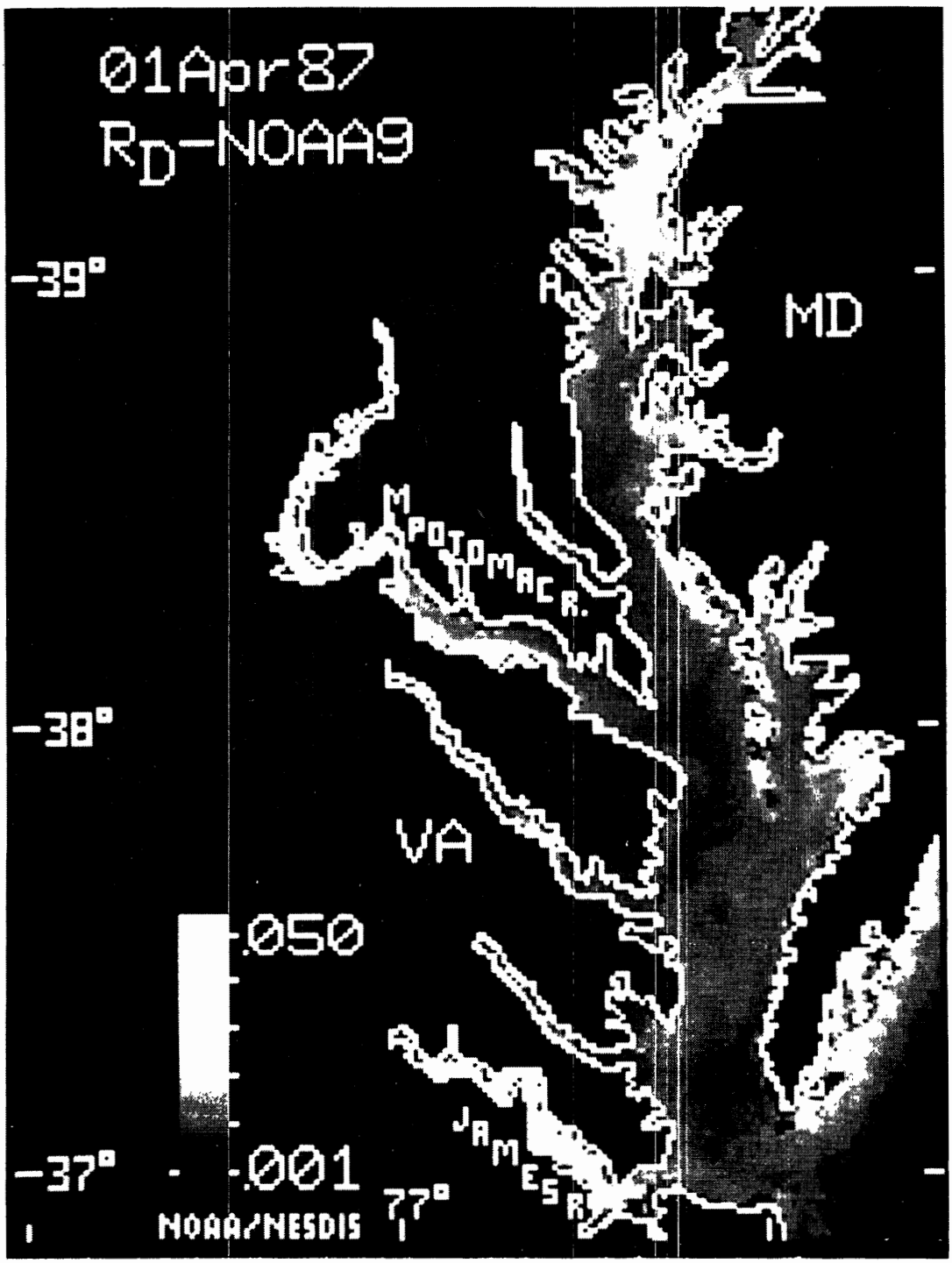

Figure 2. R_D reflectances for Chesapeake Bay found from NOAA-9 data on 01 April 1987.

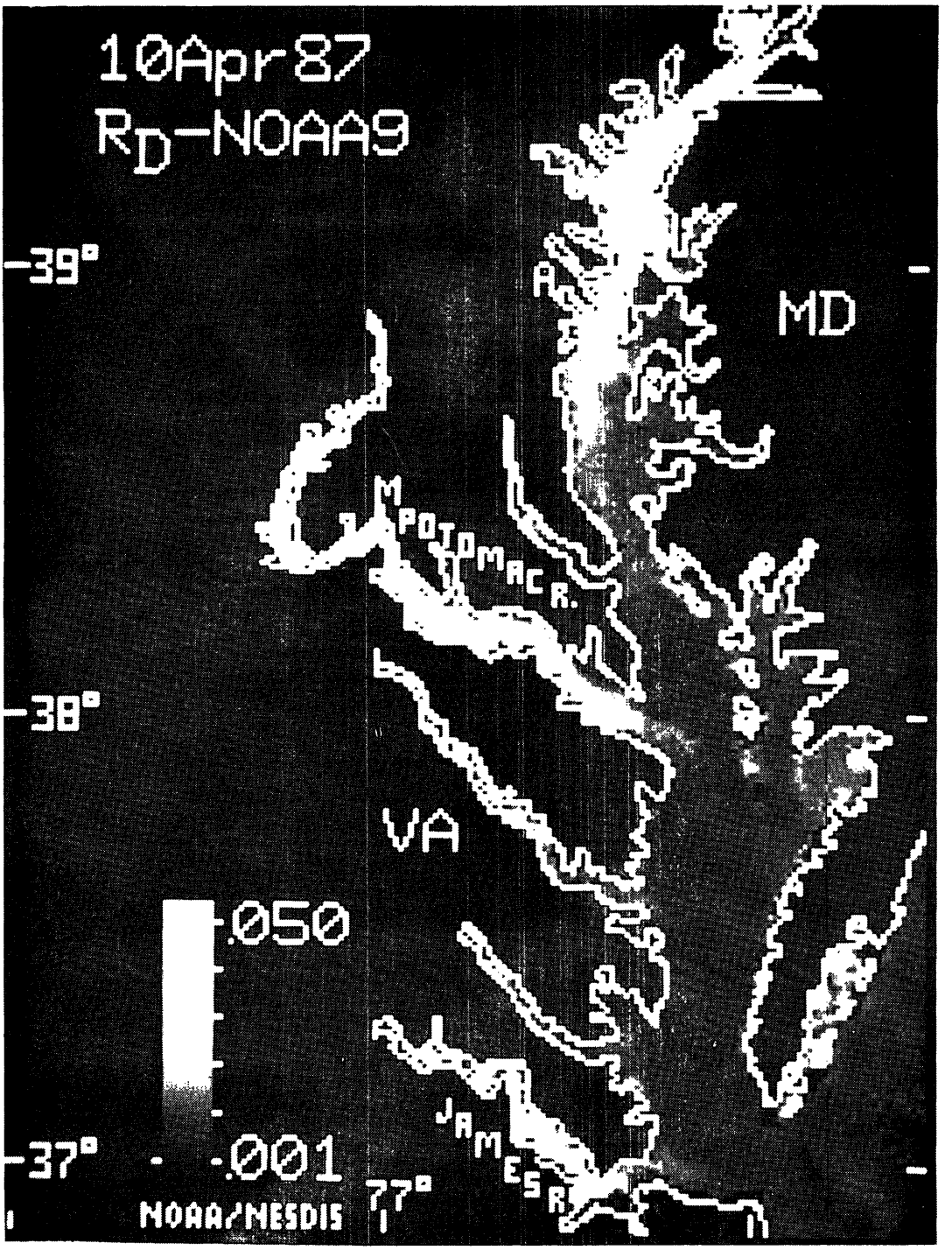

Figure 3. R_D reflectances for Chesapeake Bay found from NOAA-9 data on 10 April 1987.

with the lack of a strong peak in discharge.

The high flow events clearly carry material into the lower portions of the estuaries (Figure 4). The early March event apparently caused increased mean reflectances in the lower Potomac and James. In contrast, the March 5 and March 11 peaks in the Susquehanna did not carry sufficient material into the middle Bay to affect the mean reflectances. The April 6 peak in the Susquehanna (highest of the year) did carry sediment well into the middle Bay (Figure 3). The lower Potomac also had increased reflectances resulting from the April 6 and April 18 peaks.

Reflectances in the lower James peaked later in April, consistent with the later peak in the river discharge. The Potomac and Chesapeake reflectances decreased to summertime levels by mid-May. The lower James finally decreased to a mean reflectance of 0.015 by late June.

Monthly Mean Reflectance Variations

The variations in mean reflectances over the Bay during the spring and summer followed the seasonal changes in river flow. Total discharge into the Bay was high in March, increasing in April and decreasing through the summer (Figure 5). The mean reflectances were moderate in March and increased in April in the lower portions of the major tributaries and the middle and upper Bay (Figure 6). The means in these areas decreased into May-June. The James, which started with far higher reflectances, showed continued decreases into July-August.

Tangier Sound and a turbidity zone along the Virginia western shore below the Rappahannock maintained moderate reflectances throughout the period. As these two areas are not directly associated with any rivers, other factors beside discharge would determine their turbidity levels.

CONCLUSIONS

The reflectance data collected from the AVHRR can show variations in the turbidity or suspended sediment loads of the Bay. The imagery identifies the positions of turbid plumes caused by high river flow and reveals fronts and small scale temporal changes. Examination of the series of images shows the effects of individual high flow events. Moderate discharge can increase the turbidity of the estuarine portions more than 60 km from the fall line.

As the estuaries may take several weeks to clear, the timing of these events may be critical in determing the productivity of these portions of the Bay system during the spring and early summer. Continued analyses of these events, particularly in comparison with long-term averages, could establish the frequency and importance of the high turbidity events.

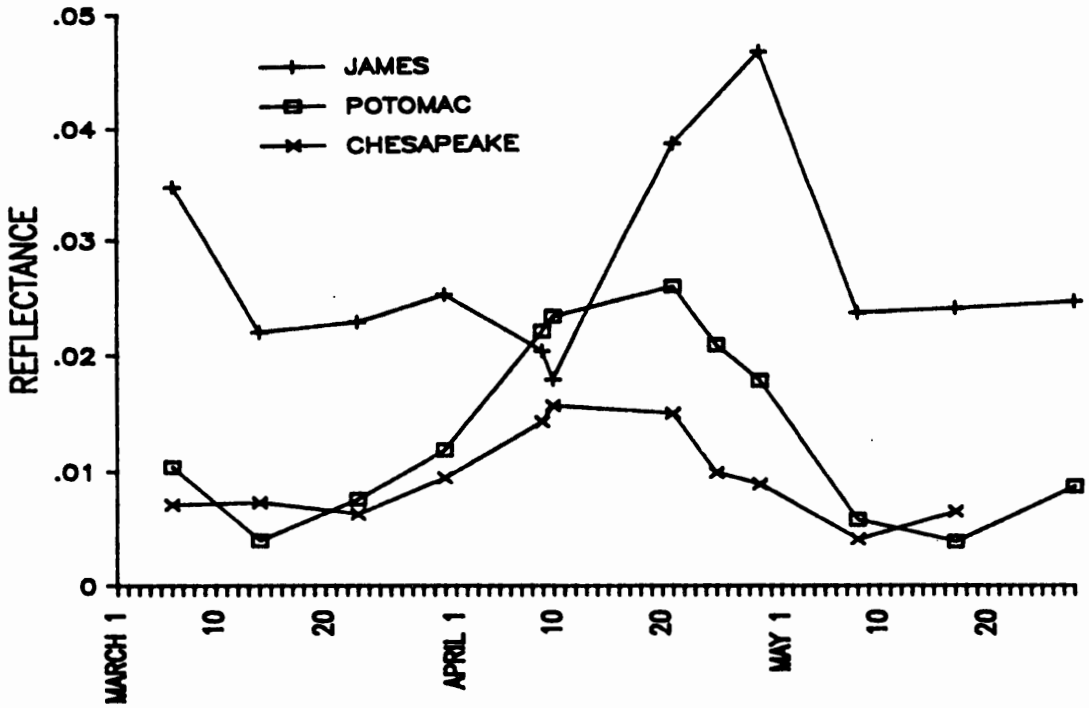

Figure 4. Variation in spatially averaged R_D reflectances in the middle Chesapeake Bay, lower Potomac, and lower James, 1 March - 28 May 1987.

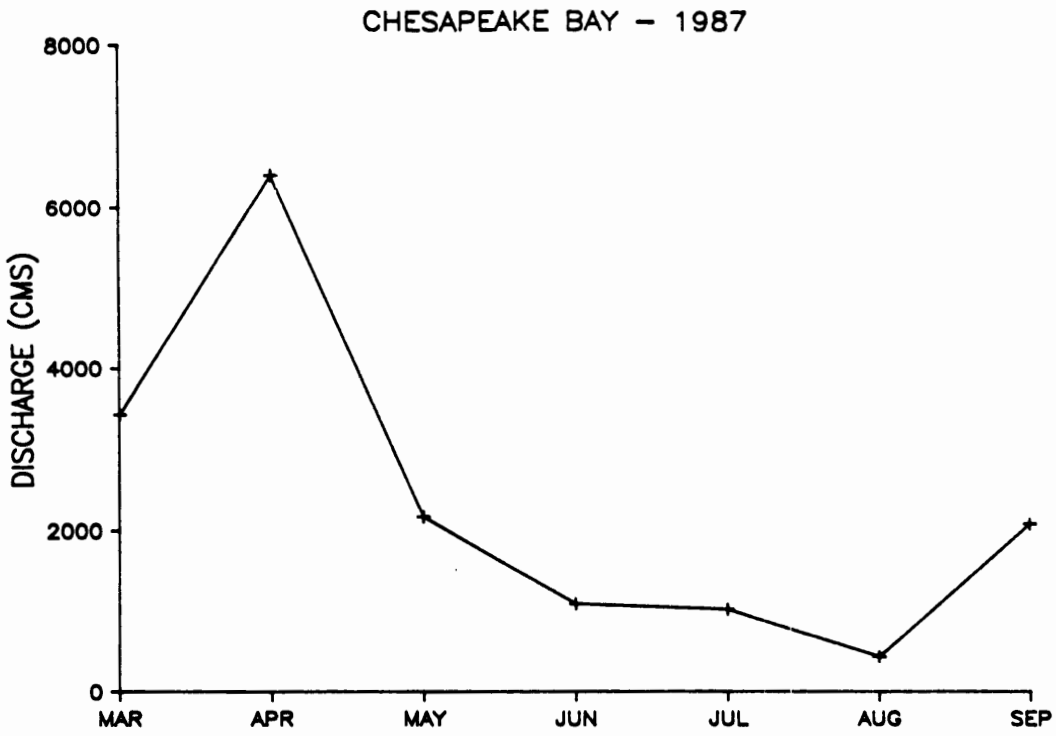

Figure 5. Mean monthly flow rate into the Chesapeake Bay in 1987.

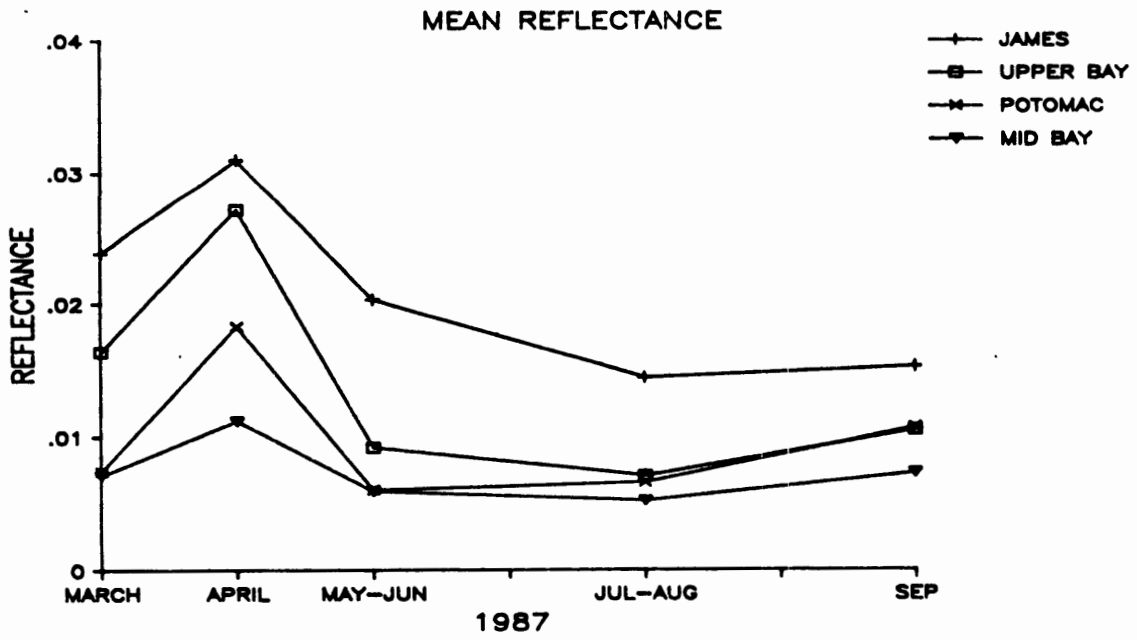

Figure 6. Monthly mean R_D reflectances the lower James, lower Potomac, upper and middle Bay, for March through September, 1987.

REFERENCES

Brown, O.B.; Evans, R.H.; Brown, J.W.; Gordon, H.R.; Smith, R.C.; Baker, K.S. phytoplankton blooming off the U.S. East Coast: A satellite description. Science 229:163-166; 1985.

Everdale, F.G. Satellite Oceanography - Volume I: NOAA-n AVHRR Digital Data. 63 p. 1986; Technical Memorandum NESDIS AISC 7, National Oceanic and Atmospheric Administration, Washington, DC.

Kidwell, K.B. NOAA Polar Orbiter Data (TIROS-N, NOAA-6, NOAA-7, NOAA-8, NOAA-9, NOAA-10) Users Guide. 1987; National Oceanic and Atmospheric Administration, National Environmental Satellite, Data, and Information Service, Washington, D.C. 20233.

Lauritson, L., Nelson, G.J., Porto, F.W. Data Extraction and Calibration of TIROS-N/NOAA Radiometers. 1979; Technical Memorandum NESS 107, with amendments, National Oceanic and Atmospheric Administration. Washington, DC.

Strong, A.E.; McClain, E.P., Improved ocean surface temperatures from space--comparisons with drifting buoys. Bull. American Meteorological Society, 65:138-142; 1984.

Stumpf, R.P. Application of AVHRR Satellite Data to the Study of Sediment and Chlorophyll in Turbid Coastal Water. 1987a, Technical Memorandum NESDIS AISC 7, National Oceanic and Atmospheric Administration, Washington, DC, 50 p.

Stumpf, R.P. Sediment transport in Chesapeake Bay during floods: analysis using satellite and surface observations. Journal of Coastal Research, 4(1):1-15; 1988a.

Stumpf, R.P., Remote sensing of suspended sediments in estuaries using atmospheric and compositional corrections to AVHRR data: 1988b, Proc. 21st International Symposium on Remote Sensing of Environment, ERIM, Ann Arbor, Michigan, October 26-30, 1987, in press.

Stumpf, R.P.; Tyler, M.A. Satellite detection of bloom and pigment distributions in estuaries. Remote Sensing of Environment, in press; 1988.

Understanding the Estuary: Advances in Chesapeake Bay Research. Proceedings of a Conference. 29-31 March 1988. Baltimore, Maryland. Chesapeake Research Consortium Publication 129. CBP/TRS 24/88.

Evaluation of Conowingo Reservoir Release for Controlling Salinity in the Upper Chesapeake Bay

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ABSTRACT

A dynamic system model with both control and predict phases is developed to achieve the effect of controllable input on an output variable combined with other uncontrollable inputs. This system uses variable coefficients and a self-tuning scheme, recursive computation algorithms, and small computational and storage requirements. Testing of present regulations will evaluate alternative release policies for potential use. This model will demonstrate the impacts of regulated riverflow from the Conowingo Reservoir on salinity in a tidal dynamic Chesapeake Bay system. Sub-tidal heights and local wind stress are considered as the uncontrollable inputs in this model. The effects on fishery spawning and drinking water are evaluated by this system as well.

INTRODUCTION

Riverflow from the Susquehanna River is discharged into the Chesapeake Bay after being regulated by a multi-objective reservoir, the Conowingo Reservoir. The circulation patterns of the Bay, typical of an estuary, are dominated mainly by tidal fluctuations, seasonal freshwater discharge, and wind

forces on the water surface. Maintaining water quality in the Bay, especially salinity, is a major concern for operation of this reservoir. The Upper Chesapeake Bay and Elk River are important rockfish (Morone saxatilis) spawning areas. Salinity levels favorable to rockfish larvae spawning and growth are between 0.0 and 5.00 ppt. Their spawning period usually occurs in the spring, which is classified as a high flow season in this area. The salinity level (chloride) of the Susquehanna must remain 250 ppm or less to meet drinking water standards for the city of Havre de Grace, downstream. New release strategies can be implemented when the required riverflow is determined. This estimate is derived by testing specified salinity levels in certain critical areas. Recent research (Hsieh 1987) indicates that the Susquehanna river-flow, Chesapeake Bay sub-tidal signals and the cross-Bay component of wind stress from the Delaware Bay mouth are the most significant forcing functions that influence daily salinity levels in the Elk River area. A multiple-input control and prediction model with self-tuning scheme and variable forgetting factor modules is used to evaluate this water resources management problem. The results of this study could be linked to a regional control center as one of the sub-systems; In this way, all specialized control loops could be used as a whole to address the management needs of the entire river basin.

There are several advantages of using the system model approach: When only select critical areas need to be evaluated for environmental effects, a model which connects mechanism sources can simplify the complexity of the problem; The system model reduces data requirements for model calibration and verification, and simulation costs for designing long-term engineering plans; The self-tuning scheme can be used to install a telemetry computer system for designing future automatic control facilities.

The design for a tidal water quality control system is shown in Figure 1. In this design, the system output, salinity, is the function of three separate inputs. The controller, combined with the cost function and system model, calculates the required riverflow when a desired or set-point salinity is proposed. System parameters and coefficients of the controller are adjusted by a recursive algorithm for each time step, after initial conditions are given. The system output shows a strong non-linearity when compared with each system input. Usually, a non-linear transfer function can be used to simulate the approximate system coefficients. However, in a noisy water environment, the order selection is problematic.

The higher-order system model is not practical for this calculation, whereas the lower-order model results in less accuracy. This problem can be solved by using recursive processes. Kalman's estimation method includes variable forgetting factor (weighting factor), and therefore enables the parameter estimates to follow both gradual and sudden

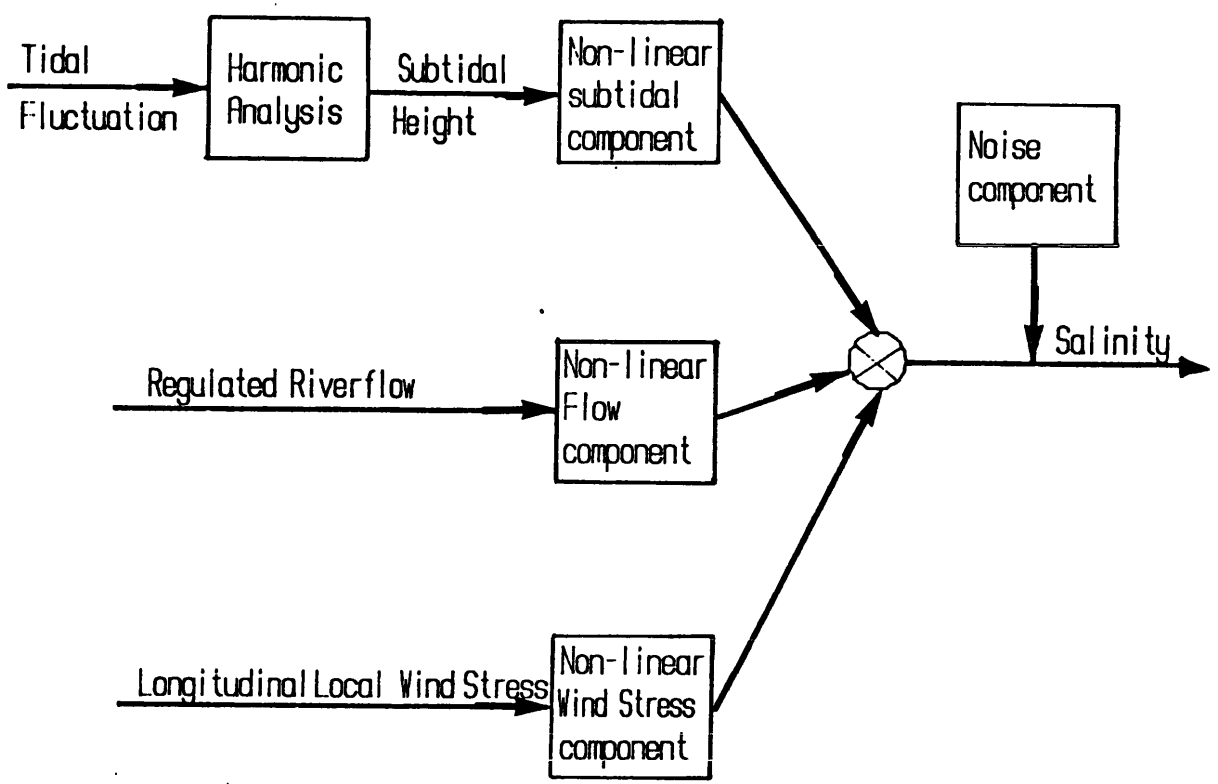

Figure 1. Salinity variation due to natural forcing system in an estuary environment.

changes in the dynamic system (Fortescue et al. 1981). Use of a variable forgetting factor with the correct choice of information bounds can avoid difficulties associated with constant exponential weighting of past data. Recursive estimation methods are no longer required to generate an optimal solution but the initial covariance can still be accurately estimated using this process.

MULTIVARIATE CONTROL AND PREDICT MODELS

Astrom (1970), Astrom and Wittenmark (1973), and Clarke and Gawthrop (1975) introduced and developed a self-tuning regulator or controller, which uses a stochastic difference equation as a system identification algorithm, and a feedback control law to minimize the variance of the output function. The basic self-tuning controller structure has been expanded and is now applicable to multivariate cases (Borisson 1975, 1979; Koivo 1980; and Bayoumi et al. 1981). Ganendra (1978) used a self-tuning controller with real-time control of release from a river-regulating reservoir. Ganendra (1976) also used a self-tuning predictor to develop a riverflow forecasting model.

To satisfy alternative release policies and downstream water quality control needs, a self-tuning predictor is used in conjunction with a self-tuning controller to generate the observed output (salinity). The generated observed output and original uncontrollable inputs, coupled with an alternative release policy, produce the optimal required riverflow through the controller. The individual system designs for the self-tuning controller and predictor are shown in figures 2a and 2b. A self-tuning controller with unknown parameters and variable forgetting factor is used to define the estuary's water quality for this multiple-input case. The governing equations for this model are summarized as:

$$A(q^{-1})y(t) = B(q^{-1})u(t-k_1) + H(q^{-1})T_C(t-Kc_)$$
$$+ M(q^{-1})WS(t-K_w) + C(q^{-1})e(t) \qquad (1)$$

where q^{-1} = backward shift operator.
y = system output (salinity).
u = system input/control variable (riverflow).
T_C = sub-tidal heights from the Chesapeake Bay mouth.
WS = wind stress from the Delaware Bay mouth.
e = a sequence of independent random numbers (noise).
$A = 1 + a_1q^{-1} + \ldots + a_nq^{-n}$.
$B = b_0 + b_1q^{-1} + \ldots + b_nq^{-n}$.
$C = 1 + c_1q^{-1} + \ldots + c_nq^{-n}$.
$H = h_0 + h_1q^{-1} + \ldots + h_nq^{-n}$.
$M = m_0 + m_1q^{-1} + \ldots + m_nq^{-n}$.

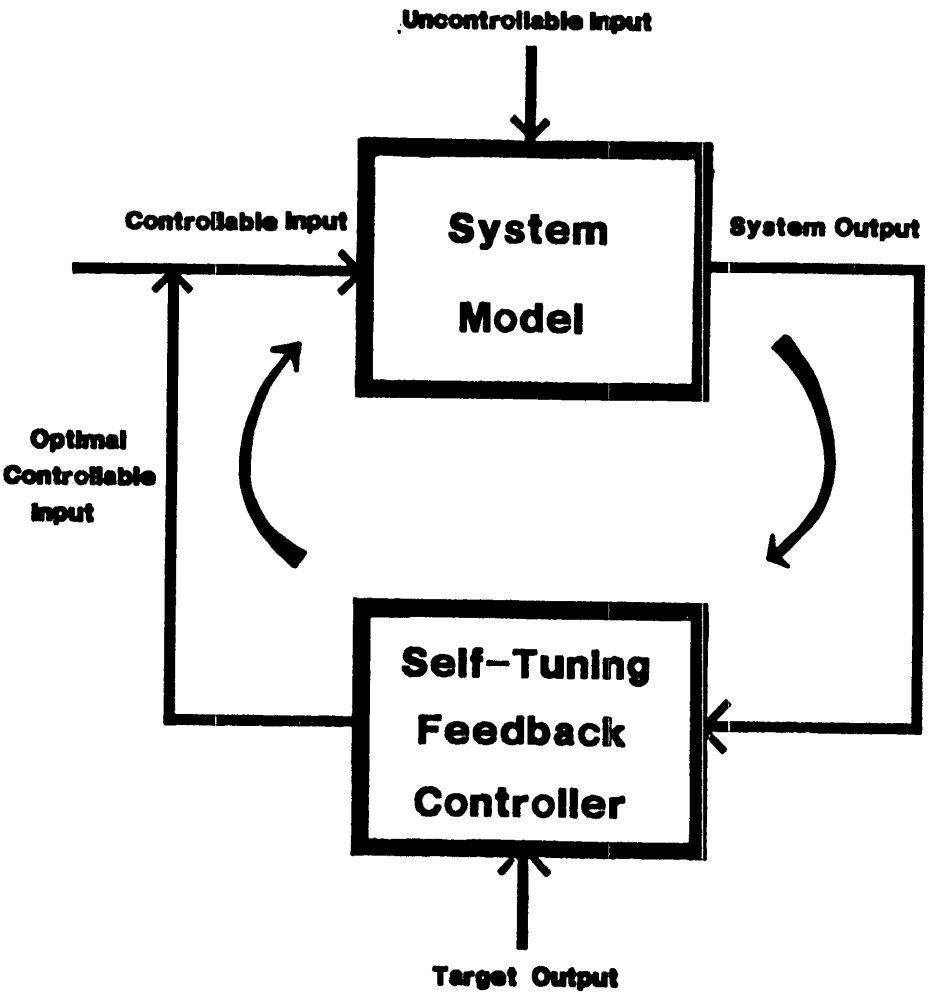

Figure 2(a) Self-Tuning Feedback Control System with Multiple Inputs.

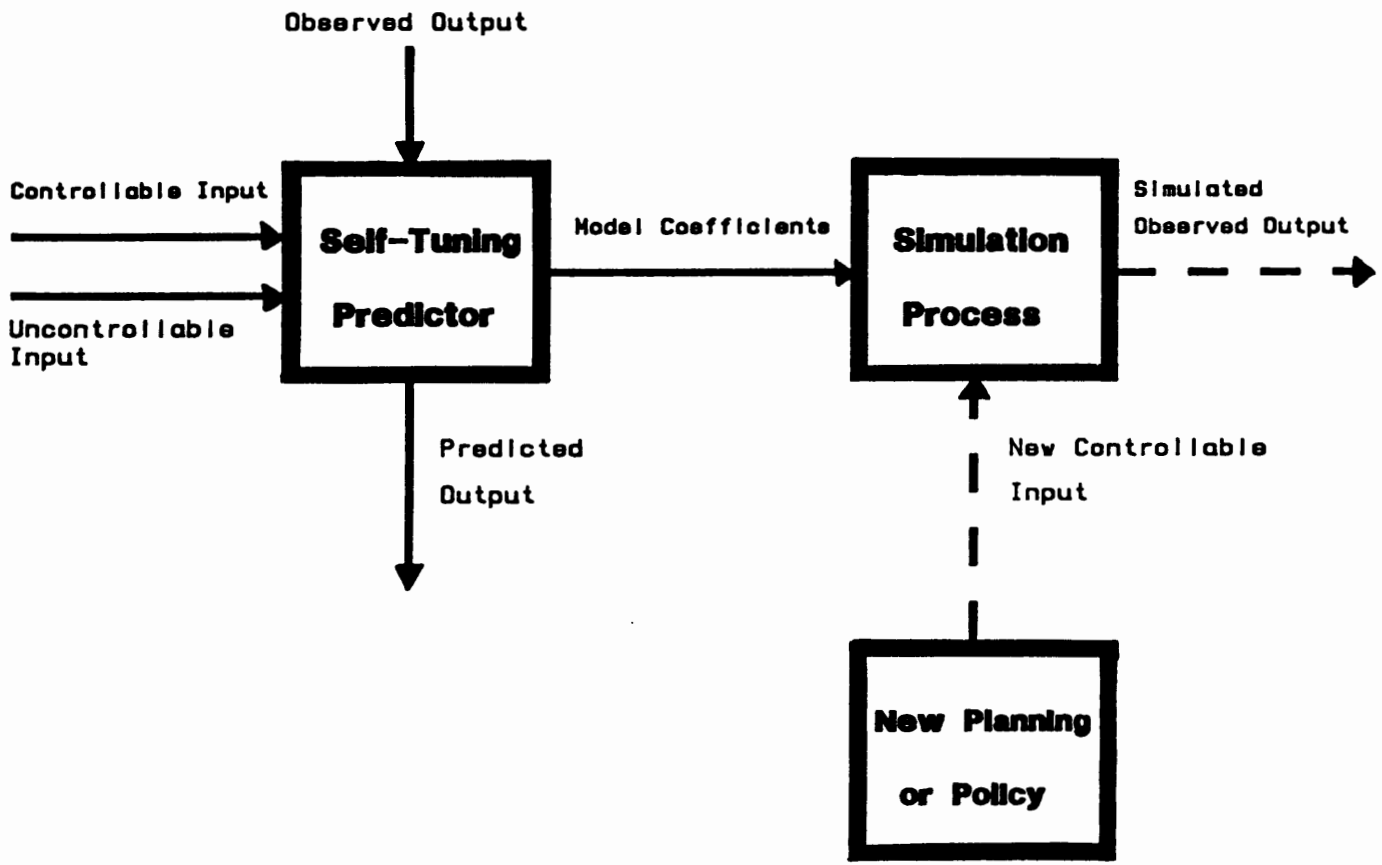

Figure 2(b) Simulation process with self-tuning predictor.

K_1 = pure time delay between riverflow and salinity.
K_c = pure time delay between tidal heights and salinity.
K_w = pure time delay between wind stress and salinity.

Using the above information and concepts of minimum square error predictor and minimum cost control, we obtain the control variable :

(2)

$$u(t) = -1/E(Gy(t)-Cw(t + k_1) + HFT_c(t-k_c+k_1) + MFWS(t-k_w+k_1))$$

where $E = BF + C(1-q^{-1}) = e_0 + e_1q^{-1} + \ldots + e_nq^{-n}$ (3)

$F = 1 + f_1q^{-1} + \ldots +_1 f_{1-k}\, q^{1-k_1}$ (4)

$G = g_0 + g_1q^{-1} + \ldots + g_{n-1}q^{1-n}$ (5)

To simplify the notation, data and parameter vectors are introduced:

(6)

$$\phi(t) = (u(t),\ldots,u(t-n),y(t),\ldots,y(t-n+1), w(t+k1\_1),\ldots, w(t+K_1-n) \ldots TC_(t+k_1-k_c),\ldots,T_C(t+k_1-k_c-n), WS(t+k_1-k_w) ,\ldots WS(t+K_1-K_w-n))$$

(7)

$$\theta^T(t) = (e_0,\ldots,e_n,g_0,\ldots,g_{n-1},-c_1,\ldots- c_n,J1_0, \ldots J1_n, J2_0,\ldots J2_n)$$

where J1=HF and J2=MF.

The final required riverflow is computed using the formula:

$$u(t) = -1/e_0(\phi(t)\theta(t) - e_0u(t) - w(t+k_1)) \quad (8)$$

The predicted output for the self-tuning predictor is computed using a similar procedure except that the minimum loss function is calculated by the predictor instead of calculating the cost function with the controller. The new parameter vector is presented as:

$$\phi(t) = [-\hat{y}(t+K_1-1),\ldots,-\hat{y}(t+K_1-n),u(t),..,u(t-n+1),T_C(t+K_1 -K_c) \ldots,TC_(t+K_1-K_c-n+1),WS(t+K_1-K_w),\ldots,WS(t+K_1-K_w-n+1), \varepsilon(t),\ldots,\varepsilon(t-n+1)] \quad (9)$$

where $\xi(t)$ = prediction error at time t.
$\hat{y}(t)$ = prediction estimate at time t.

The predicted output at time step $(t+K_1)$ is:

$$\hat{y}(t+K_1) = \phi(t)\,\theta(t) \qquad (10)$$

SIMULATION EXPERIMENTS AND APPLICATIONS

A salinity station in the Elk River area near Town Point, MD was selected as the study site. This location was chosen because of the ease in expressing transport mechanisms from several directions of forcing as well as the abundance of existing data. The flow station is located downstream from Conowingo Dam, Susquehanna River.

The subtidal height is calculated by removing the semi-diurnal and diurnal components from hourly tidal records at Hampton Road, Va. The wind data from Wilmington, Del. is converted to local cross-Bay wind stresses. The daily mean data set is obtained by subtracting several deterministic components from the hourly readings and taking the daily average. The model time step was selected as one day.

Estimation of coefficients for the self-tuning controller is made by supplying an initial coefficient vector and initial error covariance, pure delays, and the stabilizing factor. However, these values can only be rough estimates since this system is assumed to be unknown. The lower-order system is used for simplicity. The sum of square errors and minimum forgetting factor can be estimated by the degree of noise. Usually calibration is done by assuming that the desired salinity is equal to the observed salinity, then adjusting k_1, k_c, and k_w until the most optimal and stable condition is attained. The forgetting factor responds to the sensitivity of the estimated error for each time step. During the calibration process, a minimum value of forgetting factor is needed to manage the most dynamic condition. The required flow would nearly equal the measurement flow in an optimal selection of these system parameters. The final selection for this study is k_1, k_c, and k_w equal to 1, and equal to 0.20 (n=2). The self-tuning predictor is calibrated by using the sum of square errors as the single indicator.

Fishery Spawning and Drinking Water Concerns (Control Phase)

Data from the Upper Chesapeake Bay hydrographic survey (MD-DNR) were used to estimate the difference between Havre de Grace and Old Town Point when salinity information in the Havre de Grace area was not available. This difference can be used as an approximate basis for the simulation. Under this

prior consideration, eight sets of simulation tests were conducted. The target assumptions were: (1) Target values of salinity in the Elk River were selected as 0.1 and 2.0 ppt, during the spawning season; and (2) Target salinities in Elk River were 2.0 and 4.0 ppt during the low flow season. The second assumption will result in salinity levels of about 1.3 ppt and 2.5 ppt in the Havre de Grace area.

The difference between required flow and observed flow, is evaluated for excess or deficient conditions. The daily results of these simulation tests are averaged on a monthly basis (Table 1) to examine how current release policies, (5000 cfs minimum release from 15th of April to 15th of September) meet the needs for each month. The simulation shows that the current policy provides adequate water quality for striped bass spawning and larvae under the conditions during the springs of 1982 and 1983. However, greater flows are required during the fall season to meet drinking water standards, especially during October when the policy is not in effect.

Storage Reallocation Plans (Prediction Phase)

The prediction phase of the model evaluates the effects of storage reallocation at the lower Susquehanna River basin on the Upper Chesapeake Bay. Two plans at the mouth of the Susquehanna River's USGS station in Marietta are provided by the US Army Corps of Engineers (Baltimore District), both of which meet the flow target of 5000 cfs and 7400 cfs. The flow difference between each plan and the historical flow will be used to add or subtract from the flow at the USGS Conowingo Station. The new flow, considered as the new controllable input, simulates salinity levels produced by the reservoir's operation. The two uncontrollable inputs remain the same. The corresponding salinity difference (original and simulated) and flow difference (original and simulated) for each plan are shown in figures 3a-3d. The maximum difference occurs in October 1983 (0.4 ppt), when the mean salinity level is about 6.2 ppt. The effects of both storage reallocation plans are insignificant. However, more salinity stations are required to obtain more definitive conclusions.

CONCLUSIONS AND COMMENTS

(1) For a dynamic estuarine environment, a multivariate control system with variable forgetting factor (control and prediction phases) is capable of simulating sudden changes in input and output functions. This scheme eliminates non-linearity effects in a dynamic, noisy system.

Yr/Month	Target S.	Observed S.	Required F.	Observed F.	Diff. F.	% Change
1982 April	0.10	0.11	55337	55281	56.00	0.10
	2.00	0.11	50967	55281	-4313.78	-7.80
1982 May	0.10	0.84	25289	23931	1358.58	5.68
	2.00	0.84	22323	23931	-1607.69	-6.72
1982 June	0.10	0.83	54678	53159	1519.74	2.86
	2.00	0.83	50626	53159	-2532.97	-4.76
1982 Sep.	2.00	1.71	6444	6546	-101.88	-1.56
	4.00	1.71	5040	6546	-1505.57	-23.00
1982 Oct.	2.00	4.79	8255	6000	2255.58	37.59
	4.00	4.79	6666	6000	665.88	11.10
1982 Nov.	2.00	4.21	13211	11243	1968.13	17.51
	4.00	4.21	11313	11243	70.41	0.63

Yr/Month	Target S.	Observed S.	Required F.	Observed F.	Diff. F.	% Change
1983 April	0.10	0.53	37074	36265	808.51	2.23
	2.00	0.53	33697	36265	-2567.87	-7.08
1983 May	0.10	0.16	51451	51242	208.90	0.41
	2.00	0.16	47350	51242	-3891.36	-7.59
1983 June	0.10	0.23	33478	33248	229.99	0.69
	2.00	0.23	30348	33248	-2863.81	-8.61
1983 Sep.	2.00	2.53	5981	5368	621.71	11.41
	4.00	2.53	4644	5368	-724.77	-13.50
1983 Oct.	2.00	6.23	10740	7040	3664.13	52.05
	4.00	6.23	8820	7040	1780.06	25.28
1983 Nov.	2.00	3.39	23705	22681	1024.35	4.51
	4.00	3.39	21252	22681	-1428.84	-6.30

Table 1. Required riverflow from simulation experiments for multivariate control model when a given target salinity at Town Point is set.

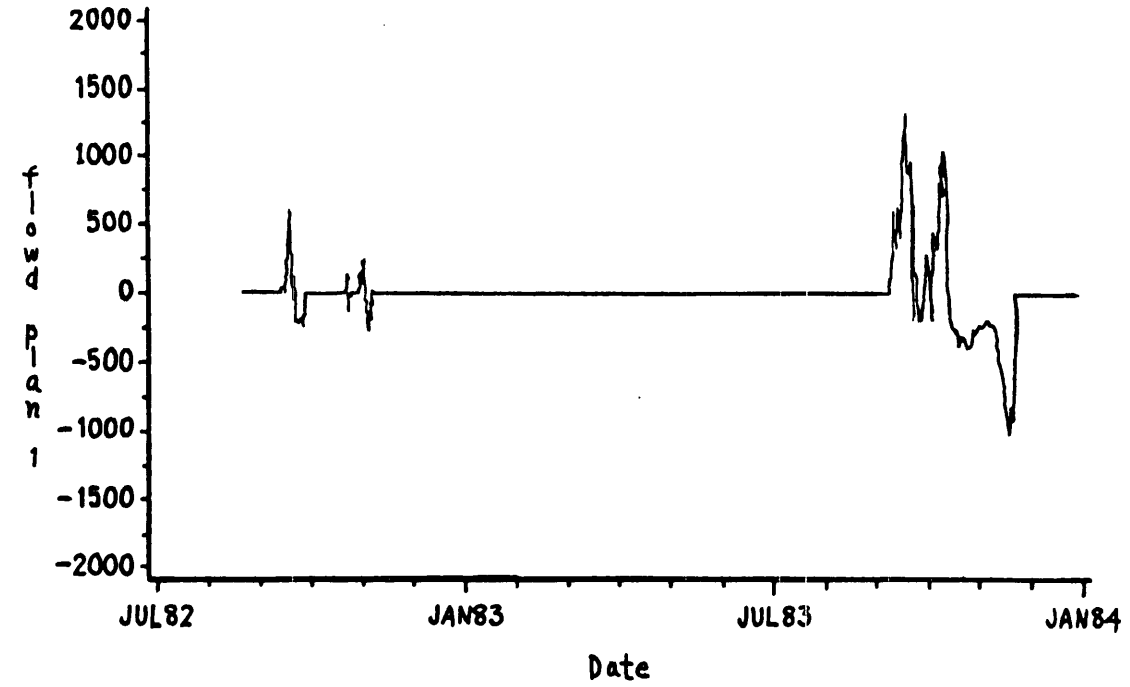

Figure 3(a) Flow change at Conowingo gauge (USGS) due to 31,000 acre-feet maximum storage plan(U.S. Army Crops of Engineers) at Mareitta gauge.

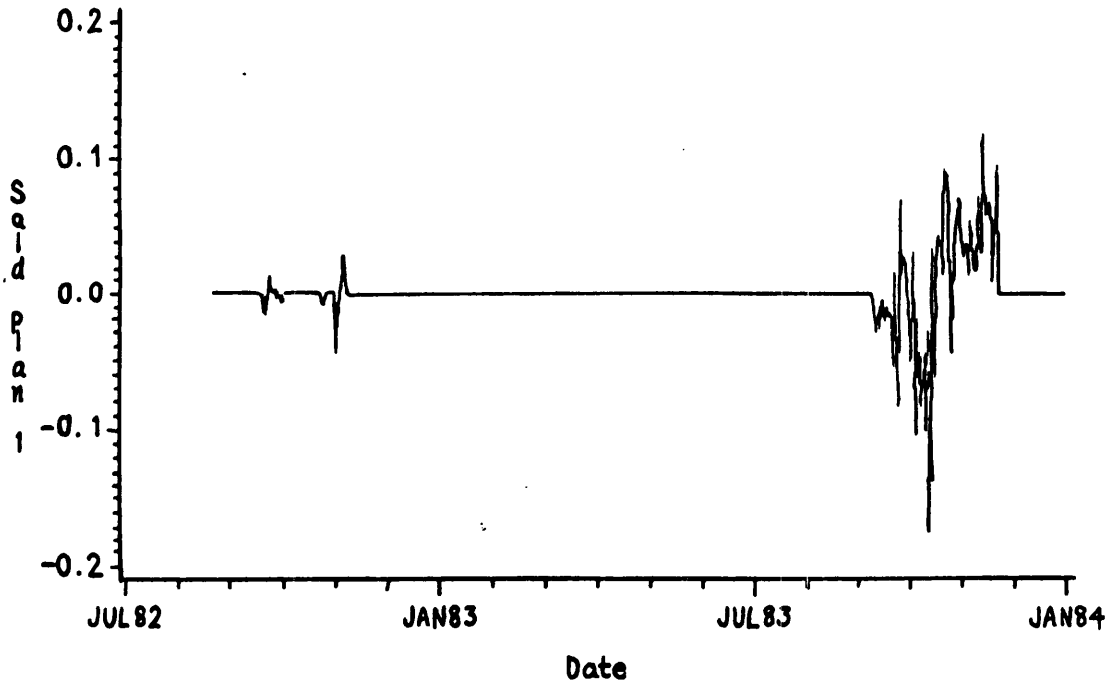

Figure 3(b) Simulated salinity change due to flow variation of 3(a).

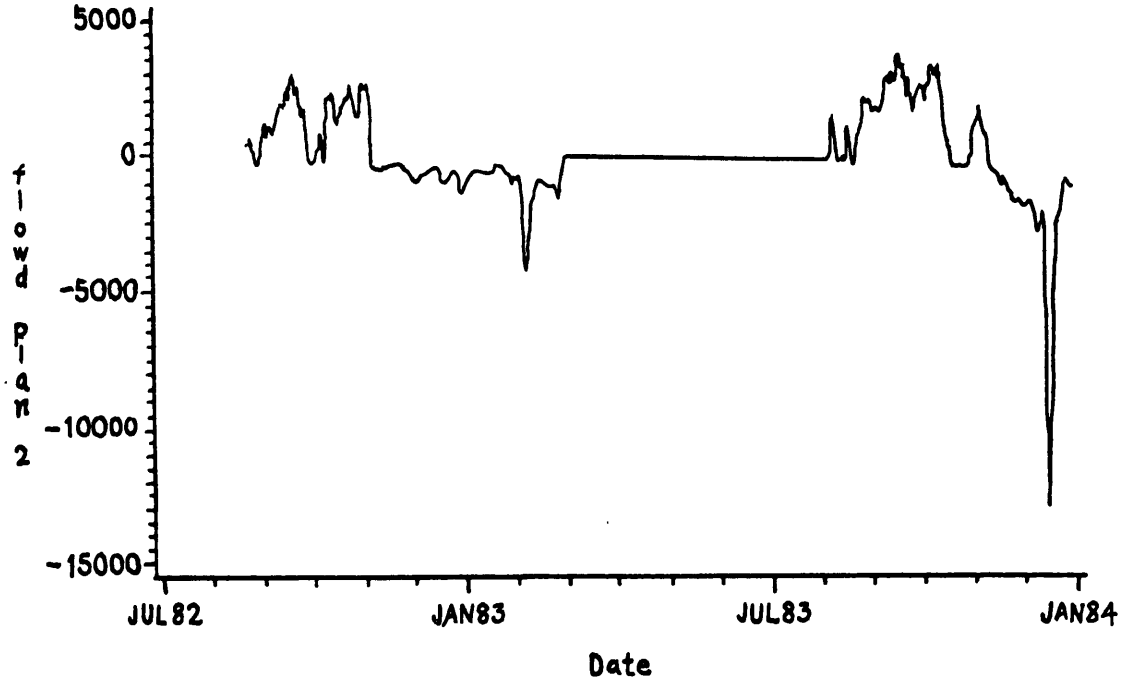

Figure 3(c) Flow change at Conowingo gauge (USGS) due to 280,000 acre-feet maximum storage plan(U.S. Army Crops of Engineers) at Mareitta gauge.

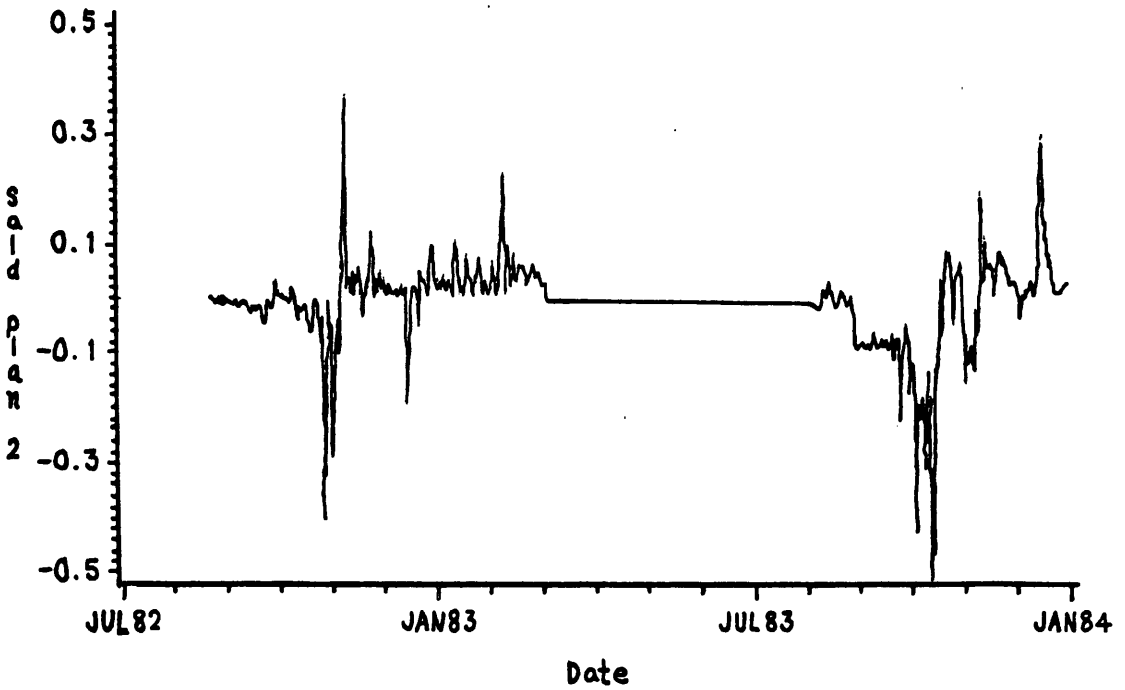

Figure 3(d) Simulated salinity change due to flow variation of 3(c).

(2) The self-tuning scheme is potentially useful for telemetry systems. This system can be linked with multi-objective reservoir operation systems to optimally utilize water resources.

(3) The pure delay between the output function and the control variable is the most critical factor affecting the stability of this system. Proper selection of the minimum forgetting factor is related to the system's stability when sudden changes in input variables occur.

FURTHER CONSIDERATIONS AND APPLICATIONS

(1) A regulation evaluation system is currently being conducted by coupling the control phase with the prediction phase. A reservoir management model using reliability programming methods can determine an optimal solution and the highest reliability for the entire system.

(2) Target stations other than Town Point are needed to verify the model and examine the Upper Bay variations before a final decision is made.

(3) With some modification, this model can be used for water quality and target flow control systems.

ACKNOWLEDGEMENTS

Support from the Water Resources Administration and the Tidewater Administration for this project are sincerely acknowledged.

REFERENCES

Astrom, K.J. Introduction to stochastic theory. Academic Press, N.Y. 1970.

Astrom, K.J.; Wittenmark, B. On self-tuning regulators. Automatica, Vol 9:185-199, 1973.

Bayoumi, M.M.; Wong, K.Y.; El-Bagoury, M.A. A self-tuning regulator for multivariable systems. Automatica 17: 575-592, 1981.

Borisson, U. Self-tuning regulators-industrial application and multivariable theory. Report 7513, Dept. of Automatic Control, 1975. Lund Inst. of Tech., Lund, Sweden

Borisson, U. Self-tuning regulators for a class of multivariable systems. Automatica 15: 209-215, 1979.

Clarke, D.W.; Gawthrop, P.J. Self-tuning controller. Proc, I.E.E., 1975. Vol 122, no.9:929-934.

Fortescue, T.R.; Kershenbaum, L.S.; Ydstie, B.E. Implementation of self-tuning regulators with variable forgetting factors. Automatica 17: 831-835, 1981.

Ganendra, T. A self-tuning predictor applied to river flow forecasting. Real-time forecasting/control of water resource system, 1976. Selected papers from an IIASA Workshop, Eric F. Wood, Editor.

Ganendra, T. A self-tuning controller applied to river regulation. Proceedings of modeling, identification and control in environment systems, 1978. Vansteenskists, Editor.

Hsieh, B.B. A system approach to analysis of water quality due to transport mechanisms in the water environment. Advances in water pollution control, 1987. Vol 3:109-118. Pergamon Journals Ltd., London.

Koivo, H.N. A multivariable self-tuning controller. Automatica 16: 351-366, 1980.

Understanding the Estuary: Advances in Chesapeake Bay Research. Proceedings of a Conference. 29-31 March 1988. Baltimore, Maryland. Chesapeake Research Consortium Publication 129. CBP/TRS 24/88.

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

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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

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

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

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

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.

BIBLIOGRAPHY

Wastler,T.A. Spectral analysis: applications in water pollution control. Washington,DC: U.S. Department of the Interior; 1969.

Wastler,T.A.; Walter,C.M. Statistical approach to estuarine behavior. J. of the Sanitary Engineering Division, ASCE. 94: No.SA 6, Proc. Paper 6311, 1175-1194; 1968.

Understanding the Estuary: Advances in Chesapeake Bay Research. Proceedings of a Conference. 29-31 March 1988. Baltimore, Maryland. Chesapeake Research Consortium Publication 129. CBP/TRS 24/88.

Application of a Multiple Linear System to Identify Tidal Signals in the Upper Chesapeake Bay

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ABSTRACT

The Upper Chesapeake Bay has a dynamic environment of physical characteristics due in part to incoming tidal fluctuations from the Delaware Bay via the C & D Canal, and the mouth of the Chesapeake Bay. The tidal propagation signals from the two bay systems are a function of most of the Upper Bay area because of the difference in distance traveled from the mouth. The high volume flow from the Susquehanna River might result in some water level change as well. Although local effects are also important factors, this study focuses on non-local sources of impact.

In order to understand energy transport due to tidal forcing and combined river runoff, it is useful to utilize a multiple transfer function model which can be used to indicate the dominant forcing factors and discern the tidal direction for specified areas. Three important frequency bands--semi-diurnal, diurnal, and 3-20 days, are used to perform this investigation from hourly sea level readings. The model's structure is defined by river flow and tidal signals from both bays as inputs, and one Upper Bay location as an output.

INTRODUCTION

Studying tidal propagation phenomena is probably the most revealing means of understanding the dynamic behavior of water movement in the Chesapeake Bay. The Upper Chesapeake Bay receives tidal fluctuation signals from the Delaware Bay through the C & D Canal, and from

the mouth of the Chesapeake Bay. Dronkers (1972) indicated that the propagation of a tide is modified by its degree of penetration into the continental shelves, coastal areas, and estuaries. The distortion of the tidal wave takes place at this point. This change is primarily caused by : the friction of the bottom, changes in tidal velocity due to variation in depth, and the shape of the estuary. Tidal prediction is primarily employed at harbors situated at coasts where tidal records include various impacts resulting from meteorological effects. Tidal prediction is difficult due to variation in the amount of river runoff and additional disturbances which may change the water level considerably. Abnormal weather changes such as tsunami may generate high tidal waves or intensive rainfall, which result in great amounts of river runoff. This directly affects the properties along the coast. Rather than focusing on single-point tidal prediction, this study uses non-local effects such as tidal signals from the two bay systems and the Susquehanna River flow, as the basic mechanisms for determining the sea-level change for several selected locations Reedy Point, which is located at the east end of the C & D Canal, is an interesting area for researchers because of the possibility of recording any signals from the Chesapeake Bay which are transported there. Havre de Grace, near the mouth of the Susquehanna River, and the Baltimore Harbor, the most important commercial port in the Bay, are included in this study as well. Two significant questions are addressed in this investigation : (i) Which bay contributes more tidal signals to the Upper Bay area ? (ii) Do any river runoff signals travel from the Susquehanna solely to the Chesapeake Bay ?

This approach is characteristic of multiple inputs/single output systems. The multiple transfer function modeling technique, which is based on the frequency domain process, produces multiple frequency response functions. These functions use a Fourier transformation which drives this system from time-domain to frequency-domain, and simultaneously estimates their partial response factor and calculates the multiple coherence. A recent study (Hsieh 1985), showed that the partial multiple coherence is a powerful technique to use for separating the relative magnitude of each factor. This technique, due to linear relationships in the system, represents only a fraction of output power at any given frequency. The difficulties in computing the inverses of complex matrices when a multiple linear system is applied, are successfully solved by the Gaussian elimination method and the partitioning approach.

TIDAL CONSTITUENTS FOR SELECTED STATIONS

Data containing hourly observations of sea-level elevation for 1983 were obtained from five stations of the National Ocean Service. Each time series, containing 8,760 records, was linearly detrended prior to cyclical analysis. Every tidal record indicates a very weak linear trend. The amplitude and phase angle can be computed for any desired period by conducting harmonic analysis. However, this technique fails to show the importance of phenomena for which the period does not exactly coincide with a specified harmonic. One method to reduce this disadvantage is to use very small frequency bands in the analysis.

Budgell (1981) stated that the main harmonic component in the tide will tend to be semi-diurnal due to the two bulges created by tidal

forces. The main lunar and solar semi-diurnal constituents that correspond with this phenomena are the tidal components M2 and S2. When the tide enters the area over a continental shelf and penetrates into an estuary, its propagation is effected by the reduced depth. In mathematical terminology, this means that non-linear terms in the hydrodynamic equation will produce new harmonics which are a combination of the two original frequencies. This study will examine the tidal constituents for four categories: (1) semi-diurnal components, (2) diurnal components, (3) shallow water components, and (4) 3-20 days components. This method of classification covers the tidal period from two hours (M12) to about 350 hours (Msf).

An investigation was made by removing M2 tides from each tidal record. A subtidal fluctuation typical of Reedy Point, Delaware, is shown in Figure 1. The rest of the stations reveal less subtidal fluctuations (Hsieh 1985). One useful technique in evaluating the fluctuation of the tide is by using a low pass filter to show its longer term variations. The longer term fluctuations for five tidal stations are shown in Figure 2. The water level at Havre de Grace is the highest while the lowest is found at Reedy Point. The trend component for each station is small enough to disregard. The freshwater inflow at Conowingo Dam accounts for approximately 50 percent of the total input of freshwater entering the Bay, with more than 85 percent of the freshwater entering the Bay above the mouth of the Potomac (Schubel 1972). In 1983, the hourly range was between 200 cfs and 90,000 cfs. Figure 2 suggests a strong seasonality in freshwater inflow and approximates an eight-months cycle.

To determine the basic structure of each of the tidal records, harmonic analysis of the above-mentioned categories is used. None of the significant components for shallow water constituents are found in the frequency band between a 2-hour and 6-hour period. Significant tidal components for semi-diurnal and diurnal periods are shown in Table 1. The M2 tide accounted for more than 75 percent of total variance at Reedy Point, but only about 20 percent of the total variance at the Baltimore station. A low percentage for the M2 tide was also found at Havre de Grace. The low percentage and small amplitude of the M2 tide could be attributed to its loss of tidal wave energy from the Chesapeake Bay entrance, due to the long distance traveled and the complex geometry of the Bay. Therefore, the local effects and low variation of the tide in the Bay's inner area are important factors to consider. However, the 60 percent of total variance at Baltimore results from longer than two days fluctuations. The periodgram reveals that most of the Baltimore signals have irregular variability. The station at Havre de Grace might receive some tidal signals from the Delaware Bay due to the effect of standing waves at the end of the estuary. The S2 component is generally more significant than the N2 component, but this phenomenon does not apply to these Upper Bay stations. In inner Bay stations such as Havre de Grace and Baltimore, the S2 component is even less than the K1 and O1 components.

FREQUENCY MULTIPLE TRANSFER FUNCTION MODEL

In order to determine the impulse response functions from the x(t) and y(t) data in the frequency domain, it is necessary to transform

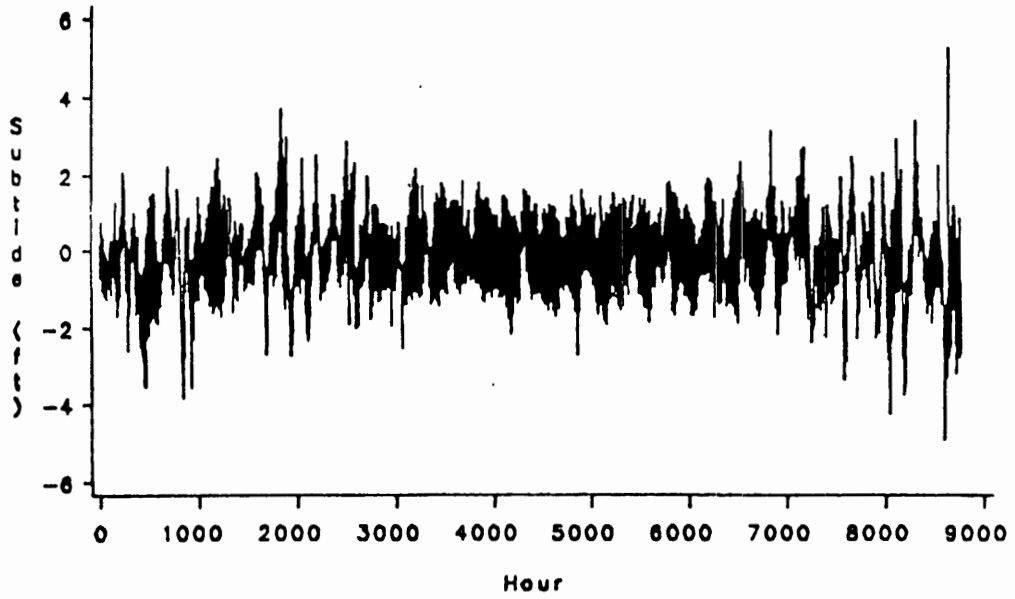
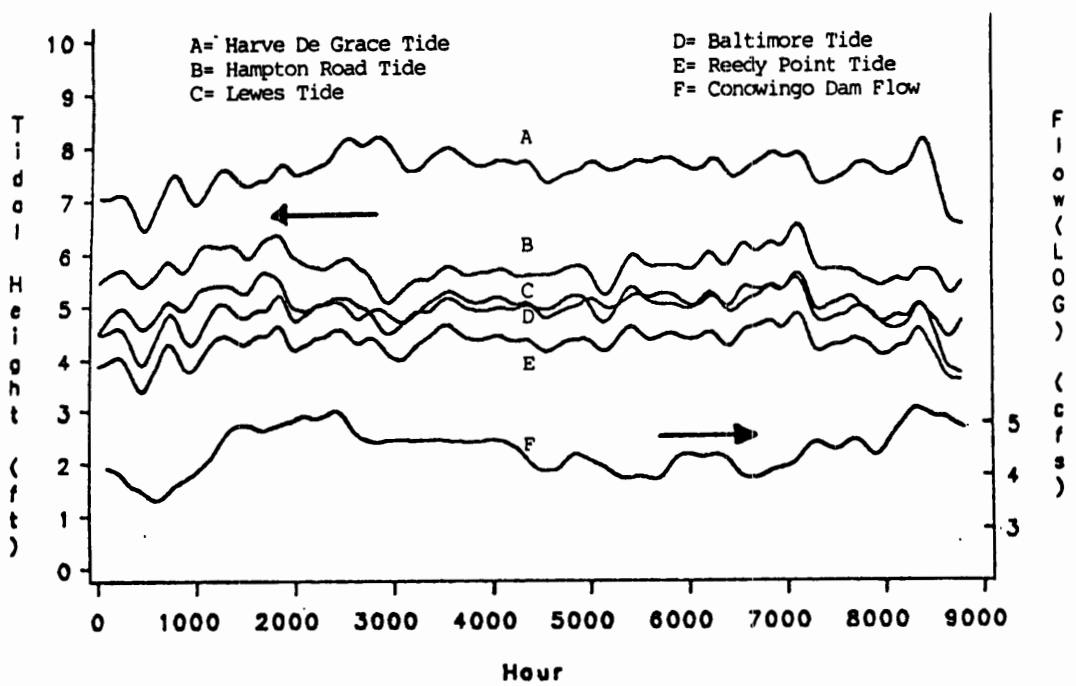


Figure 1. Hourly subtidal height at Reedy Point, Del.



Fighre 2. Low frequency variations of tidal hight and freshwater inflow for selected Upper Bay stations.

	Reedy Pt.		Havre De Grace		Lewes		Hampton Road		Baltimore	
Period	Amp.	Total Var.(%)	Amp.	Total Var.(%)	Amp.	Total Var.(%)	Amp.	Total Var.(%)	Amp.	Total Var.(%)
K2 11.97	0.17	0.37	0.06	0.18	0.16	0.51	0.09	0.40	0.04	0.13
S2 12.00	0.32	1.36	0.14	0.98	0.34	2.36	0.20	1.86	0.07	0.41
L2 12.19	0.20	0.53	0.06	0.19	0.05	0.06	0.05	0.11	0.02	0.03
M2 12.42	2.48	79.15	0.85	35.44	1.95	75.12	1.17	61.97	0.49	17.82
N2 12.66	0.50	3.18	0.18	1.57	0.48	4.51	0.29	3.87	0.12	1.08
ŊJ2 12.87	0.09	0.11	0.04	0.07	0.04	0.03	0.03	0.05	0.01	0.01
K1 23.93	0.29	1.05	0.28	3.96	0.34	2.22	0.17	1.33	0.23	3.86
P1 24.07	0.08	0.09	0.07	0.22	0.11	0.22	0.05	0.14	0.07	0.37
O1 25.82	0.22	0.65	0.19	1.83	0.28	1.51	0.14	0.92	0.18	2.53
Q1 26.87	0.03	0.01	0.03	0.06	0.05	0.06	0.03	0.05	0.03	0.11

Table 1. Significant tidal components of the Upper Chesapeake Bay for selected stations.

the input and output functions from time-domain into frequency-domain. The initial solution is obtained by using formulas 1-12 (Huthman 1978; Enochson 1968).

$$Y(t) = \int_{-\infty}^{\infty} h(\tau)x(t-\tau)\,d\tau + e(t) \tag{1}$$

disregarding e(t) we obtain :

$$X(f) = \int_{-\infty}^{\infty} x(t)\ e^{-j2\pi ft}\,dt \tag{2}$$

$$Y(f) = \int_{-\infty}^{\infty} y(t)\ e^{-j2\pi ft}\,dt \tag{3}$$

and :

$$H(f) = \int_{-\infty}^{\infty} h(t)\ e^{-j2\pi ft}\,dt \tag{4}$$

the Fourier transform H(f) of the impulse response function h(t) is called the frequency response function, which is :

$$H(f) = Y(f)/X(f) \tag{5}$$

A model of a linear system responding to multiple inputs is developed by expanding the above relationship. (Figure 3)

$$Y(f) = \sum_{I=1}^{N} Y1(f) = \sum_{1=1}^{N} H1y(f)X1(f) \tag{6}$$

This relationship can be expressed more concisely in matrix notation. Refine a N-Dimensional frequency response function vector to obtain :

$$H(f) = (H1y(f),\ H2y(f).....HNy(f))^{T} \tag{7}$$

Next, define a N-dimensional cross-spectrum vector of the output y(t) with the input X(t) :

$$Gxy(f) = (G1y(f), Gzy(f), ..., GNy(f))^{T} \tag{8}$$

Finally, define the N x N matrix of the power and cross spectra of all the inputs X(t) by :

$$Gxx(f) = \begin{pmatrix} G11(f) & G12(f) & .. & G1N(f) \\ G21(f) & . & .. & . \\ . & . & .. & . \\ GN1(f) & GN2(f) & .. & GNN(f) \end{pmatrix} \tag{9}$$

Use the system of linear equations to obtain the least squares solution for equation (6), which provides the matrix equation :

$$Gxy(f) = Gxx(f)\ .H(f)$$

The final solution to this calculation is :

$$H(f) = Gxx^{-1}(f)\ Gxy(f) \tag{11}$$

The linear relationship of the multiple system can be expressed as the multiple coherence function estimates for the discrete frequencies by the equation :

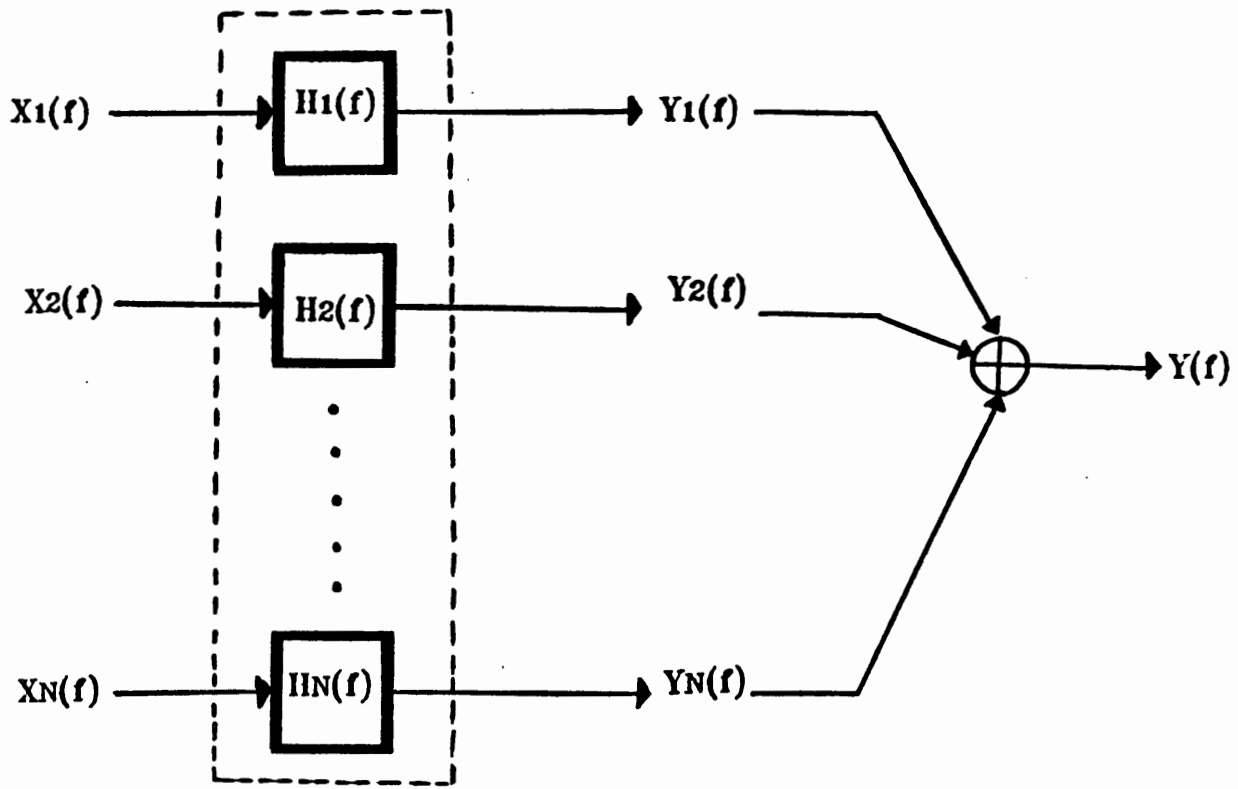


Figure 3. Multiple linear system with multiple inputs and single output.

$$Kyx^2(f) = (H1(f)*Gy1(f) + \ldots + HN(f)*GyN(f)/Gyy(f) \quad (12)$$

where : $Gyx(f) = (Gy1(f), \ldots, GyN(f))^T$
$Gyy(f)$ = output power spectrum

In order to separate the relative importance for each input on output for a particular frequency band, use equation (12) to obtain the partial multiple coherence.

$$Kyx^2(f) = HG1y + \ldots + HGNy \quad (13)$$

where : $HG_{1y} = H1(f) *Gy1(f)/Gyy(f)$

define : $TGH = HG1y^2 + \ldots + HGNy^2 \quad (14)$

to attain the partial multiple coherence for input 1 :

$$Kyx1(f) = HG1y^2 *Kyx^2(f)/THG \quad (15)$$

TIDAL SIGNALS MODEL

The multiple transfer function model, based on a frequency domain, can be applied to the tidal signals model. This model assumes that sea-level changes at selected Upper Bay locations are multiple functions of tidal propagation from Lewes, Delaware, Hampton Road, Va., and runoff from the Susquehanna River. This linear system is solved by using the Gaussian elimination method and the partitioning technique to compute the inverse of matrix Gxx(f). Since the cross-spectrum consists of a real part and an imaginary part, the condition number of matrix Gxx(f) is an important factor in determining whether or not the solution is accurate.

The Reedy Point station is selected as an example for demonstrating this approach. The multiple response function is obtained by using detrended tidal records and non-seasonal Susquehanna River flow. Detrended tidal records are used instead of subtidal height fluctuations because the greatest variation of tide is derived from the semi-diurnal and diurnal components. The final partial multiple coherence is able to demonstrate the dominant signal for each particular frequency band. The frequency band which accounts for larger variance is more significant than the others. For example, if the partial multiple coherence of M2 tide for the Delaware signal at Reedy Point has a greater variance, then the Delaware Bay contributes larger M2 fluctuations to the Upper Bay. Huthmann (1978) indicates that the multiple coherence function will be unity over the entire frequency range under ideal, noise-free conditions in which there is a true linear relationship in the multiple input/single output system. The multiple coherence function is less than unity in a natural system. In this study, two data files are constructed. One file combines the hourly tidal height with the subflow data by taking the log function and removing the 8-month significant cyclic component. The second file is constructed by daily means from the first data file. The multiple coherence over semi-diurnal components at Reedy Point is close to unity (Figure 4). Since the noises occur in part of the frequency range, the smoothing curve is obtained by using a

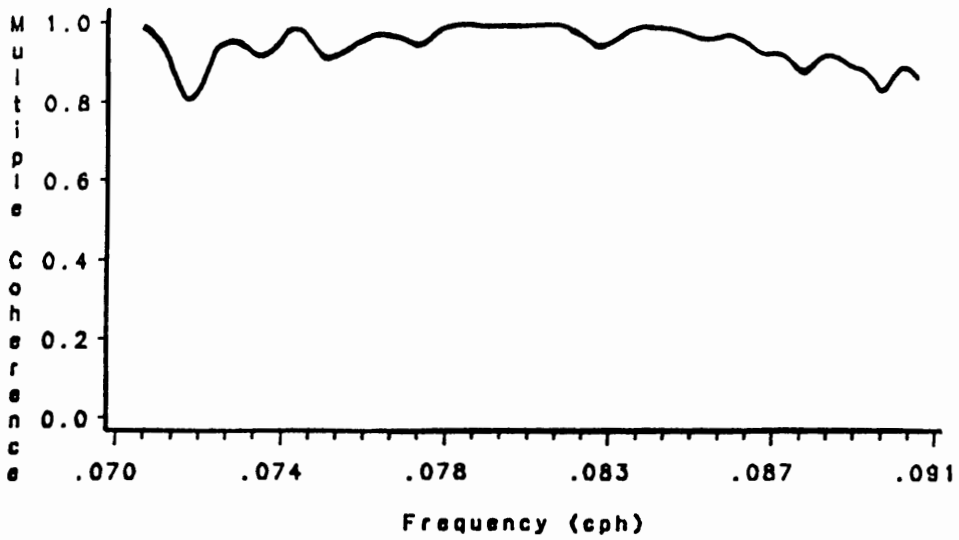


Figure 4. Multiple coherence over semi-diurnal components at Reedy Point, Delaware.

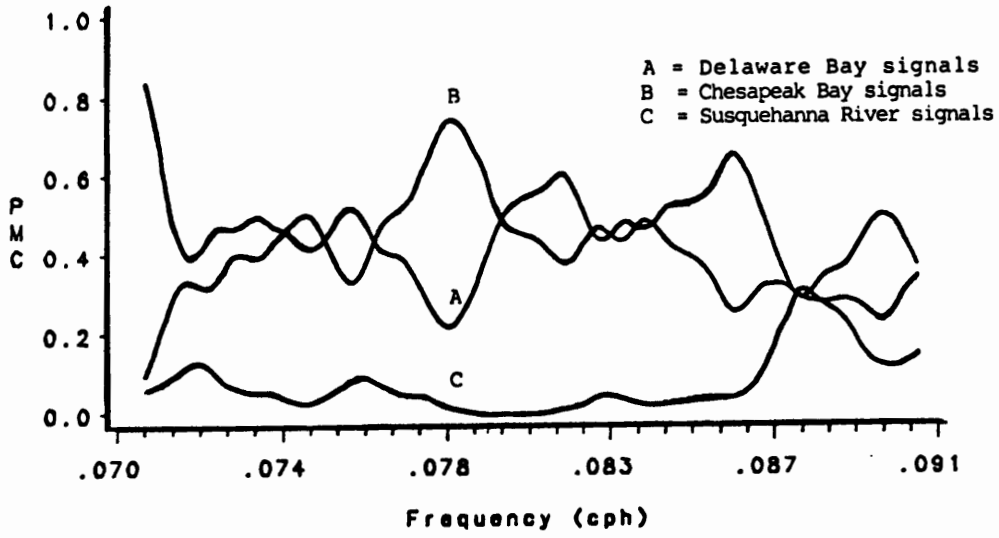


Figure 5(a). Partial multiple coherence for semi-diurnal tidal components at Reedy Point, Delaware.

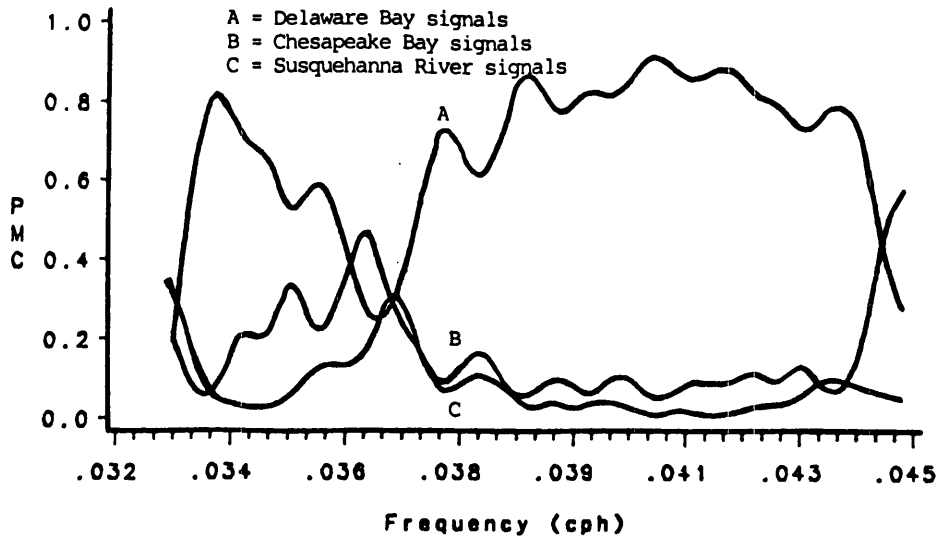


Figure 5(b). Partial multiple coherence for diurnal tidal components at Reedy Point, Del.

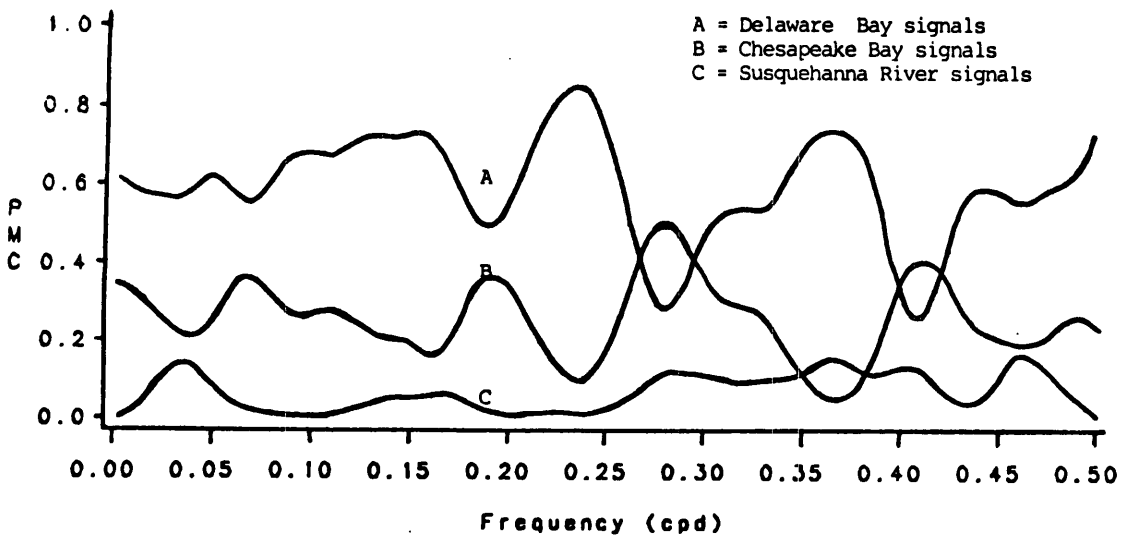


Figure 5(c). Partial multiple coherence for daily tidal components at Reedy Point, Del.

30-frequency band moving average method (for a total of 200 frequency bands). The partial multiple coherence in Figures 5a-5c separates the signals from the Delaware Bay, Chesapeake Bay and Susquehanna River flows. The Susquehanna River signal is very weak compared to the other two signals. The most significant peak in Figure 5a indicates that the N2 tide (12.66 hr) from the Chesapeake Bay is more significant than the Delaware signals. The frequency band is so close that certain peaks are smoothed by insignificant neighboring frequency bands. Therefore, the wider frequency range of the dominating source is selected (Figure 5b). For diurnal components, the Reedy Point sea-level is mainly dominated by the Delaware Bay signal except for fluctuations around 27.5 hours, which are provided by the Susquehanna River signal. For the daily mean set (Figure 5c), the Delaware signal is the major source, but two frequency bands (3 days and 2.4 days) are shared with the Chesapeake Bay signals.

DISCUSSION

The Havre de Grace and Baltimore Harbor stations receive the same inputs. Data from these stations can therefore be used to compare the variation of signals at different frequency bands due to the change in location. Several important findings are summarized below.

Semi-Diurnal Components

(i) Baltimore Harbor receives even higher tidal signals from the Delaware Bay than does Havre de Grace. The Baltimore Harbor area also has stronger K2 & M2 tidal components than the Reedy Point station.
(ii) Each station has its own higher constituents from the Chesapeake Bay signals : Reedy Point has a higher M2, Baltimore receives a stronger M2, and Havre de Grace contains a higher L2 tidal component.

Diurnal Components

(i) The Baltimore Harbor area and the Havre de Grace station show similarity in their patterns resulting from the Delaware Bay signals. Reedy Point is the highest over the entire frequency range, but the Baltimore station has a stronger K1 tide.
(ii) The Chesapeake Bay contributes very few diurnal components to Reedy Point. Baltimore receives stronger signals except at the P1 tidal frequency band.
(iii) Havre de Grace receives stronger signals at 27.78 hours from the Susquehanna River flow.

Daily Mean Component

(i) Baltimore and Havre de Grace receive similar tidal components from the Delaware Bay signal propagation. Peaks are found at 2.5 days, 4.5 days, and 3 weeks.
(ii) Again, the Chesapeake Bay contributes the same slow variation between Baltimore and Havre de Grace, but carries only a 3.5-day signal to the Reedy Point station.
(iii) The Susquehanna River flow contributes about 3-day fluctua-

tions both at Baltimore and Havre de Grace, but no significant signals are found at Reedy Point.

It would be interesting to know if significant tidal components would respond to these factors more precisely if the frequency band was specified. The partial multiple coherence for three desired categories is shown in Table 2. These values provide a relative index to determine which signals contribute to a major tidal component. The Susquehanna River responses appear to be a minor factor for every important tidal component except for the 29-day period. This phenomenon is also valid for Havre de Grace and Baltimore. A very significant finding is that the Delaware Bay signals have the greatest percentage of slow variations (Table 2). The Delaware Bay signals also show their significance in the Q1 and K1 components and the Chesapeake signals dominate L2 tidal constituents. The partial multiple coherence and amplitude (total variance) are two major factors used to calculate the amount of variation attributable to individual signals.

CONCLUSIONS

The partial multiple coherence technique, generated by taking fractions of multiple coherence, is a very powerful tool to use in describing the multiple linear system of tidal propagation signals from the two bay systems. The Susquehanna River flow plays a minor role in this case, except at 29-day period fluctuations. The Delaware Bay signals dominate the slow variation of tidal behavior. The Baltimore Harbor and Havre de Grace areas show the same pattern signals over slow tidal fluctuations. For diurnal components, the Delaware Bay signals are the controllers of the K1 and Q1 components. The O2 tidal signals from the Delaware Bay are not found in the Chesapeake Bay. The Chesapeake Bay transports a large number of semi-diurnal signals to the Delaware Bay. However, the Havre de Grace and Baltimore Harbor stations detected several significant tidal signals from the Delaware Bay. This phenomenon suggests that both bays distribute their own signals for high frequency tidal transport.

	Reedy Point			Havre De Grace			Baltimore		
period	Del.	Che.	Sus.	Del.	Che.	Sus.	Del.	Che.	Sus.
K2 11.97	0.181	0.817	0.000	0.678	0.316	0.000	0.374	0.258	0.056
S2 12.00	0.200	0.796	0.003	0.294	0.698	0.007	0.591	0.409	0.000
L2 12.19	0.364	0.622	0.011	0.339	0.620	0.001	0.255	0.719	0.007
M2 12.42	0.757	0.242	0.005	0.314	0.683	0.000	0.744	0.256	0.000
N2 12.66	0.274	0.724	0.003	0.515	0.478	0.000	0.294	0.704	0.000
μ2 12.87	0.059	0.812	0.026	0.215	0.646	0.009	0.863	0.087	0.002
K1 23.93	0.849	0.149	0.001	0.718	0.279	0.001	0.961	0.082	0.002
P1 24.07	0.903	0.059	0.014	0.086	0.870	0.025	0.751	0.210	0.017
O1 25.82	0.944	0.054	0.001	0.234	0.746	0.018	0.186	0.799	0.012
Q1 26.87	0.878	0.013	0.018	0.854	0.053	0.037	0.860	0.098	0.002
6 days	0.681	0.090	0.103	0.421	0.192	0.096	0.432	0.224	0.102
9 days	0.346	0.542	0.003	0.280	0.492	0.000	0.300	0.445	0.002
13 days	0.693	0.263	0.008	0.653	0.293	0.018	0.652	0.283	0.014
18 days	0.811	0.166	0.005	0.693	0.261	0.006	0.701	0.260	0.001
21 days	0.865	0.051	0.016	0.751	0.124	0.033	0.787	0.085	0.022
25 days	0.456	0.200	0.210	0.662	0.151	0.053	0.707	0.083	0.054
29 days	0.139	0.324	0.521	0.164	0.145	0.665	0.211	0.172	0.584
48 days	0.650	0.175	0.030	0.598	0.116	0.007	0.502	0.234	0.057

** Del. = Delaware Bay Signals Che. = Chesapeake Bay Signals Sus. = Susquehanna River Signals

Table 2. Partial multiple coherence for major tidal components.

REFERENCES

Budgell, W.P. A stochastic-deterministic model for estimating tides in branched estuaries. Manuscript Report Series, No. 10, Ocean Science and Surveys, Fisheries and Oceans, Canada 1981. Burlington, Ontario. 189 p.

Dronkers, J.J. Tidal theory and computations. Advances in Hydroscience. 10:145-226; Academic Press, Inc. 1972.

Enochoson, L.D.; Otnes, R. Programming and analysis for digital time series data. Frequency response and coherence function computations. 1968. The Shock and Vibration Information Center, Naval Research Laboratory. Washington, D.C. p. 185-210.

Huthmann, G.; Wilke, K. Time and frequency domain of water quality systems, Modeling, identification and control in environmental systems. 1978. Amsterdam : North-Holland Publishing Company; 557-583. a

Hsieh, B. Development of multiple coherence via linear system and Gaussian elimination method. November 1985. Technical Notes, MD. Dept. of Natural Resources, Annapolis.

Schubel, J.R. The physical and chemical conditions of Chesapeake Bay an evaluation. January 1972. Chesapeake Bay Institute Ref. 72-1.

Understanding the Estuary: Advances in Chesapeake Bay Research. Proceedings of a Conference. 29-31 March 1988. Baltimore, Maryland. Chesapeake Research Consortium Publication 129. CBP/TRS 24/88.

Time Variations of Bottom-Water Inflow at the Mouth of an Estuary

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ABSTRACT

Puget Sound is a fjord-like estuary, but its 30-km long entrance sill, Admiralty Inlet, has characteristics very similar to coastal plain estuaries. The replacement of bottom water in Puget Sound has been studied for many years, because it is a dominant process responsible for flushing some contaminants. Previous studies showed bottom-water inflow increased during neap tides when mixing was minimal over the entrance sill. Recent observations show the increased inflow starts before minimum neap tides, and simple model calculations with these data demonstrate this is an effect of variations in the horizontal density gradient at the mouth of the estuary caused by salinity variations outside the mouth. This time-dependent process may be responsible for changing inflow characteristics at time scales between wind effects and seasonal effects, and it may be important in other estuaries such as Chesapeake Bay.

INTRODUCTION

Puget Sound is the southernmost glacial carved estuary in western North America and is surrounded by major urban centers. The entrance sill to this estuary is topographically complex (Figure 1) and plays a major role in regulating the replacement of water inside the estuary below the sill. This process is important for removal of some contaminants. Salinity (density) at sill depth outside the estuary is always greater than inside at or below the sill, but water from outside does not flow continuously into the deeper water. Over the sill, the flow is two layered, and salinity is horizontally stratified, closely resembling coastal plain estuaries. Previous studies showed bottom-water inflow events during neap tides when mixing is least over the entrance sill (Geyer and Cannon 1982). Other studies, however, had implied that inflow also might occur on very large spring tides when the tidal excursion could transit the sill on a single flood tide. This paper describes new observations to resolve this discrepancy and to determine whether the onset of the intrusions could be predicted. Other recent observations have shown this circulation

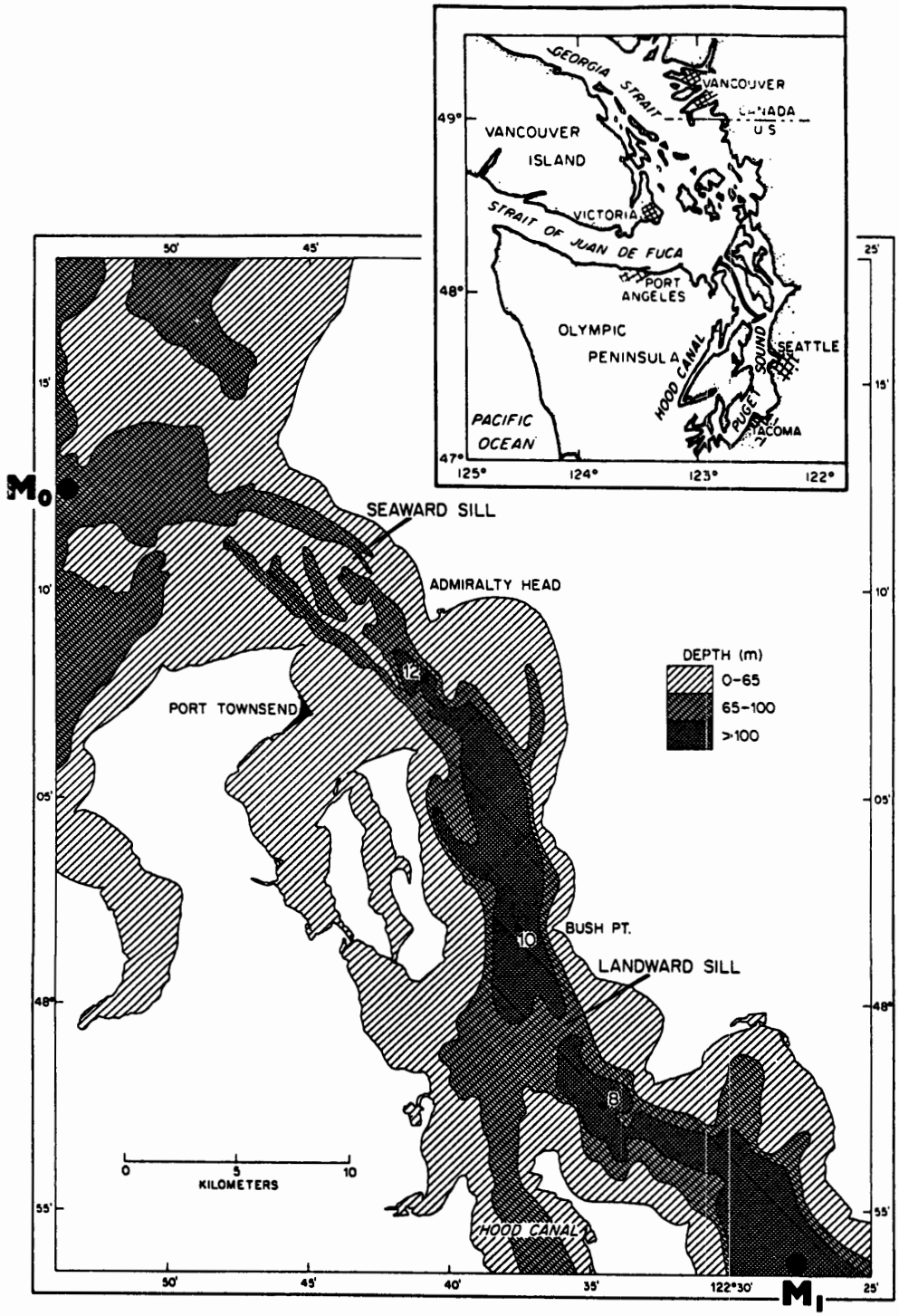

Figure 1. Topography of the Admiralty Inlet entrance sill connecting Puget Sound with the Strait of Juan de Fuca showing locations of moored instruments outside and inside the sill. Inset shows track across the sill connecting the moorings.

process is exceeded only by wind effects in the amount of non-tidal energy (Bretschneider *et al.* 1985).

OBSERVATIONS

Currents and salinity of the deep water just inside the entrance sill both show pronounced increases at about fortnightly intervals which is the dominant characteristic of inflow of new bottom water (Figure 2). The onset of these intrusions occurred in all cases before minimum neap tidal currents, and the maximum salinity occurred at about this minimum. Inflow during the largest intrusions is sufficiently large that the tidal currents do not reverse at the bottom. Spring tides all occurred during decreasing salinity in the basin. Thus, it appears clear that major inflows occur only during neap tides, but the onset occurs before the minimum neaps. Outside the sill, salinity variations are larger than inside the estuary, and they are not in phase.

Because of the relatively large variation in outside salinity and because the increase in inside salinity occurred before neap tides, it was felt that variations in the pressure gradient across the sill might play a role in the onset of the intrusions. Figure 2 shows a comparison between conditions outside at sill depth and inside near bottom. Flow inside the sill greater than 10 cm/sec shows the increase associated with increasing salinity. Large salinity differences between inside and outside were coincident with intrusions of new bottom water. A quasi steady-state balance between the pressure gradient and vertical mixing predicts currents about the same as observed. A change in salinity of 1.4 parts per thousand is about what is required to initiate intrusions.

Given that the density difference across the sill accounts for the onset of bottom-water intrusions, the cause of the seaward salinity variations needs to be explained. They may be partly a result of offshore winds affecting the flow in the Strait of Juan de Fuca and partly a result of mixing over a deeper sill in the Strait. Earlier work showed that coastal storm winds drive surface water onshore at the mouth of the Strait. This reverses the surface pressure gradient in the Strait causing inflow at the surface (Holbrook and Halpern 1982) and sometimes reversing the bottom flow and/or suppressing the inflow of saltier water (Cannon and Bretschneider 1986). The present study is the furthest into the Strait that the coastal wind effects have been seen both in the surface and bottom layers, and it appears they have an effect on salinity variations which regulate intrusions of bottom water from the Strait of Juan de Fuca into Puget Sound. The details of this work are described in Cannon *et al.* (1988).

DISCUSSION

Bottom-water intrusions are one of the major circulation features of Puget Sound, and they play a major role in flushing some contaminants. New observations show the onset of the intrusions occurs before minimum neap tides, and simple model calculations demonstrate the importance of the density gradient across the sill caused by salinity variations outside the sill. These variations may be further complicated because the entrance sill is not at the coast, and connects Puget Sound to the Pacific Ocean through another estuary, the Strait of Juan de Fuca. Wind events on the Pacific coast may be capable of causing significant salinity changes more than 135 km from the coast. However, the spring-neap cycle effect on mixing over the sill shown by Geyer and Cannon (1982) still must play the major role. Although Puget Sound primarily is a fjord, the entrance sill resembles a coastal plain estuary, and the process described here may be important for variations in the magnitude of inflow in the bottom layer of a coastal plain estuary. Because this is a time-dependent process, these results are presently being used in guiding the development of a time-dependent laterally averaged model of the Puget Sound estuary which includes this process as well as others.

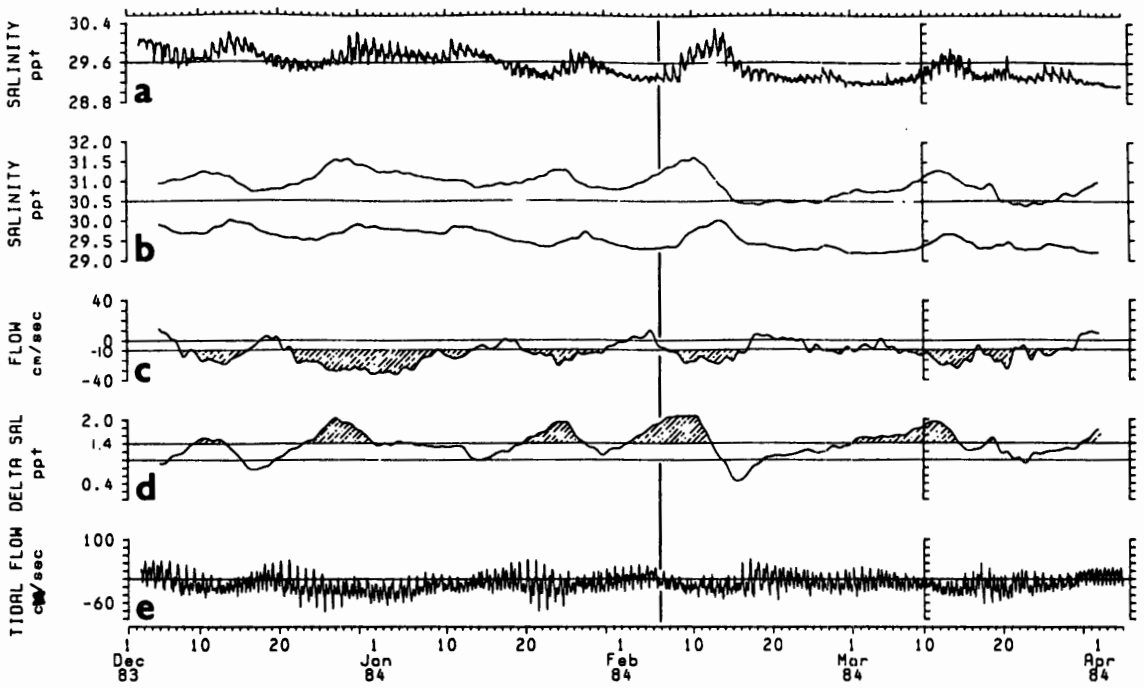

Figure 2. Observed salinity and currents at the entrance to Puget Sound showing the relationship between the salinity (density) difference across the sill and the onset of inflow of new bottom water into the estuary (vertical line at Feb. 6 shows example):

a) salinity inside at the bottom starts increasing;
b) salinity with tidal signal removed ouside at sill depth (upper) starts increasing before inside at the bottom (lower);
c) flow inside at the bottom increases inward, shaded greater than 10 cm/sec inward (plus is seaward);
d) salinity difference across the sill has increased, shaded greater than 1.4 parts per thousand (ppt);
e) tidal flow shows inflow during neap (smaller) currents.

REFERENCES

Cannon, G.A., J.R. Holbrook, and D.J. Pashinski, 1988. Forcing the onset of bottom-water intrusions over the entrance sill of a fjord. To be submitted.

Cannon, G.A., and D.E. Bretschneider, 1986. Interchanges between coastal and fjord circulation. In: Contaminant Fluxes Through the Coastal Zone, G. Kullenberg, ed., Rapp. P-v. Reun. Cons. int. Explor. Mer, 186, 33-48.

Bretschneider, D.E., G.A. Cannon, J.R. Holbrook, and D.J. Pashinski, 1985. Variability of subtidal current structure in a fjord estuary: Puget Sound, Washington. J. Geophys. Res., 90, 11949-11958.

Geyer, W.R., and G.A. Cannon, 1982. Sill processes related to deep-water renewal in a fjord. J. Geophys. Res., 87, 7985-7996.

Holbrook, J.R., and D. Halpern, 1982. Wintertime near-surface currents in the Strait of Juan de Fuca. Atmosphere-Ocean, 20, 327-339.

CONCURRENT SESSIONS AND POSTER SESSION:

TOXICS

Chairs:

Harriette L. Phelps
University of the District of Columbia

Robert C. Hale
Virginia Institute of Marine Science
College of William and Mary

Understanding the Estuary: Advances in Chesapeake Bay Research. Proceedings of a Conference. 29-31 March 1988. Baltimore, Maryland. Chesapeake Research Consortium Publication 129. CBP/TRS 24/88.

Abstract only

Dynamics of Organic Pollutants in Blue Crabs, *Callinectes sapidus*, Collected from the Lower Chesapeake Bay Region

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The blue crab, Callinectes sapidus, is an abundant and widely distributed species in the Chesapeake Bay. It also supports a valuable fishery. Little information concerning concentrations of toxic organic compounds in crustaceans of the southern Chesapeake Bay is currently available. As a consequence, a study to determine the tissue burdens and behavior of lipophilic polycylic aromatic compounds (PACs) in these organisms was undertaken. Identification of compounds was by capillary gas chromatography/mass spectrometry. Quantitation was accomplished by flame ionization detection, using 1,1'-binaphthyl as an internal standard. Highest concentrations of PACs were detected in hepatopancreas, followed by ovarian and muscle tissues. Extractable lipid levels in the tissues were positively correlated with organic xenobiotic concentrations. The major contaminants detected in blue crabs sampled from the southern bay were alkylated PACs, as opposed to unsubstituted polynuclear aromatic hydrocarbons which have been reported to predominate in molluscs and sediments of the bay. This dichotomy may be due to differences in contaminant bioavailability or in the relative abilities of the organisms to eliminate xenobiotics. Crabs from both heavily industrialized and hypothetically "clean" areas, showed evidence of exposure. Chromatographically unresolved complexes were observed in 39% of the hepatopancreas samples. Adult female crabs generally contained higher concentrations of lipophilic pollutants than other intermolt groups.

Evidence that ecdysis in crustaceans may affect the disposition of PACs was observed in laboratory experiments. Crabs were exposed to radiolabeled benzo(a)pyrene (BaP) for 36 h and allowed to depurate for 252 h. Newly molted and adult intermolt female crabs retained greater concentrations of radiolabeled material in hepatopancreas than adult intermolt males or juvenile intermolt females. A variety of oxidized biotransformation products were detected in the hepatopancreas of the crabs by high performance liquid chromatography.

Understanding the Estuary: Advances in Chesapeake Bay Research. Proceedings of a Conference. 29-31 March 1988. Baltimore, Maryland. Chesapeake Research Consortium Publication 129. CBP/TRS 24/88.

Bioassay for Phytotoxicity of Toxicants to Sago Pondweed

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INTRODUCTION

Submerged aquatic vegetation (SAV) occupies an important niche in many aquatic systems, including freshwater lakes, estuaries, and marine coastal waters. As primary producers, SAV captures much of the energy that enters the system, making it available to support a complex biological community. SAV provides food and habitat to numerous organisms common to shallow water ecosystems and maintains homeostasis in the system by buffering changes in water quality through the removal of nutrients, toxics, and particulates. The effects of the loss of SAV from an estuary can have a dramatic effect on life in the estuary, as evidenced by changes that have occurred in the Chesapeake Bay.

The decline of SAV in the Chesapeake Bay during the previous 2 decades has been substantial and well documented (Orth and Moore 1983; 1984). This decline was widespread, affecting all portions of the Bay and all SAV species (Munro and Perry 1982; Orth and Moore 1984).

Synthesis documents prepared for the U.S. Environmental Protection Agency (Kemp et al. 1982; Wetzel et al. 1982) cited water quality problems as the most likely causes of the decline. Two general mechanisms through which water quality stresses SAV were identified. These were phytotoxicity and a reduction of light energy due largely to heavy phytoplankton blooms in response to nutrient enrichment. The reduction of light energy was considered to

be the most important factor, however the effects of contaminants, alone or in combination with other stressors, are poorly understood.

To better understand the role of contaminants on SAV populations, three types of information are particularly useful: measurements of the types and concentrations of contaminants that enter the estuary; knowledge of the environmental conditions necessary for the survival of SAV; and identification of the concentrations of contaminants which negatively impact growth of SAV either directly or in combination with other environmental perturbations.

Analytical chemistry determinations have provided excellent data on the quantity of specific toxic compounds found in the Chesapeake Bay. However, such determinations are costly for routine water quality testing and have limited value for the development of water quality standards. Bioassays are more useful indicators of the presence of toxic substances and provide a more direct link with the resource to be protected than analytical chemistry data. The biological interpretation of such chemical data is often limited by a lack of knowledge of effect levels. Bioassays can be used in the laboratory to establish toxicity levels for specific contaminants or to detect adverse effects on biota of complex effluents entering the bay. The use of bioassays in conjunction with analytical determinations can be a powerful tool for providing the information necessary for legislative restrictions to improve and or maintain the quality of aquatic resources.

Optimal value of a bioassay is obtained when the culture system is well defined, has predictable variability, and is capable of supporting the test organism through all stages of its life cycle. The establishment of a bioassay system for submerged aquatic angiosperms is difficult because of the complexity of their life cycles and the paucity of specific information on the factors regulating reproductive growth. Microcosms have been used for conducting toxicity tests on SAV, but their usefulness is limited because of difficulties in maintaining cultures for long periods of time in a balanced system (Correll and Wu 1982). To avoid these problems, attempts have been made to to establish axenic cultures of submerged angiosperms. Growth in the axenic systems often has been poor or the cultures were not truly axenic (Thursby 1984; Durako and Moffler 1987).

Our laboratory has established axenic cultures of sago pondweed (Potamogeton pectinatus) collected from several locations around the Bay Stem, the Potomac River, and tidal impoundments. Sago pondweed is a perennial, narrow-leafed, submerged aquatic macrophyte found in temperate climates throughout the world (Fernald 1932). It is tolerant to mildly alkaline waters and to a wide range of salinities (between zero and about 10 ppt Yeo, 1965). Stewart (1962) reported that sago pondweed was common and abundant in fresh water and brackish portions of the Chesapeake Bay. Martin and Uhler (1939) stated that sago pondweed is one of the more important waterfowl

food plants in North America. It has a palatable rootstock, turions (tubers), leaves, stems, and seeds. Stewart (1962) and Munro and Perry (1982) reported substantial use of sago pondweed by several waterfowl species through 1959. More recently, it only rarely is found as a food item of waterfowl from the Chesapeake Bay (Munro and Perry 1982; Perry and Uhler In press). The decline of sago pondweed as an important waterfowl food in the Chesapeake Bay is probably related to the overall decline of this and other species of SAV in the Bay.

In our axenic cultures, sago pondweed grows rapidly on a defined medium and exhibits normal vegetative growth. Turions are occasionally formed in cultures placed under various environmental stresses including light deprivation and nutrient depletion. When plants are removed from the culture system they can be easily established in a variety of soils. These plants, after additional growth under long day conditions, have produced both flowers and turions. The system thus provides a convenient source of large numbers of axenic plants which are morphologically similar to plants collected from the field or grown from naturally occurring propagative structures.

While our current culture system holds many advantages, two limitations are noteworthy. First plant size is limited by the culture vessel and attainment of plants large enough to readily produce flowers may require substantial culture modification. Second and most significant, our axenic system is not capable of supporting plants through autotrophic growth. This limitation is a result of an undefined, limiting, environmental factor and not a genetic abnormality; this is confirmed by the plant's high photosynthetic activity when placed on bicarbonate enriched media, and rapid growth when transferred to soils. Pending modifications of the culture system to accommodate autotrophic growth, use of axenic cultures for bioassays requires a two tier bioassay to evaluate the effects of photosynthetic inhibitors in chronic toxicity tests. This paper reports on limited tests with plants produced or tested in axenic cultures.

EXPERIMENTS WITH ATRAZINE

In order to validate the use of axenic cultures in bioassays, we conducted a series of tests to determine how closely toxicity data derived from plants grown in axenic cultures corresponded to results from plants grown in normal, non-sterile microcosms. Our initial work was with the herbicide atrazine, we selected atrazine because of the large data base on its effects on SAV, compared to what is known of the phytotoxic effects of other herbicides or toxicants.

First, we examined the effects of atrazine on photosynthesis. After a preliminary rangefinding study, we selected geometrically arranged concentrations of atrazine to bracket a level that we expected to inhibit net photosynthesis by 50% (IC50). Plants produced in axenic culture were removed from their culture vessel

and placed in clean, non-sterile 300 ml biological oxygen demand (BOD) bottles containing atrazine plus distilled water purged to about 3 μg/l oxygen by bubbling with nitrogen; 2.5 g of sodium bicarbonate was added to each liter of test solution to provide an inorganic carbon source for photosynthesis. BOD bottles were incubated for 5 h in a 21^{0} C waterbath under daylight spectrum fluorescent lights (Vitalite®) producing about 130 μEin/m^{2}/s at the neck of the BOD bottles. Net photosynthetic activity was determined by measuring oxygen produced. Oxygen was determined with an Orion oxygen electrode following the manufacturer's directions. Photosynthesis in control plants resulted in an average of 3.5 $\pm$ 0.64 mg of oxygen produced per g dry weight per hour. Our estimate of the IC50 for atrazine was 29 μg/l (95% CI= 20-40 μg/l; Figure 1). Atrazine's IC50s for photosynthesis in aquatic plants has previously been reported to range from 55 to 117 μg/l (Jones and Winchell 1984; Jones et al. 1985; Kemp et al. 1985) which is similar to our finding, given the differences in testing protocols. Correll and Wu (1982) reported that photosynthesis of sago pondweed was stimulated at 75 μg/l, with respiration exceeding photosynthesis in plants exposed to atrazine at 650 μg/l; why their results differed substantially from ours and those of others is not known.

In the second test, we compared the responses of vegetative plugs of sago pondweed originating from our axenic stock to that of plants growing from tubers collected from the wild, when both were exposed to atrazine at 0, 100, and 1000 μg/l under non-axenic conditions. Using a split plot design, we had five replicates of each treatment level. A replicate consisted of three plant plugs and three tubers placed in a 19 l bucket. Each bucket contained a sand, peat, shell substrate and 14 l of the atrazine-water solution. Weights of plants and tubers were recorded pretreatment and again after 32 days, at which time the experiment was terminated. Water was added to each bucket to compensate for evaporation. Buckets were kept under daylight spectrum fluorescent lights 24h per day (about 75 μEin/m^{2}/s) at ambient room temperatures (about 22^{0} C). Data were analyzed by analysis of variance and Tukey's test for mean separation.

In the control group, proportional increases in biomass were similar between plugs and tubers (426% vs. 440%; Table 1). Compared to these controls, the 1000 μg/g atrazine solution depressed growth ($P < 0.05$) an average of 64% for plants started from plugs, and 50% for those started from tubers; the response of tubers to atrazine was not different from that of vegetative plant plugs ($P > 0.05$). Growth in the 100 μg/l atrazine group did not differ from controls ($P > 0.05$).

The third and most critical part of our test sequence was to determine if plants exposed to atrazine while growing under axenic conditions would respond in a similar way to plants we tested in the buckets. This test was conducted in two parts. In Part 1, we used very young plants, grown for about two weeks from a single rhizome tip. At this time, the tips were just beginning to differentiate into new leaves and additional rhizome tips and weighed <0.05 g.

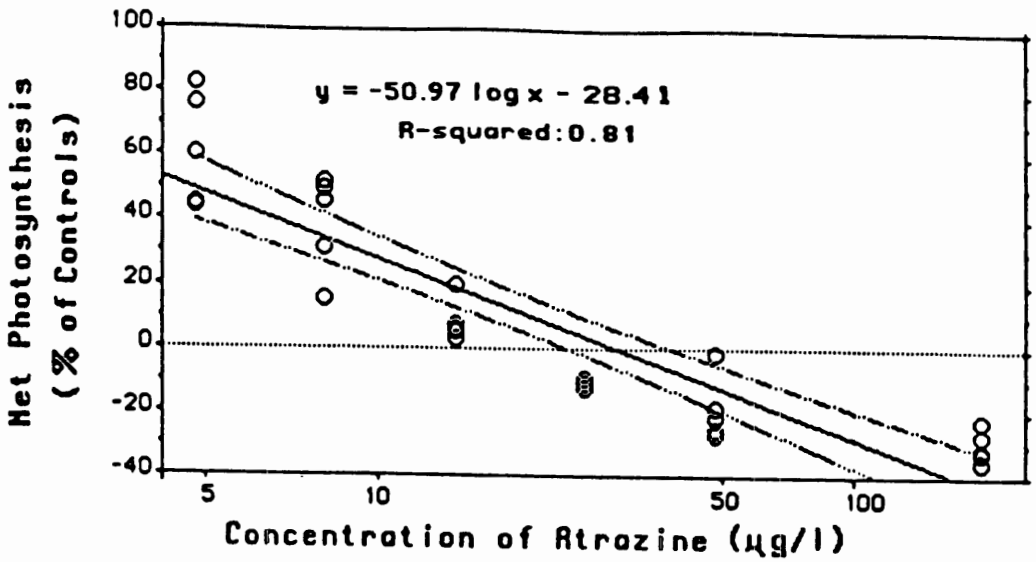

Figure 1. Net photosynthesis of sago pondweed exposed to atrazine. Photosynthesis was determined by measuring changes in dissolved oxygen.

Table 1. Biomass of sago pondweed grown for 32 days in 19 l buckets containing atrazine. Plants introduced into buckets were of two forms, rhizomous tubers and vegetative plugs produced from axenic cultures.

		Atrazine Concentration (µg/l)		
		0	100	1000
Plugs	Pretreatment	0.60±0.16A[a]	0.70±0.19A	0.68±0.10A
	Posttreatment	2.56±1.19A	2.40±0.96A	0.91±0.20B
Tubers	Pretreatment	1.00±0.62A	1.01±0.66A	1.06±0.69A
	Posttreatment	4.40±2.64A	4.19±2.69A	2.20±1.40B

[a]Means followed by different letters within the same row are significantly different ($P \leq 0.05$).

These small plants were placed into 125 ml Erlenmeyer flasks containing 50 ml of filter-sterilized media to which atrazine was added (Table 2). Plants were grown for 7 weeks. Control media and culture media containing the atrazine was replaced with a similar, freshly made media midway through the study. Plants were maintained under full spectrum fluorescent lights at about 20-22° C, with a 14L:10D light cycle.

The results of this test were inconclusive. Whereas our bucket experiment demonstrated a depression of growth at 1000 µg/l atrazine, the data from the flask seemed to show stimulatory responses to low levels of atrazine and no depression of growth at 1000 µg/l. Stimulation of growth has been previously reported to occur in Myriophyllum spicatum exposed to 5 µg/l atrazine, but at 50 µg/l growth was significantly inhibited (Kemp et al. p1985). Further examination of our data indicated much variability within the control and treatment groups and that the growth of the plants was not normally distributed. This variability was related largely to the failure of many of the plants to thrive, resulting in a highly skewed distribution of plant weights. Believing this to be related to the small, relatively morphologically undifferentiated plant material that we started with, we ran a small study (Part 2) using larger, more developed plants, averaging about 0.32 g each.

Part 2 was conducted under the same axenic and ambient conditions as Part 1, but the test period was 8 weeks and the media was not replenished. Variability within treatment groups was greatly reduced in controls from Part 2 compared to Part 1 (Coefficient of variation = 33% vs. 130%). Controls from Part 2 increased their biomass 10-fold during the 8 week test. Results of this experiment were promising (Table 2), but still not convincing as we worked only with high atrazine concentrations in a effort to elicit a plant response. However, 1000 µg/l atrazine under these conditions inhibited growth by a 57%, which is similar to that produced in the bucket experiment.

These growth data compare favorably with those of Forney and Davis (1981) who reported that atrazine's IC50 to Potamogeton perfoliatus was 907 µg/l as determined by final dry weights. They also reported IC50s of 1104 µg/l to M. spicatum and 163-532 µg/l to Vallisneria americana where leaf or plant length was the variable. Correll and Wu (1982) reported that 120 µg/l caused 100% mortality in V. americana, and Cunningham et al. (1984) found that 130 µg/l significantly reduced growth of P. perfoliatus during a 4 week exposure period. Kemp et al. (1982; 1985) reported that increase in biomass was affected at considerably lower concentrations of atrazine, with IC50s of 30-130 µg/l to P. perfoliatus and 91 µg/l to M. spicatum. The wide range of toxicity values for atrazine to SAV probably reflects the absence of a standard toxicity testing protocol and difficulties in maintaining SAV in non-axenic cultures for long periods of time.

Table 2. Biomass (g fresh weight) and number of rhizome tips (R-tips) of sago pondweed grown in axenic culture with atrazine. In Part 1, relatively undifferentiated R-tips were grown in the atrazine solutions for 7 weeks. In Part 2, more mature plants, weighing about 0.3 g when placed in the test solutions, were grown for 8 weeks.

	Atrazine Concentrations (μg/l)					
	0	1	10	100	1000	15000
Part 1						
Biomass	0.71±0.92 B	0.93±0.86 B	2.73±2.12 A	1.74±1.17 AB	0.52±0.43 B	--[b]
R-tips	3.2±2.5 BC	5.0±2.3 BC	11.2±5.6 A	6.4±2.6 B	2.6±1.6 C	--
Part 2						
Biomass	3.47±1.14 A	--	--	--	1.50±0.53 B	0.86±0.34 B
R-tips	9.0±2.4 A	--	--	--	4.2±1.3 B	4.0±1.4 B

[a]Means followed by different letters within the same row are significantly different ($P \leq 0.05$).

[b]Tests not conducted at these atrazine concentrations.

CONCLUSIONS AND FUTURE RESEARCH

Axenic culture techniques for sago pondweed can produce an abundant amount of healthy plants year-round for toxicity testing. Plants produced in axenic cultures showed photosynthetic responses that were similar to published results for SAV exposed to atrazine. Plants produced under these conditions and grown in buckets responded no differently to atrazine exposure than did plants produced by non-axenic methods. Growth of plants exposed to atrazine in axenic cultures varied depending on the starting size of the plants. Larger, more differentiated plants exposed to atrazine in axenic cultures demonstrated a depression of growth similar to those exposed under non-axenic conditions. Plants in our axenic culture system grow far more rapidly than those grown in buckets. This could reduce the amount of labor required to maintain the test plants, and also compress the time required to detect effects. We emphasize this point because the U.S. Environmental Protection Agency requires the use of fresh effluent and frequent renewal of newly collected fresh effluent in many of their aquatic bioassays. Thus, the practicality of a bioassay is partially dependent on the duration of exposure required for each test protocol.

The use of axenic testing procedures for determining the effects of toxicants on plant growth appears to be promising, but is clouded by recent findings that indicate that plants in our axenic culture system appear to be primarily heterotrophic, not autotrophic. We are attempting to refine the current culture technique in an effort to produce an axenic, autotrophic culture system. In the interim, we are continuing to develop protocols for a bioassay in which individual pollutants and effluents from various sources in the Chesapeake Bay area will be examined for phytotoxicity. In this approach, effluents will be first tested for inhibition of photosynthesis. If no inhibition is found, filter sterilized effluents will be tested for their effects on plant growth and morphology in axenic cultures. If photosynthesis is found to be inhibited by an effluent, the phytotoxicity of the effluent will be examined in our nonaxenic bucket "microcosms".

While continuing to explore the use of axenic cultures in toxicity bioassays, we are maintaining contact with State and Federal regulatory agencies concerning requirements for standard protocols for SAV toxicity testing. We hope to begin a trial program of effluent screening in 1988 to evaluate the usefulness and sensitivity of our test system and protocols.

LITERATURE CITED

Correll, D.L. and T.L. Wu. 1982. Atrazine toxicity to submersed vascular plants in simulated estuarine microcosms. Aquat. Bot. 14:151-158.

Cunningham, J.J., W. M. Kemp, M.R. Lewis and J.C. Stevenson. 1984. Temporal responses of the macrophyte, Potamogeton perfoliatus L., and its associated autotrophic community to atrazine exposure in estuarine microcosms. Estuaries 7:519-530.

Durako, M. J. and M. D. Moffler. 1987. Nutritional studies of the submerged marine angiosperm Thalassia testudinum I. Growth responses of axenic seedlings to nitrogen enrichment. Amer. J. Bot. 74(2):234-240.

Fernald, M.L. 1932. The linear-leaved North American species of Potamogeton, Section axillaries. Mem. Am. Acad. Arts Sci. 17(Part 1):1-183.

Forney, D.R. and D.E. Davis. 1981. Effects of low concentrations of herbicides on submersed aquatic plants. Weed Sci. 29:677-685.

Jones, T.W. and L. Winchell. 1984. Uptake and photosynthesic inhibition by atrazine and its degradation products on four species of submerged vascular plants. J. Environ. Qual. 13:243-247.

Jones, T.W., W.M. Kemp, P.S. Estes and J.C. Stevenson. 1985. Atrazine uptake, photosynthetic inhibition, and short-term recovery for submersed vascular plant, Potamogeton perfoliatus L. Arch. Environ. Cont. Toxicol. 15:277-283.

Kemp, W.M., W.R. Boynton, J.J. Cunningham, J.C. Stevenson, T.W. Jones and J.C. Means. 1985. Effects of atrazine and linuron on photosynthesis and growth of the macrophytes, Potamogeton perfoliatus L. and Myriophyllum spicatum L. in an estuarine environment. Marine Environ. Res. 16;255-280.

Kemp, W.M., J.C. Means, T.W. Jones and J.C. Stevenson. 1982. Herbicides in Chesapeake Bay and their effects on submerged aquatic vegetation. Pages 503-567 in Chesapeake Bay Program Technical Studies: A synthesis. U.S. Environ. Protect. Agen., Washington, D.C.

Martin, A.C. and F.M. Uhler. 1939. Food of game ducks in the United States and Canada. U.S. Dept. Agr. Tech. Bull. No. 634. Washington, D.C. 308 pp.

Munro, R.E. and M.C. Perry. 1982. Distribution and abundance of waterfowl and submerged aquatic vegetation in Chesapeake Bay. U.S. Environ. Protect. Agen. Rept. No. 600/3-82-082. Washington, D.C. 180 pp.

Orth, R.J. and K.A. Moore. 1983. Chesapeake Bay: An unprecedented decline in submerged aquatic vegetation. Science 222:51-53.

Orth, R.J. and K.A. Moore. 1984. Distribution and abundance of submerged aquatic vegetation in Chesapeake Bay: A historical perspective. Estuaries 7:531-540.

Perry, M.C. and F.M. Uhler. 1988. Food habits and distribution of wintering canvasbacks (Aythya valisineria) on Chesapeake Bay. Estuaries (In press).

Stewart, R.E. 1962. Waterfowl populations in the upper Chesapeake Region. U.S. Fish and Wildl. Serv. Spec. Sci. Rept. -- Wildlife No. 65. Washington, D.C. 208pp.

Thursby, G. B. 1984. Nutritional requirements of the submerged angiosperm Ruppia maritima in algae-free culture. Marine Ecol-Prog. Ser. 16:45-50.

Wetzel, R.L., R.F. van Tine and P.A. Penhale. 1982. Light and submerged macrophyte communities in Chesapeake Bay: A scientific summary. Pages 568-630 in Chesapeake Bay Progam Technical Studies: A Synthesis. U.S. Environ. Protect. Agen. Washington, D.C.

Yeo, R.R., 1965. Life history of sago pondweed. Weeds 13:314-321.

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Inducible Adaptations as Bioassays

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Organisms have been used to test environmental quality for a long time. The earliest test organism perhaps was human, the earliest experimenter a King and the environment being tested the King's food. While in environmental matters Kings are seldom the investigators and humans are rarely used as an indicator species, at least with their consent, death is still a widely used endpoint. Death is unambiguous and premature death indicates poor environmental quality. However, in biomonitoring and bioassay, as in medicine, additional criteria are needed. Just as death can indicate extreme conditions, sublethal biological tests can indicate less severe conditions which may nevertheless lower long-term survival or reproduction (Rand 1980).

Many indicators of sublethal stress have been identified. Some of these are adaptations of the organism and some may be usable in bioassay, particularly in water quality measurement. Reproduction is one such criterion forming a "critical interface between the individual and the population" (Widdows 1985b). Some methods using reproduction are described in Horning and Weber (1985). Growth rate or potential for growth rate is another useful criterion (Widdows 1985a, 1985b). These assays require considerable time and effort, so shorter perhaps less expensive methods have also been suggested at least for screening and monitoring, if not as definitive bioassays.

Short-term adaptations which are induced in individual organisms by an environmental stimulus seem ideally suited to testing water quality. A large number of sublethal tests, some of which may represent

adaptations, have been proposed. Many of these tests are biochemical, including mixed function oxidases, heavy-metal binding proteins, lysosomal latency, levels of steroid hormones, adenylate energy charge, O:N ratios, RNA/DNA ratios, amino acid ratios and others. They are discussed in Bayne et al (1985), in McIntyre and Pearce (1980), in Dillon and Lynch (1981) and in Ivanovici and Wiebe (1981).

What we wish to describe in this paper are two inducible responses to environmental stress, one observed in the intact organism and the other at the level of protein synthesis. Both seem useful as bioassays. A third test, based on changes in membrane microviscosity has potential but is less well developed than the first two. Each of these responses is a threshold trait but each is quantitative as well. The organismic test is general, the protein synthesis test appears to be both specific and general, as will be explained, and the membrane viscosity test is general, in the sense that the response is the same to all stressors.

These assays have been investigated with the calanoid copepod *Eurytemora affinis*, indigenous to the Chesapeake Bay and, it seems, to most other temperate estuaries in the Northern Hemisphere. The virtues of this organism for ecology, genetics and physiology have been extolled in other papers. Suffice it to say here that if indigenous species are to be used to test estuarine waters, and they should, then calanoid copepods ought to be among them. *Eurytemora* has been used at times in standard LC_{50} tests (e.g. Liden et al 1980) and in tests of reproduction (Bradley 1986)

THE WHOLE ORGANISM BIOASSAY: INDIVIDUAL TIME TO ENTER COMA (ITEC).

This assay has been used for over 10 years now, mainly to measure temperature tolerance. In its original form (Bradley 1975) the diagnostic was a combination of time to "succumb" (TS) or enter a coma and time to recover (TR) from the coma, when individual copepods were shocked at 34.5°C. At that time we used an index 30 + TS - TR as temperature tolerance. Later the index became simply TS when we found that TR added little or no information, having a high negative correlation with TR. Presently the assay consists of exposure to stepwise 1/2°C increases in temperature each 5 minutes beginning at 32°C, to accommodate the widest possible ranges of tolerance.

What we previously referred to as "The Assay" and are now calling ITEC can be used for any water soluble stressor or combination of stressors. Copepods are placed individually in vials open at both ends, with 73μ mesh enclosing the bottoms. These vials in a rack of 20 are placed in a 5 gallon aquarium filled with the test solution. In the temperature tolerance assay the water would be filtered bay water at approximately 5 ppt salinity. The copepods are then observed until inactivity and the individual times noted. If needed for further experimentation, each vial can be removed and placed in clean bay water without further harming the copepod. We have tested the correlation between this assay and survival at the individual and family levels for temperature, but not for other stressors. For temperature stress, we established quite early (Bradley 1976) that the test was valid and recently we have repeated the validity test on a much larger scale using the current assay.

The later results confirm that ITEC predicts survival quite well. The correlations with survival times were 0.80 and 0.64 for males and females, respectively. When survival was measured on full siblings of the tested copepods the correlations were 0.63 and 0.53, respectively. The latter correlations would be expected to be lower since survival was measured in siblings, with 50% relationship, rather than on the same individuals. When genetic correlations between ITEC and survival were measured the estimates were 1.09 $\pm$ .10 for males and 0.62 $\pm$ for females (Bradley et al 1988). As was the case in the earlier work (Bradley 1976) the prediction of survival was markedly more accurate at 29.5°6 than at 27°C. The former temperature represents the normal upper limit the species would experience in Chesapeake bay.

Advantages and disadvantages

The advantages of this test are that it is short term, nondestructive and quite repeatable. It also combines the advantages of static and flow through tests in its simplicity and realism. (Davis and Bradley 1988). Water constantly circulates through the vials and if the level of contaminant in the aquarium is maintained, flow through conditions are approximated quite well. The test could be used in situ by exposing racks of organisms in the mesh bottom vials to ambient water. However, with prolonged exposures in the field some prefiltering would be necessary to prevent fouling of the mesh bottoms.

There is no reason why the assay could not be used with fish larvae. Other forms may die rather than enter a coma, but even in these cases time to death would be the criterion for tolerance, and the assay would still be quantitative.

One disadvantage of this assay is variability among and within sets of organisms tested. Variation among individuals was reduced some 40-fold in tests using chlorine oxidant simply by opening the vials to the surrounding environment adding the mesh bottoms to vials). This was apparently because the conditions were now uniform among vials (Davis and Bradley, unpublished). A second minor disadvantage is that levels of contaminant must be adjusted to allow some time to elapse before coma or death.

It seems to us that physiological and biochemical tests should be added to routine testing programs. Such tests could be used for field surveys to identify hot spots and for in situ monitoring. If definitive they could be used in laboratory bioassays of whole effluent, specific contaminants and potential contaminants. Since many of the sublethal tests are stress tests they might also be used to assess the physiological condition of stocks of biota in their natural habitat or in aquaculture. These latter applications are already under serious consideration.

There are a number of biochemical and cellular indicators of stress which have potential use in bioassay. Some of these were listed earlier. In this laboratory we are presently examining two adaptive responses at the suborganismal level. These have been described quite fully in Bradley et al (1988) and the details will not be repeated here. The focus will be on their usefulness in bioassay.

STRESS PROTEIN IMMUNODETECTION ASSAY (SPIDA).

The synthesis of novel proteins, what were and still are referred to as heat-shock proteins, provides an attractive basis for a biological assay of water quality.

The heat shock response was first documented over 100 years ago but only in 1962 did the phenomenon receive further attention (Nover 1984). The response includes changes in the cell membrane, to be discussed, in the cytoskeleton and in virtually all of cellular metabolism. The most dramatic response, which occurs with types of stress other than heat shock, is the synthesis of a collection of heat shock proteins. Across the species examined there seem to be about 5 major heat shock proteins (68 - 110kD) and many smaller heat shock proteins (15 - 25 kD)

The fact that these proteins are so ubiquitous, highly conserved and are induced by a variety of stressors from oxidizing agents to chelating drugs, wounding to viral infections as well as heavy metals, makes them very attractive as indicators of stress. In view of this general response these proteins are now often referred to as stress proteins.

We have identified five common stress proteins in *Eurytemora* (Bradley et al 1988) and several others in mysids. The proteins induced and their quantity varies with the stressor. To date we have examined the response to heat shock, chlorine oxidant (35 ppb) and tributyltin (0.5 µg/L). The major proteins consistently present and identified after labeling with ^{35}S methionine (methods in Bradley et al 1988) were as follows, according to stressor:

Heat shock	Oxidant	TBT
Hsp 109[a]		
Hsp 98		
Hsp 82		
Hsp 70	Hsp 70 (lesser)	Hsp 70 (lesser)
	Hsp 52	Hsp 46
	Hsp 48	Hsp 43
Hsp 24.5	Hsp 24	Hsp 24

[a]Hsp 109 = heat shock (stress) protein with molecular wt. (approx.) 109,000D.

The method could be used for routine screening, biomonitoring or bioassay as it exists presently but it is impractical. Animals must be radioactively labelled and extracts run on separating gels to identify the newly synthesized proteins autoradiographically.

Two other treatments, anoxia and cold shock did not induce synthesis, when the animals remained under stress. As is the case with other stresses, however, protein synthesis in general was suppressed.

Advantages and disadvantages

The pattern of response to the stressors examined so far, in addition to the work of others, suggests that the response is both general and specific. The phenomenon itself occurs in response to a range of stressors, as also indicated earlier, and, in addition, there is some promise of specificity in pattern as was seen in comparing responses to heat shock, chlorine oxidant and TBT. The latter conclusion requires much more data. At the moment the method is not practical for routine screening, biomonitoring or bioassay, although it is being investigated for use in shellfish (Sanders, 1987). The stress protein assay will be routine when a simpler method of detecting the proteins, probably immunological, is developed. At the time of writing several detection methods, on which we are working, seem quite feasible.

PLASMA MEMBRANE MICRO VISCOSITY

In collaboration with Dr. James Vincent of the Chemistry Department at UMBC we have been examining phase changes in plasma membrane viscosity using the spectra of light scattered at certain wavelengths by non-lipid molecules in the lipid bilayer, excited by a laser beam--the so-called Raman effect (Bradley et al 1988). The natural probe molecule is the carotenoid astaxanthin and intact organisms are observed. By observing intensity ratios at two wavelengths, membrane phase changes can be inferred and thus any adaptation to stress at the membrane level can be detected.

Advantages and disadvantages

The advantages of the response as an assay are its simplicity (given the availability of spectroscopic equipment), its use of intact organisms and the generality of the response to a number of stressors (Kasai et al 1976). Its practical application depends on modification to allow animals to be observed in a continuous series and, most importantly, on distinguishing spectra of the carotenoid in plasma membrane phospholipids and in other lipids. Changes in the former affect membrane viscosity and would be adaptive. Changes in the latter would not be.

We need to examine whether these assays meet the criteria for bioassay or at least for tests of stress. The following are criteria gathered from several sources (Widdows 1985a,b, Livingstone 1985, Lee et al 1980, Brungs and Mount 1978) and paraphrased. Some of these papers include evaluations of current techniques. The criteria are not in order of priority.

CRITERIA FOR SUBLETHAL BIOASSAYS

1. Sensitivity to sublethal stress.
2. Rapid response time
3. Quantitative and predictable relationship to contaminant
4. Response through optimal to lethal range
5. High signal-to-noise ratio
6. Related to some ecological or pathological effect.
7. Broad application to a range of contaminants or phyla.
8. Inexpensive equipment, low running costs
9. Simplicity

Also, when used in biomonitoring, an assay should be unaffected by short-term fluctations in a stressor and should be usable in field or laboratory.

The tests described briefly in this paper, namely ITEC and SPIDA, seem to meet most of these criteria. ITEC meets all the criteria with the exception of no. 5. Stress conditions are recognized by having enough test organisms for biologically significant perturbations to be statistically significant also. ITEC is not particularly useful for biomonitoring.

The stress protein assay seems to be applicable to initial screening, to biomonitoring and to laboratory bioassay and seems to meet all the criteria. The case for no. 6, the relationship to some ecological or pathological effect, is least clear. There is no question that stress proteins are synthesized under almost any recognizable and significant stress. Stresses due to handling are not sufficient to induce them. Stress proteins also appear to confer protection to the cells until the stress passes or is accommodated in some way (Nover, 1984). It is not clear whether the organism which is the first to synthesize these proteins (and to shut down other protein synthesis) is the most sensitive or the most resistant to the stress. So we do not know the form of the relationship to an ecological effect. Having recently reviewed existing techniques used on Chesapeake bay biota (Bradley and Roberts, 1987), we, the present authors that is, believe SPIDA is competitive with any others we are aware of.

So the points made in this paper are that sublethal tests ought to be even more widely used than they are and that adaptational responses in the organisms, triggered by environment stresses, should be used as diagnostics of those stresses. The test organism is in effect being used to integrate what otherwise may be confusing physico-chemical signals into a single response.

Finally, the individual adaptations we have been discussing need to be related to responses at the population and ecosystem levels. Just as changes at the cellular level may be merely homeostatic adjustments not affecting the organism, so changes at the organismic level may have slight consequences to the population. On the other hand, if biochemical tests could be used to predict population effects then perhaps they would be more useful and would receive a better rating than they did in a survey by Brungs and Mount (1978). At that time physiological and biochemical tests received a relative utility rating (based on several criteria) lower than 13 other kinds of tests, finishing just above "in vitro".

REFERENCES

Bayne, B.L., Brown, D.A., Burns, K., Dixon, D.R. Ivanovici, A., Livinstone, D.R., Lowe, D.M., Moore, M.N., Stebbing, A.R.D. and Widdows, J. 1985. The Effects Of Stress And Pollution On Marine Animals. Praeger Special Studies, New York, 384 pp.

Bradley, B.P. 1975. The anomalous effect of salinity on temperature tolerance on the copepod *Eurytemora affinis*. Biol. Bull. 148:26-34.

Bradley, B.P. 1976. The measurement of temperature tolerance: Verification of an index. Limnol. and Oceanog. 21:596-599.

Bradley, B.P. 1986. Traits, problems and methods in copepod life history studies (Proc. 2nd Int. Conf. Copepoda, Nat. Mus. Can. Syllogeus 58:247-253).

Bradley, B.P. and M.H. Roberts, Jr. 1987. Effects of Contaminants on estuarine Zoopolankton In: Contaminant Problems and Management of Living Chesapeake Bay Resources (S.K. Majumdar, L.W. Hall, Jr. and H.M. Austin, Eds.) Philadelphia:Pennsylvania Academy of Science, pp. 417-440.

Bradley, B.P., R. Hakimzadeh and J.S. Vincent 1988. Rapid responses to stress in the copepod *Eurytemora affinis* (Hydrobiologia, in press)

Brungs, W.A. and D.I. Mount 1978. Introduction to a discussion of the use of aquatic toxicity tests for the evaluation of the effects of toxic substances In: Estimating the hazards of chemical substances to aquatic life (J. Cairns, Jr. K.L. Dickerson and A.W. Maki, Eds.) ASTM STP 65).

Davis, R.M. and B.P. Bradley 1988. Effects of chlorine oxidant and low oxygen on copepods Proc. 15th Symp. Env. Effects of Water Chlorination. (R.L. Jolley, Ed.) Oak Ridge National Lab. (in press).

Dillon, T.M. and M.P. Lynch 1981. Physiological responses as determinents of stress in marine and estuarine organisms pp. 227-241. In G.W. Barrett and R. Rosenberg (Eds.) Stress effects on Natural Ecosystems John Wiley, N.Y. 305 pp.

Horning, W.B. and C.I. Weber (Eds.) 1985. Short term methods for estimating the chronic toxicity of effluents. EPA /600/4-85/014.

Ivanovici, A.M. and W.J. Wiebe 1981. Towards a Working 'Definition' of 'Stress': A review and critique p. 13-27 In: (Barrett, G.W. and R. Rosenberg, Eds.) Stress Effects on Natural Ecosystems John Wiley.

Kasai, R., Y. Kitajima, L.E. Martin, Y. Nozawa, L. Skriver, and G.A. Thompson, Jr. 1976. Molecular control of membrane properties during temperature acclimation. Membrane fluidity regulation of fatty acid desaturase action? Biochem. 15: 5228-5233.

Lee, R., J.M. Davies, H.C. Freeman, A. Ivanovici, M.N. Moore, J. Stegeman and and J.F. Uthe 1980. Biochemical techniques for monitoring biological effects of pollution in the sea (Biochemistry Panel Report) In McIntyre A.D. and J.B. Pearce (Eds) Biological Effects of Marine Pollution and the Problems of Monitoring. Rapports et Proces - Verbaux Des Reunious Vo.. 179 ICES Copenhagen..

Liden, L.H., D.T. Burton, L.H. Bongers and A.F. Holland 1980. Effects of chlorobrominated and chlorinated cooling waters on estuarine organisms Water Pollut. Contr. Fed. J. 52:173-186.

Livinstone, D.R. 1985. Biochemical measurements In Bayne, B.L., Brown, D.A., Burns, K., Dixon, D.R. Ivanovici, A., Livinstone, D.R., Lowe, D.M., Moore, M.N., Stebbing, A.R.D. and Widdows, J. The Effects Of Stress And Pollution On Marine Animals. Praeger Special Studies, New York, 384 pp.

McIntyre, A.D. and J.B. Pearce (Eds.) 1980. Biological Effects of Marine Pollution and the Problems of Monitoring. Rapports et Proces - Verbaux Des Reunious Vol. 179 ICES Copenhagen.

Nover, L. (Ed.) 1984. Heat shock response of eukaryotic cells. Springer - Verlag, Berlin.

Rand, G.M. 1980. Detection:Bioassays pp. 390 - 403 In: (Guthrie, F.E. and J.J. Perry, Eds.) Introduction to Environmental Toxicology Elsevier, N.Y.

Sanders, B.M. 1987. The stress protein response; A novel monitoring technique. 8th Annual SETAC meeting Pensacola, FL. November 1987.

Widdows, J. 1985a. In Bayne, B.L., Brown, D.A., Burns, K., Dixon, D.R. Ivanovici, A., Livinstone, D.R., Lowe, D.M., Moore, M.N., Stebbling, A.R.D. and Widdows, J. The Effects Of Stress And Pollution On Marine Animals. Praeger Special Studies, New York, 384 pp.

Widdows, J. 1985b. Physiological responses to pollution Mar. Pollut. Bull. 16:129-133.

Understanding the Estuary: Advances in Chesapeake Bay Research. Proceedings of a Conference. 29-31 March 1988. Baltimore, Maryland. Chesapeake Research Consortium Publication 129. CBP/TRS 24/88.

Abstract only

The Role of Photochemistry in the Bioavailability of Toxic Substances in Estuaries; Implications for Monitoring

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Interaction of solar photons with toxic substances in surface waters can have profound effects on the chemical and biological behavior of those substances. The study of these effects is in its infancy. However, a number of significant examples can already be cited. In some cases, the result of photochemical reactions is detoxification. One example is the reduction by photochemically produced hydrogen peroxide of free chlorine discharged from power plants and wastewater treatment plants. Another example is the photoreduction of Cr (VI), a toxic and carcinogenic form of chromium, to the harmless Cr (III) form. This has been observed as active process in Back River and is probably responsible for removal to sediments of much of the chromium discharge from the large wastewater treatment plant on that tributary. On the other hand, cases are known in which toxicity is sharply increased by photochemical transformations. Such reactions are associated with weathering of petroleum at oil spill sites. Oxidation of Parathion to Paraoxon, a more toxic compound than its parent, can be accomplished by peroxides. Other examples are known, but much research on these processes remains to be done.

The existence of photochemical transformations that affect toxicity has important implications for the design of environmental monitoring programs. Large diurnal variations in the concentrations of some toxic substances may occur, requiring special sampling precautions. Programs designed to monitor "at the pipe" may grossly under- or overestimate the environmental impact of toxic substances, depending on the nature of subsequent photochemical transformations. Laboratory toxicity studies involving aquatic organisms may generate misleading results unless attention is paid to light flux.

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Abstract only

Dechlorination: Is it the Answer?

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Since World War II, sharp declines in fish catches in freshwater tributaries of the Chesapeake has corresponded in time with the construction of numerous waste water treatment plants. The coincidence of these events has made chlorine, used in water water disinfection, a suspect in the declines of catches. As a consequence, Maryland and to a lesser extent Virginia have become national leaders in the installation of dechlorination systems at wastewater treatment plants. Typically these systems employ SO_2 to destroy residual chlorine. Research in our laboratory shows that dechlorination does not completely destroy all components of residual chlorine, as had been thought. Experience at three plants suggests that 10% of the residual chlorine typically is unreactive to SO_2 on the time scale of holdup in a treatment plant. The actual compounds involved have not been identified, but experience with model compounds in the laboratory suggests that peptides and secondary amines can form chloramines that fail to react with SO_2. The sulfite-resistant fraction of residual chlorine at treatment plants can be partially extracted into nsoctanol. This evidence of mild hydrophobicity suggests that these compounds may prove toxic to fish. Certain hydrophobic organic chloramines have proven to be nearly as toxic as free chlorine to fish. Clearly further toxicological investigations are in order. Additional work in our laboratory has failed to find compelling evidence that the discharge of sulfite in dechlorinated effluents is itself an environmental hazard, although some release of toxic copper from sediments may result. In the course of our field studies, we did observe evidence that the dechlorination process is often poorly controlled, resulting in periods when residual chlorine is released and periods when gross excesses of sulfite are released.

Understanding the Estuary: Advances in Chesapeake Bay Research. Proceedings of a Conference. 29-31 March 1988. Baltimore, Maryland. Chesapeake Research Consortium Publication 129. CBP/TRS 24/88.

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.

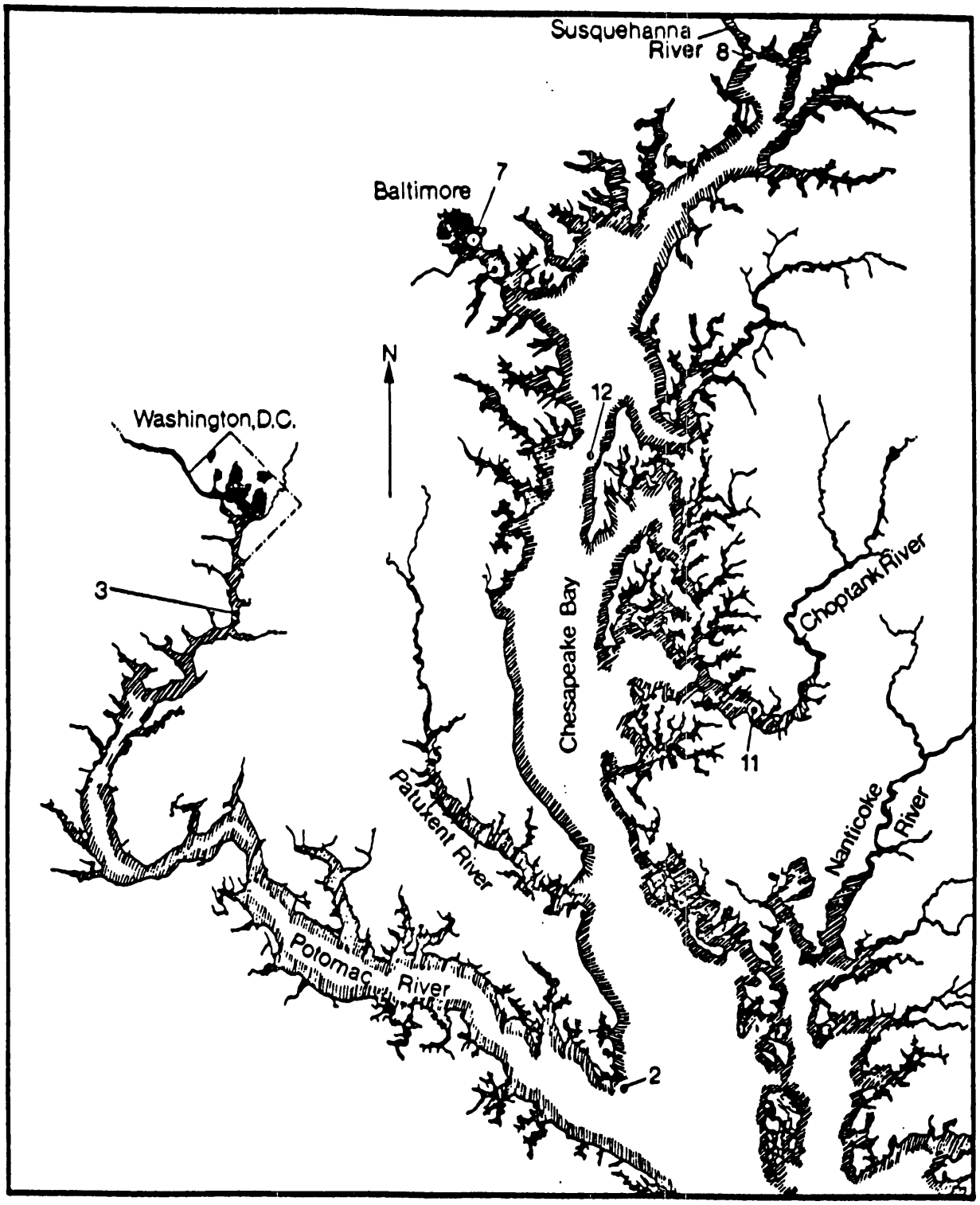
INTRODUCTION

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).

Figure 1. Station Locations



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

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

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.

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 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

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

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

Location	Butyl Tin (ng/l) 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 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		

(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.

REFERENCES

Cross, J.N., Hardy, J.T., Hose, J.E., Hershelman, G.P., Antrim, L.D., Gossett, R.W. and Crecelius, E.A, 1987. Contaminant concentrations and toxicity of sea-surface microlayer near Los Angeles, California. Mar. Environ. Res. (in press).

Crow, S.A., D.G. Ahern, W.L. Cook, and A.W. Bourquin. 1975. Densities of bacteria and fungi in coastal surface films as determined by membrane adsorption procedure. Limnol. Oceanogr. 20:644-646.

EPA. 1982. Results of the Nationwide Urban Run-off Program. Vol II, Appendices. Washington, D.C.

Grant, G.C. and J.C. Olney, S.P. Berkowitz, J.E. Price, P.O. Smyth, M. Vecchione, and C.J. Womack. 1979. Middle Atlantic Bight zooplankton: second year results and a discussion of the two-year BLM-VIMS Survey. Virginia Institute of Marine Science, Gloucester Point, Virginia 23062. Special Report in Applied Marine Science and Ocean Engineering, No. 192. Chapter 4, 236 p.

Gucinski, H. 1985. Correlation of biophysical surface characteristics with hydrodynamic properties of adhesive biofilms. Ph.D. Thesis, Roswell Park Mem. Inst. SUNY at Buffalo.

Gucinski, H. 1986. The effect of sea surface microlayer enrichment on TBT transport. Oceans '86. Proceedings Vol. 4 Organotin Symposium. pp. 1266-1274.

Hardy, J.T., E.A. Crecelius, L.D. Antrim, S.L. Kiesser, and V.L. Broadhurst. 1987. Aquatic Surface Microlayer Contamination in Chesapeake Bay. Contract Report to Maryland Department of Natural Resources, Energy Administration, Power Plant Research Program, Annapolis, MD 39 p.

Hardy, J.T., S. Kiesser, L. Antrim, A. Stubin, R. Kocan, and J. Strand. 1988a. The sea-surface microlayer of Puget Sound: Part I. Toxic effects on fish eggs and larvae. Mar. Environ. Res. 23.

Hardy, J.T., E.A. Crecelius, L.D. Antrim, V.L. Broadhurst, C.W. Apts, J.M. Gurtisen, and T.J. Fortman. 1988b. The sea-surface microlayer of Puget Sound: Part II. Concentrations of contaminants and relation to toxicity. Mar. Environ. Res. 23.

Hardy, J.T., J.A. Coley, L.D. Antrim, and S.L. Kiesser. 1988c. A hydrophobic large-volume sampler for collecting aquatic surface microlayers: characterization and comparison to the glass plate. Can. J. Fish. Aquatic Sci. 45(5).

Hartwig, E.O. and F.L. Herr. 1984. Chemistry and biology of the sea-surface interface relationships to remote sensing. Off. Naval Res. Workshop, Sanibel Island, Florida, Oct. 23-25, 1984.

Hunter, J.R. 1981. Feeding ecology and predation of marine fish larvae. In: R. Lasker (ed.), Marine fish larvae, pp. 33-77. Washington Sea Grant Program.

Kocan, R.M., M.L. Landolt, and K.M. Sabo. 1982. Anaphase aberrations: a measure of genotoxicity in mutagen-treated fish cells. Environ. Mutag. 4:181-189.

Lippson, A.J., M.S. Haire, A.F. Holland, F. Jacobs, J. Jensen, R.L. Moran-Johnson, T.T. Polgar, and W.R. Richkus. 1979. Environmental atlas of the Potomoc Estuary. Prepared for the Maryland Department of Natural Resources, Power Plant Siting Program, by Martin Marietta Environmental Center, Baltimore, Maryland, 280 p.

Longwell, A.C. and J.B. Hughes. 1980. Cytologic, cytogenetic and developmental state of Atlantic mackerel eggs from sea surface waters of the New York Bight, andprospects for biological effects monitoring with ichthyoplankton. Rapp. P.V. Reun. Cons. Int. Explor. Mer, 179:275-291.

Olson, M. 1983. Surface energies and chemical analysis of the initial stages of marine microbiological fouling. Trident Scholar Proj. Rep. No. 127, US Naval Academy, Annapolis, MD.

Provenzano, A.J., Jr., J.R. McConaugha, K.B. Philips, D.F. Johnson, and J. Clark. 1983. Vertical distribution of first stage larvae of the blue crab, Callinectes sapidus, at the mouth of Chesapeake Bay. Estuarine Coastal Shelf Sci. 16(5):489-499.

Provenzano, A.J., Jr., J.M. McCanaugha, and D.F. Johnson. 1982. Significance of the neuston layer in the dispersal of larvae of the blue crab Callinectes sapidus. J. Shellfish Res. 3(1):99.

Sieburth, J.M., 1982. Microbiological and organic-chemical processes in the mixed layer and the surface skin of the sea. Nato Adv. St. Symp., Durham, NH, July 1982.

Wass, M.L. (Ed.). 1972. A checklist of the biota of lower Chesapeake Bay. Special Scientific Report No. 65. Virginia Institute of Marine Science, The College of William and Mary, Cloucester Point, VA 290 p.

Zaitsev, Y.P. 1971. Marine neustonology. Acad. Sci. Ukr. SSR. (Trans. from Russian). National Marine Fisheries Service and the National Science Foundation, Washington, D.C., National Technical Information Service, Springfield, Virginia. 207 pp.

Understanding the Estuary: Advances in Chesapeake Bay Research. Proceedings of a Conference. 29-31 March 1988. Baltimore, Maryland. Chesapeake Research Consortium Publication 129. CBP/TRS 24/88.

Assessing the Impact of DOD's Installations on the Water Quality of the Chesapeake Bay Region

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INTRODUCTION

The Department of Defense (DoD) recently completed a two-year study to determine the relative impact of its activities on the water quality and living resources of the Chesapeake Bay and its tributaries. This study, a major DoD contribution to the 1984 Joint Resolution with the Environmental Protection Agency (EPA), also identified existing DoD projects and programs that either protect Bay resources or reduce the severity of adverse impacts, and provided recommendations for potential restoration activities.

A total of 66 DoD installations were evaluated, representing all in the drainage basin with the potential to impact the Bay's water quality by virtue of their size, their proximity to the Bay, or the types of activities which they perform. Included were 37 Navy, 22 Army, six Air Force and one Defense Logistics Agency installation. A wide range of activities having the potential to impact the Bay's water quality were examined, among which were point and nonpoint sources, storage and disposal of hazardous materials, munitions production and testing, maintenance operations, and ongoing environmental programs.

PROJECT OVERVIEW

The study was divided into three phases. Phase I defined the recent historical and present pollution potential of all 66 installations. A preliminary screening procedure was developed, identifying 37 installations with a significant impact potential. A computerized data base describing the water quality on and in the immediate vicinity of each installation was also developed. Phase II developed and tested a detailed assessment methodology on six of the 37 installations to define more precisely the character and extent of military activities on the Bay's water quality. Phase III applied this methodology to the remaining 31 installations identified in Phase I as needing more detailed assessment, and summarized potential impacts and program recommendations for each installation. Regional and Bay wide impacts were also described.

SCREENING PROCEDURE

A two tiered screening procedure was designed to provide two different perspectives of an installation's impact. The first used on-site data to evaluate an installation's potential to impact the Bay and its tributaries. Twenty-four screening criteria, grouped in four categories -- nonpoint sources (4 criteria), point sources (3), hazardous materials (11) and environmental programs (6) -- were used. Criteria selected were key military activities with the highest potential to have a significant impact on surface waters, and included all major potential pollutant sources identified by the EPA Chesapeake Bay Program. This procedure allowed for the pollution impact potential of military activities to be compared on an installation by installation basis.

The second tier used off-site data to develop a set of seven vicinity screening criteria. These were used to evaluate an installation's impact potential relative to its surrounding environment; i.e. its proximity to significant ecological resources and its relative impact on local receiving waters

PHASE I

A preliminary screening of all 66 DoD installations under evaluation was made during Phase I, which assigned each installation to one of four study groups:

1. Significant water quality impact potential;
2. Poorly defined, but likely significant impact potential;
3. Poorly defined, but likely insignificant impact potential;
4. Insignificant impact potential.

Thirty-seven installations were initially rated in Study Group 1 (12) or Study Group 2 (25). These were addressed in more detail in Phases II and III of the study. An installation generally was assigned to Study Group 1 if observations indicated contamination of surface waters in excess of water quality criteria or guidelines. A Study Group 2 installation demonstrated potential pollutant sources with characteristics similar to those found in Study Group 1. However, the lack of appropriate data to verify the existence of pollutants in the receiving waters adjacent to these installations precluded their assignment to Study Group 1.

The remaining twenty-nine installations were rated in Study Group 3 (17) or Study Group 4 (12), and did not receive further detailed study. This reflects the likely absence of any significant pollutant sources at these installations, or that they had significantly reduced or eliminated practices that at one time created water quality concerns. Nevertheless, recommendations for improvements at these locations were idebtified during Phase III.

PHASE II

Phase II developed a detailed assessment methodology to more clearly define the probable character and extent of an installation's impact on water quality and living resources in its immediate vicinity, its tributaries, and the Bay proper. The methodology used available data to quantify the impacts of contaminant sources such as conventional pollutants, toxics, and turbidity. Phase II also evaluated the methodology's effectiveness on the six installations selected from the Phase I initial screening.

It should be noted that the assessment methodology is highly dependent on the availability of data on contaminant source characteristics and receiving water quality in the vicinity of the installation. Where detailed information was lacking, and a potential impact was probable, recommendations were made to fill information gaps.

PHASE III

In Phase III, the refined installation assessment methodology was applied to the remaining 31 installations which were under expanded evaluation. A major goal of applying this methodology was to more precisely define the impact of installations initially placed in the two "poorly defined" impact categories. (Study Groups 2 and 3)
A report summarizing DoD impacts by installation, Service, geographic region and Bay-wide was produced.

In addition, the Phase III report included recommendations for all 66 installations, identifying practices or projects that could be implemented at specific locations to improve the water quality and living resources of the Bay. General cost estimates and a qualitative description of expected benefits were prepared for each major program recommendation, to aid in developing an implementation strategy for these actions.

FINAL SCREENING RESULTS

Results of the final screening placed 15 installations in Study Group 1, 16 in Group 2, 21 in Group 3, and the remaining 14 in Group 4. The relatively large number of installations which remain in Study Groups 2 and 3 reflects the general lack of water quality data needed to establish the existence or nonexistence of potential impact from known pollutant sources on those installations.

GENERAL FINDINGS

Several major findings emerged from the study. DoD installations, singly or in aggregate, do not appear to be major contributors to far-field, long-term trends of declining environmental integrity of the Bay system. Nevertheless, DoD recognizes that more careful manangement of all lands adjacent to the estuary is required to reverse past declines in the Bay's quality.

In terms of conventional pollutants, it appears that military installations contribute relatively insignificant loadings of both point and nonpoint source pollutants to the Bay and its tributaries. Significant reductions in DoD pollutant sources have been achieved over the past several years. With several exceptions, the region of influence of military activities appears to be limited to the immediate vicinity of each installation.

Areas of ongoing concern relate primarily to activities that are difficult to control or regulate, such as stormwater runoff and abandoned hazardous waste disposal sites. In particular, the discharge of toxics from poorly defined point and nonpoint sources is potentially the most important issue related to preservation of water quality on or near military installations in the Bay area. Certain toxic constituents are of special concern due to their tendency to adsorb to sediment and to accumulate in the estuarine sediment bed. Although limited, preliminary data on toxics contamination have become available at many installations as part of the Defense Environmental Restoration Program, results are generally inconclusive with respect to assessing the need, if any, for specific controls or cleanup of toxic pollutant sources.

GENERAL RECOMMENDATIONS

Long-Term Monitoring Needs

The study concluded that the control of toxics and nutrients from poorly defined point and nonpoint sources is the most important issue related to the preservation of local receiving water quality near military installations. But, there is a lack of data at many installations to adequately quantify discharge characteristics, levels of impact, and required controls on such discharges. For these installations, a long-term monitoring program is recommended for toxics in sewage or industrial waste treatment plant effluent, toxics in intermittent storm water drainage, and field monitoring for conventionals and toxics in immediately adjacent receiving water and sediments. Although not currently required at many sites, recent experience suggests many NPDES permit requirements will be upgraded to include monitoring for toxics.

Nonpoint Source Runoff Control

Evidence of nonpoint source contributions such as erosion, sediment runoff, and stormwater discharges was found at a number of military installations. While some have begun actions to address these problems, their effectiveness in controlling nonpoint source runoff is uncertain. A systematic evaluation of nonpoint pollution sources would provide the necessary information to develop comprehensive action plans to reduce these problems.

Hazardous/Toxic Materials

Recommendations for hazardous waste management improvements focus on the implementation of and strict adherence to pertinent Resource Conservation and Recovery Act (RCRA) regulations. Recurring problems include provision of adequate conforming storage areas; prompt submittal of all paperwork requirements, including permits and manifests; and timely removal of materials from short-term storage.

Additional Recommendations

Other recommendations are less universal, focusing on problem areas common to smaller numbers of installations. Included are the proper operation and maintenance of wastewater treatment plants, improved retention of environmental personnel, increased participation by tenant organizations in installation environmental programs, and enhancing DoD's role in the Bay's restoration and protection plan.

CURRENT DOD INITIATIVES

DoD is taking a number of steps to implement the recommendations in this study, and to meet the commitments made in the 1987 Chesapeake Bay Agreement. Particular attention is being focused on four areas:

- o Communicating study results both internally and to the general public.
- o Updating the Joint Resolution between DoD and EPA.
- o Helping develop the Federal Facilities Plan and the Coordinated Work Plan required by the 1987 Bay Agreement.
- o Inventorying public access opportunities.

Results of the study are being closely evaluated, and hard decisions are being made as to where to allocate scarce resources. Through efforts such as this detailed water quality assessment, DoD will continue to work to help restore the Bay and its resources.

ACKNOWLEDGMENTS

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REFERENCES

Tetra Tech, Inc., 1986. Water Quality Assessment of DoD Installations/Facilities in the Chesapeake Bay Region: Phase I Report - Data Gathering and Installation Screening.

Tetra Tech, Inc., 1987. Water Quality Assessment of DoD Installations/Facilities in the Chesapeake Bay Region: Phase II Report - Development and Testing of Installation Assessment Methodology.

Tetra Tech, Inc., 1988. Water Quality Assessment of DoD Installations/Facilities in the Chesapeake Bay Region: Phase III Report - Overall Approach, Findings and Recommendations.

Understanding the Estuary: Advances in Chesapeake Bay Research. Proceedings of a Conference. 29-31 March 1988. Baltimore, Maryland. Copyright by Chesapeake Research Consortium 1988.

Some Histologic Gill Lessions of Several Estuarine Finfishes Related to Exposure to Contaminated Sediments: A Preliminary Report

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ABSTRACT

Collections were made during 1983, '84 and '85 in the Elizabeth River, whose sediments are heavily contaminated with PAHs, heavy metals and other anthropogenic materials. Comparison samples were from the "cleaner" Nansemond River, another subestuary feeding into Hampton Roads (the lower James River) nearby. Most samples from all stations included three transient quasi-catadromous nektonic sciaenids, Atlantic croaker (Micropogonias undulatus), spot (Leiostomus xanthurus) and weakfish (Cynoscion regalis), and two endemic estuarine benthic fishes, hogchoker (Trinectes maculatus) and oyster toadfish (Opsanus tau).

Processed gills of all species exhibited microscopic lesions of four general types or categories; Ballooning Dilatations (BD) or lamellar hyperemia; Hypertrophy (HPT) of filament and lamellar cells; Hyperplasia (HPL) of filament and lamellar cells; and, Growth Deformities (GD) of gill arches, filaments and lamellae. Various subtypes occurred within each category of lesion.

Occurring in individuals from both subestuaries, highest prevalences and/or severities, most often both, were in samples from the Elizabeth. Within the Elizabeth samples, occurrence and/or severity generally were higher (usually highest) in those individuals from ER Station 7, where sediments are most heavily contaminated by PAHs, and ER Upriver than at the ER Downriver station. Differences in occurrence and/or severity of lesions in the several species were also seen. Variations in exposure to the sedimentary toxicants and susceptibility to toxic insult are most probable causes.

INTRODUCTION

The Elizabeth River, a subestuary of the James River (Fig. 1), is heavily contaminated by industrial, domestic, urban, and agricultural wastes. Sediments of certain sites carry especially heavy burdens of

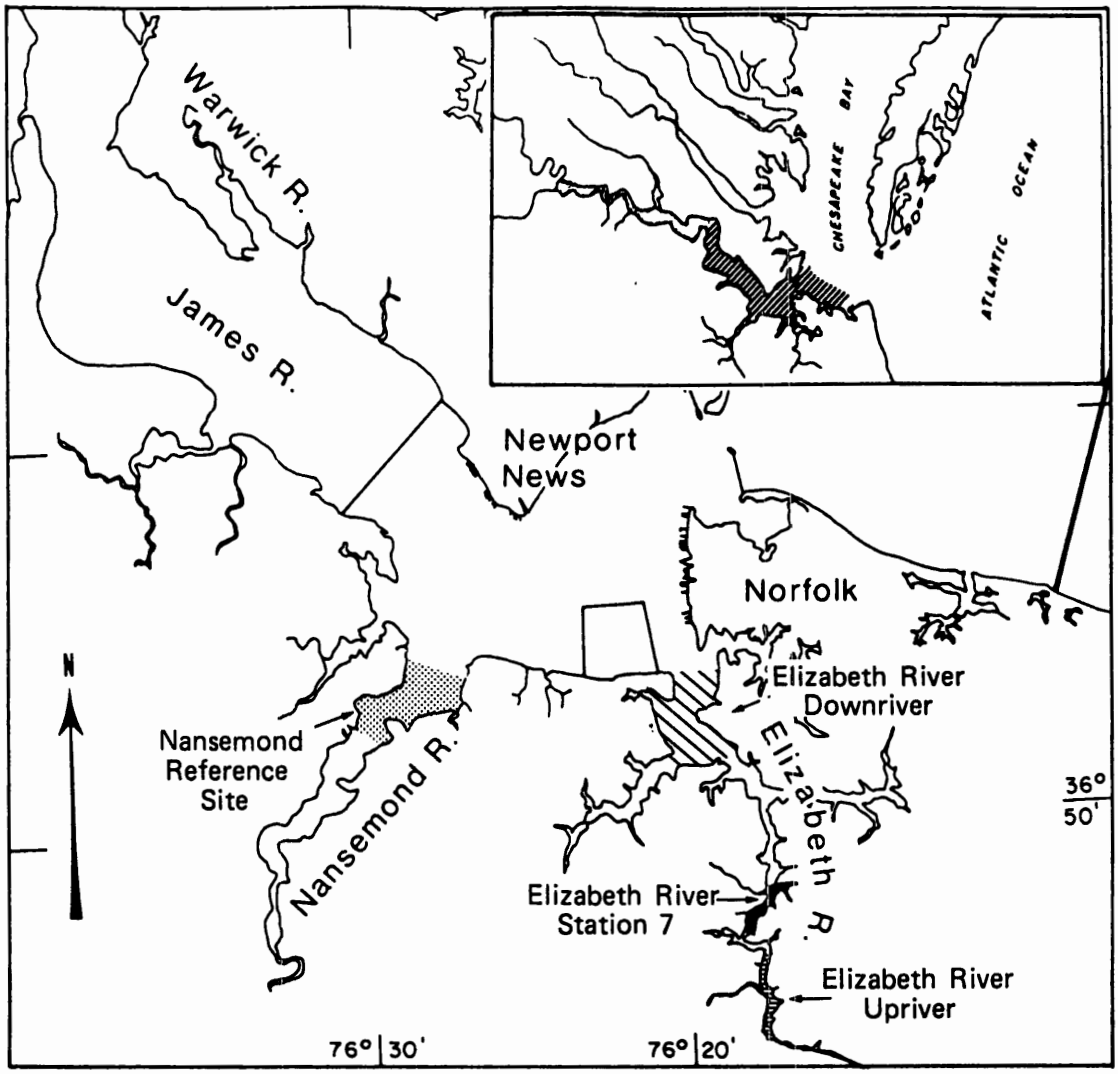

FIGURE 1
SAMPLING SITES
ELIZABETH RIVER, CONTAMINATED
NANSEMOND RIVER (REFERENCE) 'CLEANER'
(Insert shows relative position of enlarged area.)

PAHs (up to 3900 ppm for example, Hargis et al. 1984). At others, especially near shipbuilding and ship repair yards, heavy metals abound. Numerous sewage and surface drainage outfalls make point source contributions, and non-point source effluvia enter all along.

Because of the extensive lockage and the freshwater drainage (or water supply) canal system of the Intracoastal Waterway at the upper, or southern, end of the Elizabeth River, fishes migrating in from higher salinity waters of the Chesapeake Bay and the adjacent Atlantic must enter at its mouth downriver at the northern end, (see Fig. 1). Therefore, oceanic or Bay migrants collected at ER Upriver had to have entered from Hampton Roads at the River's mouth and passed through ER Downriver and ER Sta. 7 and been exposed to the Elizabeth's waters, suspended or emulsified materials, and surficial sediments along the way.

To determine possible effects of contaminated sediments and sediment-exposed estuarine waters on health of feral fishes we have studied the gross (ulcerations, externally-visible cataracts and fin rot; Hargis and Colvocoresses 1986 and Huggett et al. 1987) and microscopic eye pathology (Hargis and Zwerner, 1988) of several marine/estuarine and estuarine-endemic finfishes collected from the Elizabeth and Nansemond Rivers. This preliminary report deals with the histopathology and features of the occurrence and severity of microscopic lesions in gill materials from some of those collections.

MATERIALS AND METHODS

From 1983 to 1985 standardized trawl samples from a series of upriver-downriver stations in the Elizabeth yielded over 70,000 individuals of 8 species. Comparison collections came from the nearby "cleaner" Nansemond. All were examined grossly. Five species, the hogchoker (Soleidae), the oyster toadfish (Batrachoididae) and the spot, Atlantic croaker and weakfish (all of the family Sciaenidae), were selected for histopathological examination because of their availability, presumed different susceptibilities to toxic damage and ecological similarities and differences.

Subsamples of individuals bearing externally-visible lesions (most from the Elizabeth River as it turned out) were selected where available. Animals were killed by severing the spinal column just behind the head, necropsied and organs fixed in Dietrich's AFA and 10% NBF. Lesion-bearing individuals were chosen to determine if externally-visible lesions would be accompanied by internal ones and to increase the chances of finding internal lesions. Tissues collected routinely included areas of skin containing visible lesions, eyes, gills, livers, kidneys and intestines. Paraffin-embedded materials sectioned at 5-6 µm and stained in H&E were examined microscopically. Lesions were identified to subtype, enumerated, graded on a scale of increasing severity ranging from 1 to 5, recorded and representative microphotographs taken. Subtypes of lesions were grouped into categories based upon their basic affinities.

Occurrence data were based upon presence or absence of a particular lesion category in individuals. Mean severities presented were based

upon simple addition of the severity determinations for each subtype of lesion within each particular lesion category with the resultant sum divided by the number of individuals recorded as exhibiting that lesion category. In devising the rankings of percentage occurrence presented below differences of several tenths of a percentage point (i.e. = to or < 0.3%) were deemed insignificant and the data points involved to be essentially equal (i.e. ∿): For the mean severity rankings several hundredths (i.e. = to or < 0.03) was the criterion employed.

Microscopically-observed gill lesions were grouped into four primary categories; 1) Ballooning Dilatation (BD) (of lamellae), 2) hypertrophic responses (Hypertrophy - HPT), 3) hyperplastic growths (Hyperplasia - HPL), and 4) growth aberrations involving significant portions of filaments and/or arches, and lamellae collectively termed Growth Deformities (GD). All categories had several subtypes of lesions. Each subtype included some lesions which presented somewhat different appearances. Results on the occurrence and severity of the four types of general lesion categories are presented in Table I, Parts A and B, by species and by station respectively. Appearance of a normal gill is pictured in Figure 2.

RESULTS AND DISCUSSION

This report treats histologic lesions in the gill materials from five species taken in the 1983, '84 and '85 collections. Details are as follows:

Ballooning Dilatation (BD) (Figs. 3 and 8), or abnormal collection of blood in the lamellae (usually outside of the capillaries), called telangiectasia by some fish pathologists, occurred in 214/241 (88.8%) of all individuals and in all species. Ranked percentage occurrence of BD by species was; Hogchoker, 52.0% < Toadfish, 80.6% < Spot, 92.6% < Weakfish, 95.5% < Croaker, 98.3%: Ranked mean severity was; Toadfish, 1.40 < Hogchoker, 1.62 < Spot, 1.73 < Croaker, 1.98 < Weakfish, 2.17. Even with this ubiquitous lesion a larger proportion of each of the sciaenids had ballooning dilatations than the hogchoker and toadfish and were more severely affected.

Of the 214 cases of BD, 22, or 10.3%, were from Nansemond fishes; 192, or 89.7%, came from the Elizabeth River. Of the Nansemond fishes alone BD occurred in 22 of 27 individuals, or 81.5%; Elizabeth's were 192 of 214, or 89.7%. Ballooning Dilatation occurred at all stations and its percentage occurrence by station was closely grouped from 81.5% in the "reference" Nansemond to 90.6% at the ER Upriver site. The percentage occurrence ranking was Nansemond, 81.5% < ER Downriver, 89.0% < ER Sta. 7, 89.7% < ER Upriver, 90.6%. Ranked mean severity by station was; Nansemond, 1.41 < ER Downriver, 1.75 < ER Upriver 2.04 ∿ ER Sta. 7, 2.05. Clearly BD, though found in fishes from all stations, occurred least in the "cleaner" Nansemond, and most in the more polluted Elizabeth: Likewise, severity was lowest in the Nansemond and highest in the Elizabeth. Within the Elizabeth, percentage occurrence and mean severity were lowest at ER Downriver and highest at the upstream stations, ER Sta. 7 and ER Upriver. Occurrences and severities at ER Sta. 7 and ER Upriver were quite close or equal, respectively.

TABLE 1

Occurrence and Severity of Gill Lesions by Species and by Station

[Symbols: No. = Total Individuals; Occ. = No. with Lesions (Occurrence); % O. = % Occurrence; $\bar{X}$ Sev. = Mean Severity; BD = Ballooning Dilatation, HPT = Hypertrophy, HPL = Hyperplasia, GD = Growth Deformities. Calculations of mean severity involve only affected animals, no negatives included.]

			BD			HPT			HPL			GD		
A.	SPECIES	No.	Occ.	% O.	$\bar{X}$ Sev.	Occ.	% O.	$\bar{X}$ Sev.	Occ.	% O.	$\bar{X}$ Sev.	Occ.	% O.	$\bar{X}$ Sev.
	Weak.	67	64	95.5	2.17	36	53.7	1.79	50	74.6	3.33	14	20.9	1.07
	Croak.	59	58	98.3	1.98	28	47.5	1.95	45	76.3	2.70	10	17.0	1.05
	Spot	54	50	92.6	1.73	15	27.8	1.97	44	81.5	2.20	9	16.7	0.89
	Toadf.	36	29	80.6	1.40	20	55.6	1.65	16	44.4	1.54	26	72.2	1.98
	Hogch.	25	13	52.0	1.62	17	68.0	2.35	10	40.0	2.80	1	4.0	1.00*
	Totals	241	214	88.8		116	48.1		165	68.5		60	24.9	

			BD			HPT			HPL			GD		
B.	STATION	No.	Occ.	% O.	$\bar{X}$ Sev.	Occ.	% O.	$\bar{X}$ Sev.	Occ.	% O.	$\bar{X}$ Sev.	Occ.	% O.	$\bar{X}$ Sev.
	Nanse.	27	22	81.5	1.41	9	33.3	1.12	19	70.4	2.38	1	3.7	1.00**
	ER Down.	82	73	89.0	1.75	36	43.9	1.79	60	73.2	2.26	17	20.7	1.87
	ER Sta. 7	68	61	89.7	2.05	39	57.4	1.91	41	60.3	3.51	24	35.3	1.42
	ER Up.	64	58	90.6	2.04	32	50.0	2.26	45	70.3	2.50	18	28.1	1.06
		241	214	88.8		116	48.1		165	68.5		60	24.9	

* 1 case only. ** Also 1 case only; further, GD noted involved lamellae only and not filaments.

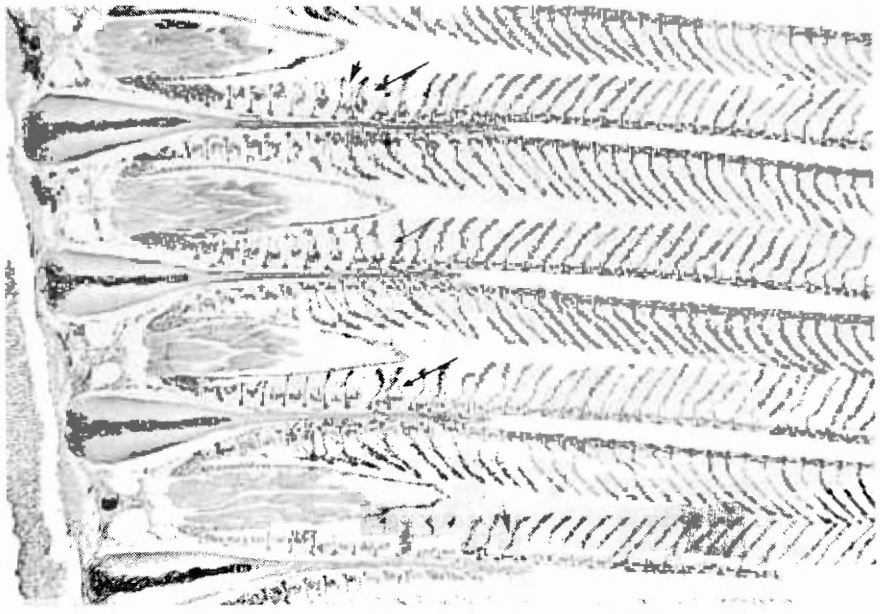

FIGURE 2
Weakfish, *C. regalis,* from Nansemond River
showing appearance of "normal" filaments and lamellae.
Dietrich's AFA fixative often causes lamellar
delamination (Arrows)!
Dietrich's, H&E, Ca. 100X

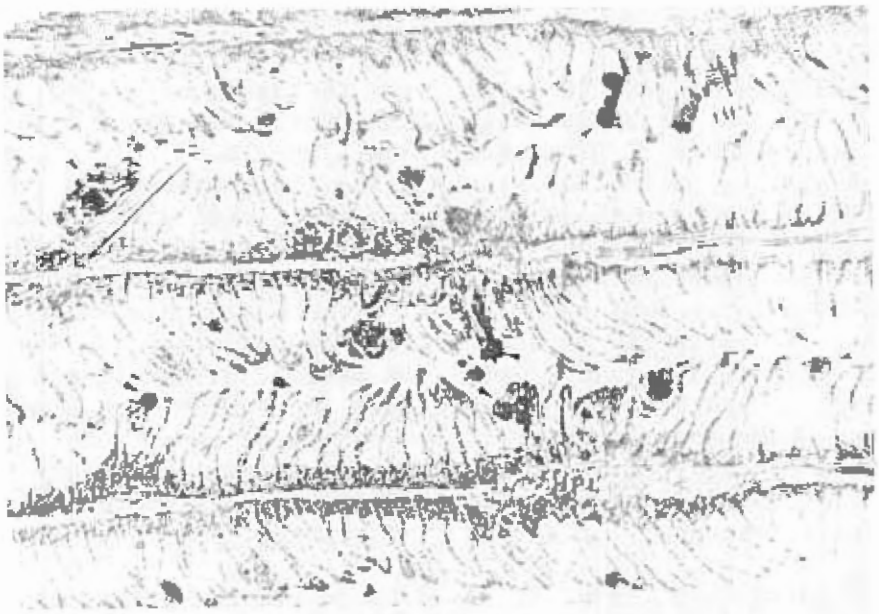

FIGURE 3
Weakfish, *C. regalis,* from ER Sta. 7 showing
ballooning dilatation **(BD)** (Arrows). Hyperplasia **(HPL)** of
the basal interlamellar type also present.
Dietrich's, H&E, Ca. 100X

Hypertrophy (HPT) (Fig. 4), abnormally enlarged cells of soft tissues of the lamellae and filaments, occurred in 116/241 (48.1%) of all individuals and in all species. Ranked percentage occurrence of HPT by species was; Spot, 27.8% < Croaker, 47.5% < Weakfish, 53.7% < Toadfish, 55.6% < Hogchoker, 68.0%. Mean severity was; Toadfish, 1.65 < Weakfish, 1.79 < Croaker, 1.95 ∿ Spot, 1.97 < Hogchoker, 2.35. Hypertrophy occurred more frequently in the hogchoker than in the toadfish and the sciaenids. Its severity was greatest in hogchoker as well. Percentage occurrence in toadfish was greater than in the spot and other sciaenids, while mean severity in toadfish was lower than in all other species.

Of the 116 cases of HPT, 9, or 7.8%, were in Nansemond fishes while 107, or 92.2%, occurred in Elizabeth River fishes. Of the Nansemond River fishes alone, 9 of 27, or 33.3%, were affected, while 107 of 214, or 50.0%, of those from the Elizabeth had HPT. Hypertrophy occurred at all stations and its percentage occurrence by station was; Nansemond, 33.3% < ER Downriver, 43.9% < ER Upriver, 50.0%, < ER Sta. 7, 57.4%: Mean severity was; Nansemond, 1.12 < ER Downriver, 1.79 < ER Sta. 7, 1.91 < ER Upriver, 2.26. Though HPT occurred at all stations both occurrence and severity were less in the Nansemond and at ER Downriver than at the upstream stations, ER Sta. 7 and ER Upriver.

Hyperplasia (HPL) (Figs. 3, 5 and 6), abnormally large numbers of cells of soft tissues of the lamellae and filaments, occurred in 165/241 (68.5%) of individuals collected and in all species. Ranked percentage occurrence of HPL by species was; Hogchoker, 40.0% < Toadfish, 44.4% < Weakfish, 74.6% < Croaker, 76.3% < Spot, 81.5%: Mean severity was; Toadfish, 1.54 < Spot. 2.20 < Croaker, 2.70 < Hogchoker, 2.80 < Weakfish, 3.33.

Of all 165 HPL cases, 19, or 11.5%, were in Nansemond fishes, while 146, or 88.5%, came from the Elizabeth. Of the Nansemond River fishes alone HPL occurred in 19 of 27 individuals, or 70.4%: In the Elizabeth it was 146 of 214, or 68.2%. Hyperplastic lesions occurred at all stations and their percentage occurrence by station was; ER Sta. 7, 60.3% < ER Upriver, 70.3% ∿ Nansemond, 70.4% < ER Downriver 73.2%. ER Upriver and Nansemond were essentially equal. That its occurrence was lowest at heavily contaminated ER Sta. 7 instead of highest seems anomalous. Mean severity by station was; ER Downriver, 2.26 < Nansemond, 2.38 < ER Upriver, 2.50 < ER Sta. 7, 3.51. Severity indices of ER Downriver and Nansemond were close: ER Upriver was somewhat higher than Nansemond and ER Downriver, while ER Sta. 7 was considerably higher. Thus, the anomaly of a lower percentage occurrence at heavily-contaminated ER Sta. 7 was balanced somewhat by the far greater severity of HPL at that station.

Growth Deformity (GD) (Figs. 7, 8 and 9), which can appear as branched lamellae, deformed or forked filaments or deformed arches, with one or more in the same gill, occurred in 60/241 (24.9%) of all individuals collected and in all species. In some cases of deformed arches filament forking occurred very close to the arches, themselves. A majority of gill filaments affected by GD were well-formed otherwise and appeared functional. Percentage occurrence by species was; Hogchoker, 4.0% < Spot, 16.7% ∿ Croaker, 17.0% < Weakfish, 20.9% <

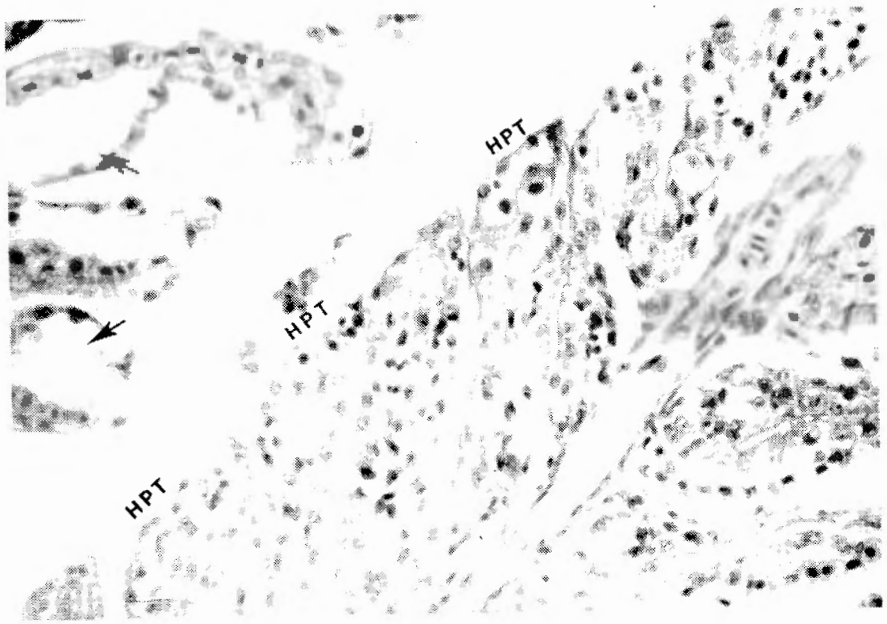

FIGURE 4
Oyster Toadfish, *O. tau,* from ER Upriver showing hypertrophy **(HPT)** of lamellae. Fixation delamination (Arrows). Dietrich's, H&E, Ca. 500X

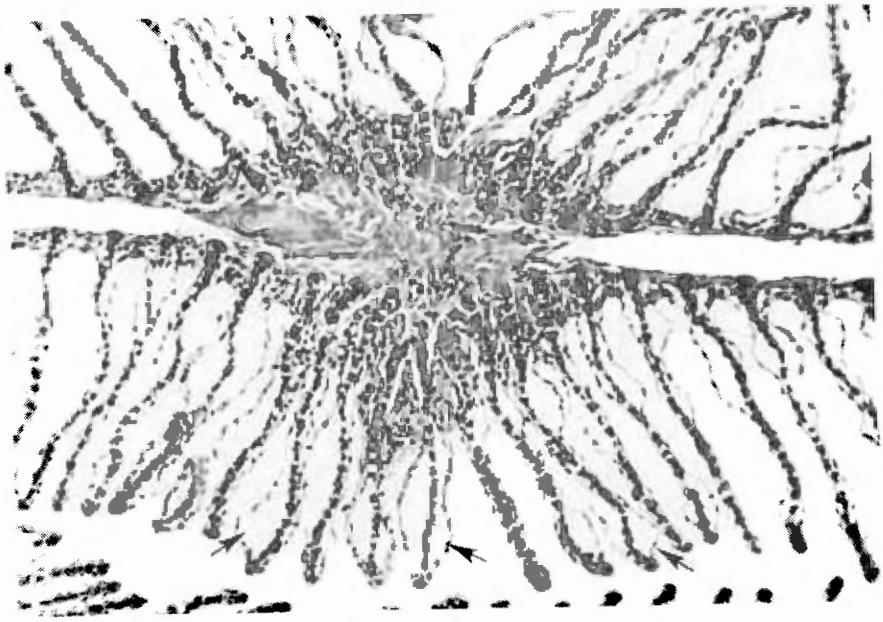

FIGURE 5
Weakfish, *C. regalis,* from ER Upriver showing the peculiar "medusa-shaped" interlamellar hyperplastic **(HPL)** lesion frequently seen in this species. Fixation delamination (Arrows). Dietrich's, H&E, Ca. 300X

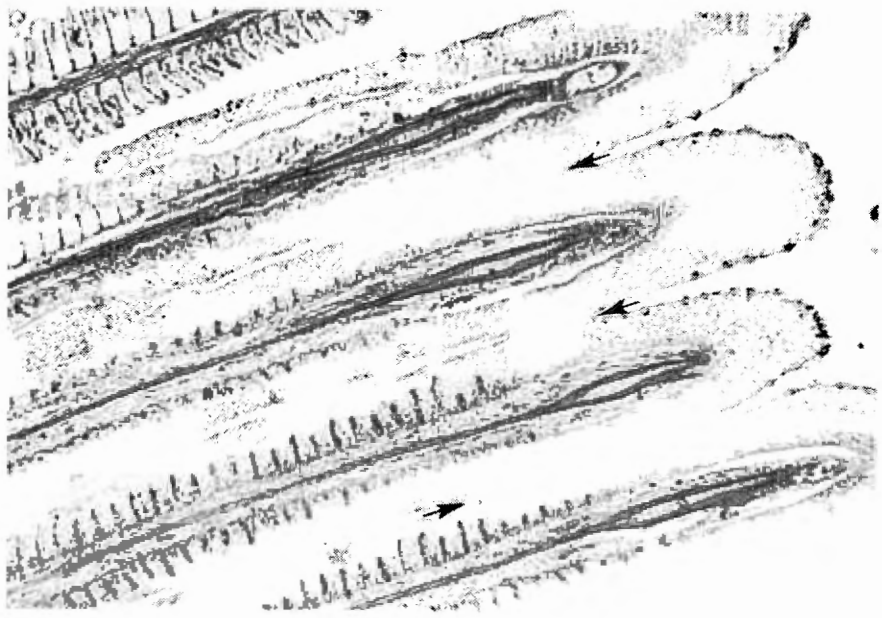

FIGURE 6
Croaker, *M. undulatus,* showing hyperplasia **(HPL)** with spongiosis. Note marked overgrowth of lamellae and filaments and interlamellar hyperplastic growth (fusion). See also "bridges" between filaments in lesion (Arrows). Dietrich's, H&E, Ca. 225X

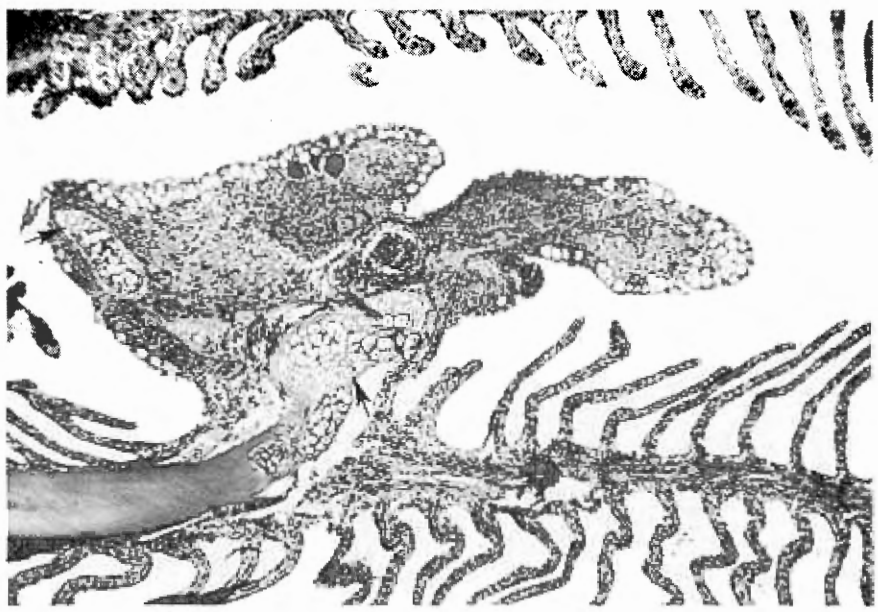

FIGURE 7
Toadfish, *O. tau,* from ER Sta. 7 showing gnarled growths on a filament exhibiting growth deformity **(GD).** Note cartilaginous deformities (Arrows). Dietrich's, H&E, Ca. 100X

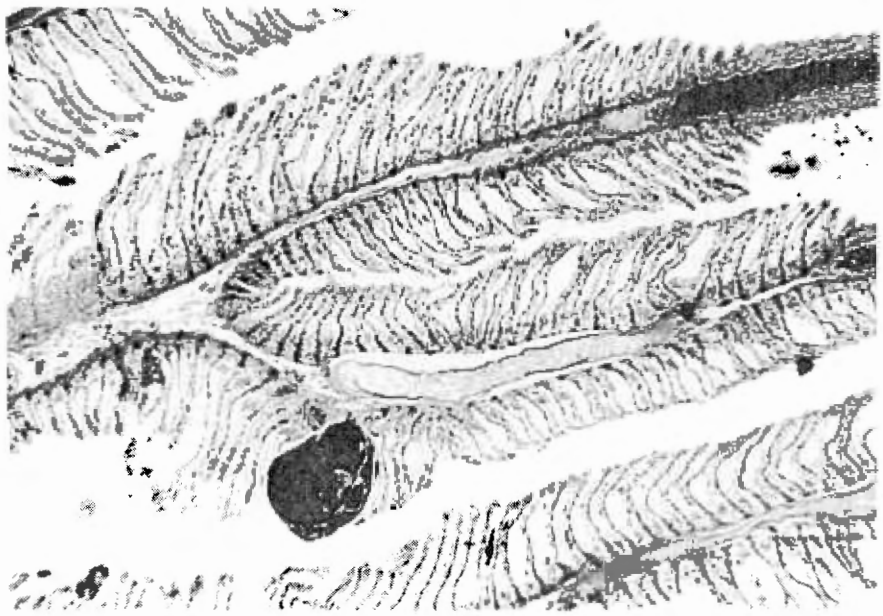

FIGURE 8
Weakfish, *C. regalis,* showing the growth deformity **(GD)** forked or branched filament. A ballooning dilatation (BD) also appears.
Dietrich's, H&E, Ca. 125X

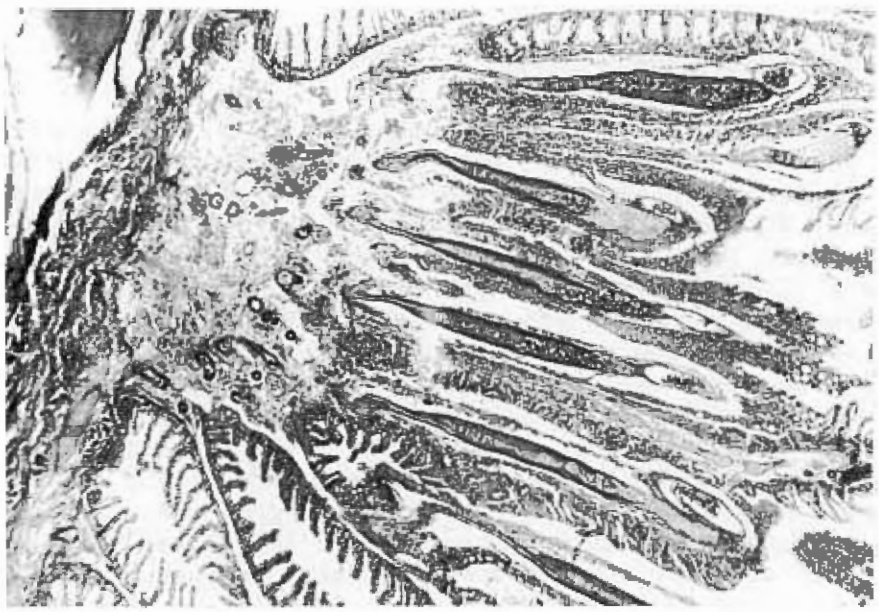

FIGURE 9
Croaker, *M. undulatus,* from ER Upriver showing growth deformity **(GD)** at bases of several adjacent filaments.
Dietrich's, H&E, Ca. 100X

Toadfish, 72.2%: Mean severity was; Spot, 0.89 < Croaker, 1.05 ∿ Weakfish, 1.07 < Toadfish, 1.98. (Hogchoker is not included in the mean severity ranking because only a single individual was involved and no average was possible). Toadfish had larger numbers and more severe GD than all other species. Within the sciaenids, weakfish had more and more severe GD lesions than the other two species. Croaker GD, though essentially equal in occurrence to that in spot, was somewhat greater in severity. All sciaenids were fairly close in occurrence and severity of this lesion.

Of the 60 cases of GD the Nansemond had 1, or 1.7%, while the Elizabeth had 59, or 28.3%. Of the Nansemond River fishes alone only 1 of 27, or 3.7%, had GD, while 59 of 214 of the Elizabeth River fishes, or 27.6%, were affected. Percentage occurrence of GD by station was; Nansemond, 3.7% < ER Downriver, 20.7% < ER Upriver, 28.1% < ER Sta. 7, 35.3%. Mean severity by station was; ER Upriver, 1.06 < ER Sta. 7, 1.42 < ER Downriver, 1.87. [As indicated, only 1 fish in the Nansemond was affected by a subtype of this lesion category. It was rated 1.0 in severity. Because 1) it was the only Nansemond individual with GD and 2) only the "soft" lamellae, as opposed to the cartilaginous or bony elements of the filaments and arches of most other cases, were involved it is accorded little weight in these comparisons.] Essentially, Nansemond exhibited no GD lesions which were comparable to those in most Elizabeth fishes. Within the Elizabeth, occurrence was lowest at ER Downriver and highest at ER Sta. 7. That mean severity was higher at ER Downriver (1.87) than at ER Sta. 7 (1.42) and ER Upriver (1.06) was due to the fact that a single affected individual at ER Downriver showed a uniquely high mean severity datum of 8.0 making the calculated mean very high in comparison with the mean severities of GD observed at other stations.

DISCUSSION AND CONCLUSIONS

1. Microscopic studies of gills from 241 individuals of 5 species of fishes revealed lesions of 4 categories; Ballooning Dilatation, Hypertrophy, Hyperplasia and Growth Deformities. All species were affected. The first three histologic gill lesions (BD, HPT and HPL) occurred in both subestuaries, the highly-contaminated Elizabeth River and the "cleaner" Nansemond. The single GD seen in the Nansemond River involved the soft lamellae alone while most others of this lesion category involved the bony or cartilagenous support elements of filaments and/or arches. Consequently, the Nansemond fishes could be considered essentially GD-free. There were differences in occurrence and severity of all four lesions (BD, HPT, HPL and GD) between rivers, stations of the Elizabeth and species also, apparently reflecting the geographical variations in levels of chronic toxicity of sediment-borne contaminants. These differences also reflect variations in time and nature of exposure to the toxicants and susceptibility of the fishes to toxic insult by biologically active contaminants. Ballooning dilatation (BD), the most ubiquitous lesion probably was brought about by handling in the most severe cases, as well as by toxic stress. HPT, HPL and GD are all clearly related to chronic toxicity and, perhaps, other long-term environmental stressors and not to handling.

2. Though exceptions (explainable) occurred, several general patterns emerged:

   a. Ballooning dilatations (BD) were the most numerous of the four lesions: Growth Deformities (GD) were the least. The ranked sequence of mean occurrence of lesion categories was GD, 24.9% < HPT 48.1% < HPL, 68.5% < BD, 88.8% (Table I, Part A).

   b. A lower percentage of lesions of all types occurred in the Nansemond than the Elizabeth (Table I, Part B). Severity was lower in the Nansemond River fishes as well. Undoubtedly, this reflects the more polluted condition of the sediments and waters of the Elizabeth and, probably, the greater toxicity of the contaminants involved as well. At ER Sta. 7 PAHs related to creosote and other petroleum hydrocarbons predominate and are most likely candidates as primary contributors to occurrence and severities of the microscopic gill lesions seen in our collections, especially those taken at the two upstream stations, ER Sta. 7 and ER Upriver.

      Within the Elizabeth there were fewer and less severe gill lesions at the ER Downriver station, with the single exception of GD. Usually, lesions were highest at the upstream locations, ER Sta. 7 where sediments are most heavily contaminated by PAH's, among other pollutants, or ER Upriver. Between ER Sta. 7 and ER Upriver occurrence and severity of BD were essentially equal: Occurrence and severity of GD was greater at ER Sta. 7 than at E.R. Upriver. With HPT and HPL the relative position of one or the other of these lesions at the same two upstream stations was reversed. Significance of these last findings is not clear at this point.

      Basically the patterns of general occurrence of microscopic lesions of gills agreed with those of earlier studies which employed the occurrence of gross lesions as a basis for analysis of the possible effects of contaminated sediments on the health of finfishes in the Elizabeth River, between stations in the Elizabeth and between the Elizabeth and Nansemond Rivers (Hargis and Colvocoresses 1986 and Huggett et al. 1987). Gross lesions were virtually non-existent in the Nansemond samples. Within the Elizabeth, where the largest number of both gross and microscopic gill lesions occurred, fewer were downstream than upstream generally. Both were usually more prevalent (occurrence) or severe at ER. Sta. 7 and/or ER Upriver, the upstream stations.

   c. Of the 5 species of finfishes studied, the most "resistant" to toxic effects from contaminants in the Elizabeth (and Nansemond) in terms of occurrence and/or severity of microscopic lesions of the gills were the more-or-less endemic bottom-dwelling hogchoker and toadfish, usually in that order (Table I, Part A). Generally the least resistant were the more pelagic, ocean-spawning, migratory sciaenids. One exception was Hypertrophy (HPT), with the hogchoker highest in occurrence at 68.0% and the toadfish second at 55.6%. Comparative mean

severity of HPT was also highest in the hogchoker (2.35) but lowest in the toadfish (1.65). HPT may require close and frequent contact with the source of toxicants, the sediments, for its initiation and/or continuance. This condition would be met by hogchokers and toadfish since they live nearest the bottom and are in closest and most frequent contact with the sediments and with those waters affected most by sediment-associated toxicants. The soleid flatfish, hogchoker, usually is found on or in the bottom sediments. That HPL was more severe though not more prevalent in hogchoker than in all other species except the weakfish is probably related to the long-term and regular close contact with bottom sediments and bottom water of this benthic fish.

The other exception was Growth Deformity (GD) in which toadfish was highest by far in both percentage occurrence (72.2%) and mean severity (1.98). Since initiation of most gill GD must occur early in the development of the organ systems involved and toadfish are the single species of those studied here whose eggs, larvae and, probably, early juveniles develop closest to the bottom, it is not surprising they were most severely affected. Indeed, it would have been surprising had they not been.

d. Of the three sciaenids, weakfish generally were highest in occurrence and/or severity of lesions, while spot were least. Where reversals in either were observed (Table 1, Part A) as in HPT, spot and croaker (and in that order) were more affected than weakfish. This probably reflects their more intimate contact with the source of the toxicant, the contaminated sediments, during feeding. The greater occurrence of HPT in these two species agrees with the likelihood that a closer association with contaminated sediments is involved with this lesion as discussed above when considering its occurrence and severity in the bottom-dwelling hogchoker and toadfish.

Generally, our histopathological findings confirm shipboard and laboratory experiences that hogchoker and toadfish are most resistant to trauma or ecological insult, with spot, croaker and weakfish increasingly susceptible and in that order. Weakfish, the predator operating at the highest trophic levels of the three sciaenids, seems generally the most susceptible of all five species studied to toxic stress in relation to likelihood of direct exposure to the contaminated sediments and closely-associated benthic waters.

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LITERATURE CITED

Hargis, W. J., Jr. and J. A. Colvocoresses. 1986. EPA Workshop: Use of finfish as indicators of toxic contamination. Selected Gross Pathological Features. In Technical Resources, Inc. (Eds.). Proceedings EPA Airlie House Workshop, July 1986. Technical Resources Inc., 3202 Monroe St., Suite 300, Rockville, MD 20852, USA.

Hargis, W. J., Jr. and D. E. Zwerner. 1988. Effects of certain contaminants on eyes of several estuarine fishes. In Stegeman, J. J. and M. N. Moore (Eds.) Responses of Marine Organisms to Pollutants. Marine Environmental Research 24 (1988). Elsevier Applied Science Publishers, Ltd., Barking, England.

Hargis, W. J., Jr., M. H. Roberts, Jr. and D. E. Zwerner. 1984. Effects of contaminated sediments and sediment-exposed effluent water on an estuarine fish: Acute toxicity. Marine Environmental Research 14 (1984). Elsevier Applied Science Publishers, Ltd., Barking, England.

Huggett, R. J., M. E. Bender and M. A. Unger. 1987. Polynuclear aromatic hydrocarbons in the Elizabeth River, Virginia. In Dickson, K. L., A. W. Maki and W. Brungs (Eds.) Fate and Effects of Sediment-Bound Chemicals in Aquatic Systems. Society of Environmental Toxicology and Chemistry Special Pub. No. 2, Pergamon Press, New York, USA.

Understanding the Estuary: Advances in Chesapeake Bay Research. Proceedings of a Conference. 29-31 March 1988. Baltimore, Maryland. Chesapeake Research Consortium Publication 129. CBP/TRS 24/88.

Abstract only

Chemical and Biological Analysis of the Effluents from Oil and Water Separators in Commercial Shipyards

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Bilge waters, fuel tanks washings, waste oils, etc., are pumped into a shipyard's oil/water separator prior to areas of a ship being certified as gas-free for repairs or modifications. Frequently, these oil/water separators are simple, baffled tanks that rely on quiescence to effect a gravity separation of the oily material and the water. This sort of system provides inadequate treatment of water containing refined petroleum products, or of an oil-water mix that may contain surfactants. The Virginia Water Control Board conducted a study in October - December, 1987 of the effluents from the oil/water separators at three commercial shipyards on the Elizabeth River, in Norfolk, Virginia. The chemical characteristics of the effluents were addressed, along with their acute and chronic toxicities to the sheepshead minnow, Cyorinodon variegatus, and the mysid shrimp, Mysidoosis bahia. The 48-hour acute tests generated LC50s as low as 0.6%, and the 7-day chronic tests generated NOECs (No Observed Effect Concentration) as low as 0.01%. The toxicity of these effluents can be attributed to priority pollutant metals and volatile and extractable organic compounds, since many of these exceeded water quality criteria. This investigation demonstrates the inadequacy of oil/water separators for the treatment of this wastewater, and the importance of coordinating chemical and biological analyses to characterize potential water quality impacts from effluents.

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Abstract only

Acidic Conditions in Maryland Coastal Plain Streams: Potential Impacts on Anadromous Fish

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During the spring of 1987, a synoptic survey of headwater streams in Maryland estimated that more than one-half of the streams (2600 km) in Maryland's Coastal Plain are sensitive to acidification (i.e., have acid neutralizing capacities $\leq$ 200 µe/qL). Nearly one-quarter of these streams (1137 km) are acidic (i.e., have pH $\leq$ 6.0). Recent research has shown that streams in the Coastal Plain of Maryland also exhibit acidic pulses during precipitation events and that these conditions may affect the survival of early life stages of anadromous fish.

Laboratory studies demonstrated that the early life stages of blueback herring and American shad are very sensitive to moderate acidity and increased concentrations of inorganic aluminum. These anadromous alosid species are among the most sensitive yet studied. Fertilized alosid eggs were generally less sensitive to pH and aluminum than pre-feeding larvae. Blueback herring and American shad larvae tolerated pH 6.5, but succumbed to pH 5.7 and 6.2. The toxicity to pH to larvae was increased by simultaneous exposure to total monomeric aluminum concentrations as low as 100 µg/L for blueback herring and 15 µg/L for American shad.

A direct link between surface water acidification and mortality of alosid early life stages has not yet been convincingly established in Maryland's Coastal Plain streams. Although in-situ bioassays using blueback herring and American shad early life stages have been conducted in Lyons Creek since 1984, abnormally dry springs during this period have produced few acidic episodes in this stream that approached acutely toxic conditions. Hence, field confirmation of laboratory-derived toxicity predictions is lacking for blueback herring and American shad.

In another study, the addition of calcium carbonate has been shown to mitigate acidic pulses during precipitation events in two Maryland Coastal

Plain streams. In-situ bioassays in these streams suggests that yellow perch egg hatchability is improved in those portions of the streams where acidic conditions have been ameliorated.

The adverse impact of acidification on the reproductive success of anadromous alosids and other fish species in Chesapeake Bay will likely vary among years, among streams, and among species.

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Sediment Toxicity Testing in the Chesapeake Bay

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The process of formation of estuarine sediments tends to sorb and concentrate water-born pollutants. Concentrations of pollutants in estuarine sediments can become very large (Tsai, e.a. 1979). However, conventional chemical measurements of sediment-sorbed pollutants may not indicate their bioavailability or effects on growth, reproduction and behavior of infaunal benthic organisms. Ways to determine relationships between estuarine sediment pollution and effects on estuarine biota are undergoing evaluation.

Chapman, e.a. (1987, in press) has developed and tested the concept of a Sediment Quality Triad to assess pollution-induced degredation. This approach combines measurements of sediment chemical contamination, sediment toxicity through bioassays, and benthic infaunal community structure. The bioassays used by Chapman include burrowing behavior of a clam (Macoma balthica), a a 48-hour mussel larva development test with sediment elutriate, a 10-day amphipod survival test with a West coast amphipod, and a 4 week copepod reproduction bioassay with a West coast species. Chapman reported the mussel larva and amphipod bioassays gave the best correlation with infaunal diversity, and he was able to make some general correlations with measurements of chemical sediment contaminants.

Adapting the Sediment Quality Triad concept for the Chesapeake Bay requires the development of new bioassays with

East coast and euryhaline species. Tsai, e.a. (1979) described sediment toxicity testing for Baltimore harbor sediments using sediment suspensions with the mummichug (Fundulus heteroclitus), the spot, and the soft-shell clam (Mya arenaria). Tsai reported good negative correlation of sediment toxicity with benthic infaunal diversity, but the suspended sediment bioassay caused difficulty with all but the mummichug bioassay. The present paper describes the development and field testing of a burrowing bioassay for sediments with the soft-shell clam (Phelps, 1985, 1987) and a 96-hour sediment toxicity test with the euryhaline oyster larva, Crassostrea gigas that uses solid sediments (Phelps and Warner, 1988).

CLAM BURROWING BIOASSAY

A clam burrowing bioassay with M. arenaria was developed in the laboratory (Phelps, 1985). The bioassay was conducted by placing 20 clams on a one-liter sediment sample in a 5 inch square plastic box with 5 cm of test water, noting the number of clams completely burrowed at increasing intervals, and calculating the estimated time for 50% of clams to burrow (ET50) by Logit analysis (Phelps, e.a. 1985). Up to five sediment samples could be tested simultaneously with a control. The bioassay organisms were young M. arenaria one to two centimeters in length, obtained from Dr. Mike Castagna, Virginia Institute of Marine Science, and stored in sediment at 13 ppt salinity and 10 deg. C. up to three months before use. Experiments showed the clams should be warmed to room temperature for at least six hours before the bioassay. Clams that had been

used in bioassays causing burrowing inhibition were considered possibly impaired and not used in subsequent bioassays.

For the field testing of the bioassay, sediment samples were collected on the EPA OSV Antelope cruise 5/17-23 1985 (EPA 1985a, 1985b) starting from Annapolis, MD, collecting down the Chesapeake Bay, at ocean sites, up the Delaware Bay, through the Delaware Ship Canal and back to Annapolis, MD. At the beginning of the EPA cruise the clam burrowing bioassay was tested on sediment samples with water from the sampling sites and controls were similar to laboratory bioassays. However, with samples from higher salinities the control burrowing time increased significantly (Table 1). As the cause of the increased burrowing time was unknown, the sediment burrowing assays were postponed until they could be run under controlled laboratory conditions. The sediment samples collected during the cruise were stored at 4 - 10 deg. C.

Table 1

Bioassay Control Burrowing Speeds

Date (Ship)	Salinity (ppt)	ET50 (95% F.L.) (hrs.)
5/18	15	0.89 (.67-.84)
5/19	28	0.76 (.62-.93)
5/20	34	2.15 (1.6-3.0)
5/22	32	1.85 (1.2-2.9)
(Laboratory)		
5/26	13	0.76 (.58-112.)
5/27	13	0.51 (.44-.59)
5/27	13	0.89 (.65-1.23)
6/4	14	0.24 (.19-.32)
6/18	14	0.09 (.06-.14)

Underlined ET50 values significantly different from controls.

Immediately upon return, 5/24-27, two experiments on the effects of sediment and test water salinity on clam burrowing speed were conducted at the Chesapeake Biological Laboratory (CBL):

Experiment 1: Sediments at different salinities, clams and test water at one salinity. The test water and the clams were at the ambient salinity of the Chesapeake Biological Laboratory, 14 ppt. Sediments were adjusted to 3, 8, 13, 28 and 32 ppt. To adjust sediment salinity, CBL control (sandy) sediment was rinsed three times at a 10:1 sediment:water ratio using dilutions of artificial sea water (Instant Ocean). Standing water was poured off the sediment and test water carefully layered over the sediment before adding the clams for the burrowing assay. The results clearly showed that changes in sediment salinity for the most part did not affect burrowing speed as long as the clams had been held at the salinity of the overlying water (Table 2).

Table 2

Effect of sediment salinity on burrowing speed

Water salinity: 14 ppt
Clam acclimation: 14 ppt

Burrowing Speed	Sediment Salinity (ppt) 3	8	14	28	32
ET50 (hrs.)	.45	.30	.50	.66	<u>1.05</u>
(95% F.L.)	(.37-.55)	(.23-.38)	(.43-.57)	(.53-.80)	(.87-1.25)

Underlined ET50 value significantly different from control

Experiment 2: Sediment at one salinity, clams acclimated for various times to test water of different salinities. The sediment was CBL control (sandy) at 13 ppt. Clams were

acclimated in artificial test water (Instant Ocean) at 3, 8, 13, 18, 22 and 32 ppt. and tested after 6, 12, 24, 30 and 48 hours acclimation. Clam burrowing speed was significantly slowed by exposure to test water salinity different from acclimation salinity up to a 48 hour acclimation period (Table 3). Test water salinity lower than acclimation salinity caused the greatest burrowing inhibition.

Table 3

Sediment salinity constant at 13 ppt:
Clams acclimated and tested at differing water salinities.

Accl. Time (hrs.)	ET50 (95% F.L.) (hrs.) Water and Clam Salinity (ppt) 3	8	13(control)	18	22	32
6	//	37 (.45-3037)	0.5 (.43-.57)	0.71 (.58-.87)	1.07 (.83-1.38)	
12	//	20 (.28-1380)	0.6 (.53-.69)	1.10 (.90-1.34)	0.81 (.67-.90)	
18	//	6 (3.0-11.2)	0.8 (.63-1.08)	1.8 (1.2-2.9)	2.0 (1.2-3.1)	26 (4.9-141.)
30	//	11 (4.4-31.)	1.1 (.86-1.5)	2.9 (2.1-4.0)	1.9 (1.4-2.6)	2.7 (2.0-3.7)
48	//	2.5 (2.0-3.2)	0.97 (.71-1.3)	5.4 (0.0-30)	3.2 (2.1-5.2)	2.4 (1.8-3.1)

// no burrowing observed
Underlined ET50 values are significantly different from controls

As a result of the laboratory experiments, all the sediments were immediately tested under laboratory conditions (5/25-27) with test water and clams at the control salinity, 13 ppt. Eleven out of 39 sediments (28%) showed significant inhibition of clam burrowing speed.

Table 4

Sediment Samples: Clam Burrowing Speeds (hrs)

Station	EPA No.	Salinity (ppt)	ET50 ship (95% F.L.)	ET50 laboratory (95% F.L)
(Chesapeake Bay Samples)				
4.1W	008	13	1.8 (1.4-2.3)	1.1 (.84-1.4)
4.3E	009	18		0.71 (.53-.94)
4.3C	010	16		1.26 (.76-2.1)
4.3W	011	13		0.65 (.55-.76)
5.1C	012	18		0.66 (.56-.76)
5.1W	013	16		0.39 (.34-.46)
5.3E	014	17		0.86 (.71-1.04)
5.3C	015	20		0.61 (.51-.74)
5.3W	016	17		0.84 (.71-1.0)
6.3C	017	26	0.95 (.69-1.3)	3.0 (2.5-3.5)
7.2	018	28		0.60 (.47-1.3)
7.2E	019	26	0.49 (.39-.63)	1.13 (.85-1.51)
7.4N	020	33 (est.)		0.64 (.53-.78)
7.4	021	33	0.43 (.33-.55)	0.43 (.33-.55)
LB5	023	32		0.96 (.81-1.1)
LB6	024	32	4.7 (3.2-6.8)	7.4 (5.4-10.0)
(Ocean Samples)				
A010	--	33		0.41 (.34-.50)
NB1	025	34	0.72 (.57-.91)	0.95 (.85-1.12)
DN2	--	32	0.46 (.38-.55)	0.78 (.65-.92)
206	--	32	0.75 (.55-1.06)	1.2 (1.0-1.5)
(Delaware Bay Samples)				
DB2E	026	30	0.49 (.40-.60)	0.65 (.59-.76)
DB2C	027	30	1.5 (1.2-2.3)	0.96 (.76-1.2)
DB2W	028	29	0.85 (.71-1.0)	0.53 (.41-.69)
DB3E	029	23	0.82 (.63-1.0)	0.60 (.46-.81)
DB3C	030	24	0.51 (.42-.63)	0.78 (.69-.88)
DB3W	031	27	0.84 (.69-1.0)	1.4 (1.2-1.7)
DB4E	--	16		1.2 (1.2-1.4)
DB4W	--	17		1.1 (.84-1.6)
DB5E	--	9		1.4 (1.2-1.8)
DB5C	--	9		0.80 (.66-.97)
DB5W	--	7		3.5 (2.5-5.0)
DB6E	--	2		2.1 (1.7-2.6)
DB6W	--	2		5.6 (4.2-7.4)
(Mid-Chesapeake Bay Samples)				
MCB2.1	--	1		2.5 (1.9-3.3)
MCB2.2	--	3		1.2 (.035-1.47)
MCB3.2C	--	19		0.43 (.36-.51)
MCB3.2W	--	13		0.59 (.50-.70)
MCB3.3E	--			0.20 (.10-.38)
MCB3.3C	--			0.24 (.19-.32)

Underlined values are significantly different from controls

EPA sediment analysis was available for four of the eleven samples causing burrowing inhibition (EPA 1985c) (Table 5).

Table 5

Chemical and Physical Analysis of Sediment Samples

Station	EPA No.	ET50 (95% F.L.)	% Solids	Cd ppm	Cr ppm	Cu ppm	Pb ppm	Ni ppm	Zn ppm
(Chesapeake Bay Samples)									
4.1W	008	1.1 (.84-1.4)	3.7	54	485	362	540	460	2840
4.3E	009	0.71 (.53-.94)	43.9	4.4	20	14	11	15	85
4.3C	010	1.26 (.76-2.1)	23.6	8.5	52	34	41	39	164
4.3W	011	0.65 (.55-.76)	62.7	3.2	32	13	18	35	106
5.1C	012	0.66 (.56-.76)	64.1	3.1	42	26	28	36	134
5.1W	013	0.39 (.34-.46)	53.0	3.8	15	11	15	10	62
5.3E	014	0.86 (.71-1.04)	85.1	2.4	2	2	3	7	5
5.3C	015	0.61 (.51-.74)	26.2	7.6	49	27	27	31	148
5.3W	016	0.84 (.71-1.0)	33.3	6.0	36	25	25	32	107
6.3C	017	3.0 (2.5-3.5)	62.5	3.2	26	11	14	20	65
7.2	018	0.60 (.47-1.3)	73.2	2.7	11	3	4	8	37
7.2E	019	1.13 (.85-1.51)	81.2	2.5	7	2	3	7	31
7.4N	020	0.64 (.53-.78)	84.4	2.4	6	2	3	7	19
7.4	021	0.43 (.33-.55)	83.5	2.4	5	2	4	12	22
LB5	023	0.96 (.81-1.1)	56.2	3.6	24	13	12	23	63
LB6	024	7.4 (5.4-10.0)	32.7	6.1	49	22	28	44	163
(Ocean Samples)									
NB1	025	0.95 (.85-1.12)	93	2.1	2	2	3	6	4
(Delaware Bay Samples)									
DB2E	026	0.65 (.59-.76)	84.4	2.4	9	2	6	10	31
DB2C	027	0.96 (.76-1.2)	83.8	2.4	9	3	6	10	43
DB2W	028	0.53 (.41-.69)	86.7	2.3	5	2	3	6	15
DB3E	029	0.60 (.46-.81)	79.6	2.5	8	4	8	7	40
DB3C	030	0.78 (.69-.88)	81.3	2.5	6	2	6	7	35
DB3W	031	1.4 (1.2-1.7)	84.5	2.4	8	2	6	8	44

Underlined ET50 values are significantly different from controls

It was apparent there was no correlation of burrowing speed inhibition with any of the sediment chemical or physical constituents analyzed by EPA. The lack of correlation of burrowing speed inhibition with the conventionally measured chemical and physical characteristics of the sediment was consistent with results from artificial contamination of marine sediment with copper (Phelps, e.a., 1985) which found clam

burrowing inhibition was related only to changes in sediment pore water chemistry. Other investigators have reported estuarine biota are sensitive only to ionic and unsorbed chemical species (Sunda and Guillard, 1976). If future studies hope to relate sediment chemical analysis to effects on living organisms, sediment pore water chemical analysis will be necessary.

Four of the eleven sediments causing inhibition of burrowing speed were retested with the bioassay over a three day period (Table 6). There was enough of one sample (LB6) to use previously untested material on the third day.

Table 6

Burrowing Speed Laboratory Retests

Sediment Sample	Test Date	ET50 (95% F.L.)
LB6	5/25	7.4 (5.4-10.0)
	5/27	0.80 (.70-.92)
(fresh)	5/27	1.20 (.93-1.55)
DB6E	5/26	2.1 (1.7-2.6)
	5/27	1.6 (1.2-2.0)
	5/28	0.17 (.13-.22)
DB6W	5/26	5.6 (4.2-7.4)
	5/27	5.5 (3.3-9.1)
	5/28	0.24 (.17-.35)
6.3C	5/27 (9AM)	3.0 (2.5-3.5)
	5/27 (7PM)	1.1 (.83-1.4)

Underlined values significantly greater than simultaneous controls

All of the retested sediment samples showed loss of the burrowing inhibition by the third day. The loss of burrowing inhibition by sediments within three days following the cruise is

consistent with laboratory studies on copper-spiked sediment showing a sharp decrease in burrowing inhibition within three days associated with loss of sediment pore water copper (Phelps, e.a. 1985).

Sediment toxicity was tested by adding 20 clams to two sediments that had shown inhibition of burrowing, MCB3.3C and MCB3.3E, and a control sediment, and holding on the flowing water sea table for two weeks (6/4 - 6/18). One test sediment (MCB3.3E) had a 10% loss (2 clams), and the surviving clams showed no debility in burrowing speed. This failure of aged (several day old) sediments to cause long-term mortality was consistent with previous laboratory studies showing sediments were only toxic to clams when they caused burrowing inhibition, i.e. had toxic pore water (Phelps, e.a. 1985).

These post-cruise studies showed that the sediment samples rapidly lost the ability to inhibit burrowing so probably lost pore water toxicity as well. It is now apparent long-term mortality studies must be started at the time of sediment collection. It would be useful to repeat and extend this burrowing bioassay study to include such toxicity testing.

In general, the shipboard clam burrowing bioassay for estuarine sediments had controls similar to laboratory tests and was not significantly affected by any test conditions except changes in salinity of the test water. The test clams could be held aboard ship at 10 deg. C if placed at room temperature for six hours before the bioassay. Therefore the burrowing bioassay could be applied to fresh estuarine sediment samples on shipboard

as long as the clams and the overlying test water were at the same salinity. The shipboard testing of the clam burrowing sediment bioassay confirmed the importance of conducting the bioassay within a day or two of collecting the sediment. Perhaps the generally negative results for the clam burrowing bioassay reported by Chapman, e.a. (1987) were due to the up to one week storage of sediments before testing.

OYSTER LARVA SEDIMENT BIOASSAY

A 96-hour estuarine sediment bioassay was explored using mortality and metamorphosis of Crassostrea gigas larvae. The eyed pediveliger larvae are readily available almost year round from a West Coast oyster hatchery. The larvae were pretreated for 20 hours with epinephrine to induce settlement (Coon, e.a. 1986). Larval batches varied in competency.

For the bioassay, one ml of sediment pressed through a 149 u Nytex screen was placed in a 2.75 ml well of a 24-well Falcon tissue-culture plate. One ml water was layered on top and 20 - 40 larvae placed on the sediment-water interface. After four days at room temperature the well contents were filtered through 149 u Nytex mesh and the retained larvae examined for mortality and metamorphosis. Staining with neutral red improved live-dead determination (Crippen and Perrier, 1974).

C. gigas larvae showed good survival from 8 - 34 ppt salinity with average control seawater mortality of 3% and control sediment mortality of 7%. There was no metamorphosis below 23.5 ppt. salinity. Copper-spiked sediment as a negative control caused increased mortality and inhibited all

metamorphosis. Baltimore Harbor sediment caused 100% mortality.

Stored control sediment (17 ppt) showed increasing toxicity to larvae and lowered pH after two months at 4 deg. C or 0 deg. C (frozen). Fresh Baltimore Harbor sediment lost toxicity when stored for one week at four deg. C or frozen, as did stored copper-spiked sediment. It is recommended that bioassays be carried out with fresh sediment. One bioassay with native Crassostrea virginica larvae showed responses similar to C. gigas (Phelps and Warner, 1988).

The Sediment Quality Triad concept should be applied to evaluating pollutant-caused degredation of Chesapeake Bay sediments. Combination of the short-term M. arenaria burrowing bioassay with the 96-hour oyster larva sediment toxicity bioassay using either C. gigas or C. virginica will now permit rapid bioassay testing of Chesapeake Bay sediments. An amphipod bioassay may also be developed. From these studies it is apparent that storage of sediment samples affects bioassay results. The clam burrowing and oyster larva bioassays are compact and can be used on shipboard to test the fresh sediments collected during cruises. These bioassays can also be used in the laboratory with experiments on sediment fluxing of toxicants. The effectiveness of chemical toxicants under estuarine conditions is often not known. All sediment studies should be combined with bioassays.

AKNOWLEGMENTS

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REFERENCES

Chapman, P.M., R.N. Dexter and E.R. Long. 1987. Synoptic measures of sediment contamination, toxicity and infaunal community composition (the Sediment Quality Triad) in San Francisco Bay. Mar. Ecol. Prog. Ser. 37:75-96.

Chapman, P. (in press). Marine Sediment Toxicity Tests. In: Symposium on Chemical and Biological Characterization of Sludges, Sediments, Dredge Spoils and Drilling Muds. ASTM Special Technical Publication.

Coon, S.L., D.B. Bonar and R. M. Weiner. 1986. Chemical production of cultchless oyster spat using epinephrine and norepinephrine. Aquaculture, 58:255-262.

Crippen, R.W. and J.L. Perrier. 1974. The use of neutral red and Evans blue for live-dead determinations of marine plankton. Stain Technology 49:97-104.

EPA. 1985a. Cruise Plan, Operation MAREST, Cruise 85-I, OSV Antelope, May 17-23 1985. EPA Annapolis Field Center, Annapolis, MD.

EPA. 1985b. Cruise Report, Operation MAREST, Cruise 85-I, OSV Antelope, May 17-23, 1985. EPA Annapolis Field Center, Annapolis, MD.

EPA. 1985c. U.S. EPA Contract Laboratory Program, Case No. 1689HO. Sample Management Office, Alexandria, VA.

Phelps, H.L. 1985. Development of a Shellfish Behavior Bioassay for Chesapeake Bay Sediments. Final technical report to the University of the District of Columbia Faculty Senate. 25pp.

Phelps, H.L., W.R. Pearson and J.T. Hardy. 1985. Clam burrowing behavior and mortality related to sediment copper. Mar. Poll. Bull. 16:309-313.

Phelps, H.L. 1987. Clam Burrowing Bioassays of Chesapeake and Delaware Bay Sediments, Operation MAREST. Final technical report to EPA. 17pp.

Phelps, H.L. and K. Warner. 1988. Development of an Estuarine Sediment Solid-Phase Bioassay Using Oyster (Crassostrea gigas) Larval Mortality and Metamorphosis. Final technical report to U.S. Fish and Wildlife Service, Annapolis, MD. 30pp.

Sunda, W. and R.R.L. Guillard. 1976. The relationship between cupric ion activity and the toxicity of copper to phytoplankton. J. Mar. Res. 34:511-529.

Tsai, C.F., J. Welch, K.W. Chang, J. Shaeffer and L.E. Cronin. 1979. Bioassay of Baltimore Harbor sediments. Estuaries 2:141-153.

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Abstract only

Development of an Estuarine Solid-Phase Sediment Bioassay Using Molluscan Larvae

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Pollutants in estuarine water are exposed to a large sorptive area of suspended solids and become concentrated in sediments. It is not known whether sediments are safe sinks for estuarine pollutants. One relatively inexpensive method of determining the bioeffectiveness of sediment-sorbed pollutants is through bioassay of the solid-phase sediment. An estuarine sediment bioassay has been explored using the sensitive early life-history larval stage of available molluscs: Crassostrea gigas for euryhaline sediment and Corbicula fluminea for tidal freshwater estuarine sediment.

Crassostrea gigas eyed pediveliger larvae are readily available almost year round from a West Coast oyster hatchery. Mature Corbicula larvae are released from adult clams collected from the Potomac during the spring and fall spawning seasons. Except for the pretreatment of C. gigas larvae with epinephrine to induce settlement, the bioassay technique was similar for both species. Sediment was pressed through a 149 u Nytex screen and approximately one ml placed in a 2.75 ml well of a 24-well Falcon tissue-culture plate. One ml water was added to each well and 20 - 40 molluscan larvae placed on the sediment-water interface. After four days at room temperature the well contents were filtered through 149 u Nytex mesh and the retained larvae examined for obvious mortality.

C. gigas larvae showed good survival from 8 - 34 ppt salinity with average control seawater mortality of 3% and control sediment mortality of 7%. Copper-spiked sediment was used as a negative control and caused increased mortality in both C. gigas and Corbicula larvae. Field sediments from Baltimore Harbor and Anacostia Navy Yard caused mortalities of 100% and 79% for C. gigas and Corbicula larvae respectively.

Stored control sediment (17 ppt) had increasing toxicity to larvae and lowered pH after two months at 4 deg. C or 0 deg. C (frozen). Fresh Baltimore Harbor sediment was much more toxic than sediment stored for one week at four deg. C or frozen. Stored copper-spiked sediment lost 39% of toxicity after eight days storage at four deg. C. It appears that diagenesis of sediment-sorbed esturine toxics may occur rapidly in storage and it is recommended that biotoxicity tests be carried out with fresh sediment. One comparison bioassay with Crassostrea virginica larvae showed a response to copper-spiked sediment similar to C. gigas. This bioassay may be a useful indicator for field studies of estuarine sediment pollution and for experimental studies on sorption and desorption of bioeffective sediment pollutants.

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CONCURRENT SESSIONS AND POSTER SESSION:

DISSOLVED OXYGEN

Chairs:

William Rickards
Virginia Sea Grant College Program

Jack Greer
Maryland Sea Grant College Program

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Abstract only

Organic Carbon, Oxygen Consumption, and Bacterial Metabolism in Chesapeake Bay

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Bacterial metabolism is the most important oxygen consuming process leading to oxygen depletion in Chesapeake Bay. The bacteria are extremely versatile in their ability to utilize available organic carbon sources for their metabolism and growth. Current evidence indicates that phytoplankton production is the principal source of organic carbon fueling the bacterial oxygen consumption. However, the linkage among phytoplankton production, bacterially labile organic pools and bacterial metabolism is neither simple nor direct. Questions persist regarding the temporal and spatial scales of organic carbon production, speciation and consumption. Because of these complexities it is crucial that appropriate estimators of bacterially utilized carbon pools be identified and measured in order to assess water quality and to project trends in oxygen depletion.

With these goals in mind, we simultaneously estimated organic carbon pools (chlorophyll a [CML], biochemical oxygen demand [BOD], amino acids [AA], and carbohydrates [CHO] and bacterial abundance and metabolism at stations along two transects in the mesohaline portion of Chesapeake Bay. One transect was located off the Patuxent River (PAX) while the other was off the Great Wicomico River (GWR) in Virginia. During the summer of 1987 the deep water portions of both transects experienced severe oxygen depletion and were anoxic for differing periods. Anoxia was transient at the GWR transect but persisted longer at the PAX transect.

On a seasonal scale, BOD ranged from about 0.5 to more than 7 mg/l and reached a maximum in the surface waters along the western portion of the GWR transect. During the spring, however, BOD was highest in the bottom water at both transects and averaged 3 - 4 mg/l. This inverse relationship with depth mirrored the spring CHL depth distribution. Bottom water CHL, up to 80 ug/l, in April was about 3 times greater than in the surface. Throughout the water column in spring, most BOD was particulate in nature, but during summer, dissolved organics dominated the BOD especially below the euphotic zone. In many cases nearly 100% of the BOD was dissolved. Dissolved free AA (0.5 - 1

umolar) and dissolved CHO (0.5 - 2 umolar) were present in high concentration, and during the summer represented approximately 50% of the BOD. These substrates may be even more important than their static concentrations imply because high bacterial turnover rates, up to about 50%/h, require a rapid production of the substrate. Dissolved free CHO appear to be a major organic substrate fueling bacterial oxygen consumption in the Bay. However, during highly stratified summer conditions AA concentrations and metabolism, which peaked near the pycnocline, could represent a major biological barrier to reaeration of deep waters.

Spatially, we found that organic carbon concentrations, including CHL, BOD, AA, and CHO, were higher at the GWR transect than at the more northerly PAX transect. Similar results were found for bacterial abundance and metabolism. These results would be consistent with the hypothesis that organic carbon from the southern mesohaline Chesapeake Bay drives oxygen depletion farther to the north. The lateral distribution of these parameters suggests the possibility that there may be a significant interaction with Potomac River water entering the Bay on the west.

Understanding the Estuary: Advances in Chesapeake Bay Research. Proceedings of a Conference. 29-31 March 1988. Baltimore, Maryland. Chesapeake Research Consortium Publication 129. CBP/TRS 24/88.

Fluxes of Carbon, Nitrogen, and Oxygen Through Estuarine Bacterioplankton

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Over the past decade biological oceanographers and aquatic ecologists have discovered that microbial plankton less than about 10 microns in diameter often dominate the biomass and fluxes of C,N,P and O in most marine systems (Pomeroy, 1974; Ducklow, 1983). This finding extends to the heterotrophic bacterioplankton, most of which are less than 0.5 microns in diameter. Bacterioplankton metabolize on the average about 40% of the daily primary production in habitats ranging from polluted estuaries to the open sea (Hobbie and Cole, 1984; Ducklow, 1982, 1986). The pathways over which energy and nutrients flow to the bacteria are not well documented, and the consequences of the domination of nutrient cycling by very small organisms are poorly understood. However it has been suggested that longer, less efficient food chains result from such a trophic foundation, because larger organisms like copepods and fish larvae are generally unable to harvest micron-sized particles efficiently. This had led to the idea that the bacterioplankton are a net sink for energy and carbon in marine foodwebs (Ducklow et al., 1986) rather than an important source of nutrition for economically important organisms.

Ecological and fisheries research has begun to show that this situation occurs in Chesapeake Bay. That is, it appears that much of the primary production flows through the lower trophic levels dominated by nanophytoplankton, bacteria, and protozoans with a relatively small fraction appearing in yields of harvestable species. At the same time we have begun to realize that nutrient inputs to the Bay are flowing not to fish and shellfish, but into the microbes (Nixon et al., 1986), resulting in seasonal anoxia. Nutrient inputs

are increasing, but we cannot yet predict the fate of these inputs, or the consequences of continuing increases with much accuracy.

In order to improve our understanding of the mechanisms by which nutrient inputs are dissipated through microbial food chains and cause net consumption of oxygen in Chesapeake Bay, we began a comprehensive study of the flows of C,N and O through bacterioplankton in the mid-Bay region in 1984. This paper summarizes some of our findings from the last four years.

Our sampling design has addressed lateral (cross-Bay) and vertical (top-to-bottom) variations in bacterial abundance and production in the region between the Bay Bridge and the Patuxent River (Malone et al., 1986). In different years we have altered the temporal resolution of our studies to observe responses to climatic and meteorological forcings on scales of days to months, with particular emphasis on the period of rapid oxygen decline (May), and the period of maximum stratification and anoxia (August). In this discussion we concentrate on bacterial processes in the surface layer and euphotic zone.

After four years of study we have a good composite view of the seasonal cycle, and some impression of interannual variability (Figure 1). Overall, both bacterial abundance and production are closely tied to the annual temperature cycle, with low values in late winter and early spring, and maxima in summer. Mean monthly abundance reaches a maximum in July or August (Figure 1a) whereas production is highest in June (Figure 1b). This phase shift indicates that in situ removal of bacteria by bacteriovores is not closely tied to their production. Summertime levels of abundance range from 5 to over 25 million cells per ml (Malone et al., 1986). These are among the highest sustained levels of bacterial abundance observed in estuaries (cf. Coffin and Sharp, 1987).

By regarding these data together we can estimate specific growth rates (production/abundance, units per day). During all periods, the gross growth rates (0.3-1.3 per day) are well in excess of the dilution rate of the surface layer (0.03-0.08 per day) in this area. This large difference between the production and dilution of the bacteria suggests that unless the grazers are capable of removing nearly all of the bacterial production each day, the bacteria will accumulate in this region, as the high abundance levels also indicate.

Interannual variations in abundance and production are also out of phase. Bacterial abundances are fairly uniform from year to year except in late summer (Figure 1a) when stratification, temperature and anoxia are at their seasonal maxima. We do not yet understand this variability. High growth rates relative to low dilution rates in summer, when flows are low, suggest that variations in the couplings between bacteriovores and their prey are a possible cause of the variability in summer. Bacterial production levels are similar in summer, but vary by 2-4 fold in spring (Figure 1b), when interannual variations in freshwater flows are greatest. Again, we do not understand this pattern, but suggest that it is related to variations in exogenous inputs of inorganic nutrients and organic matter which are greatest in spring (Malone et al., in press).

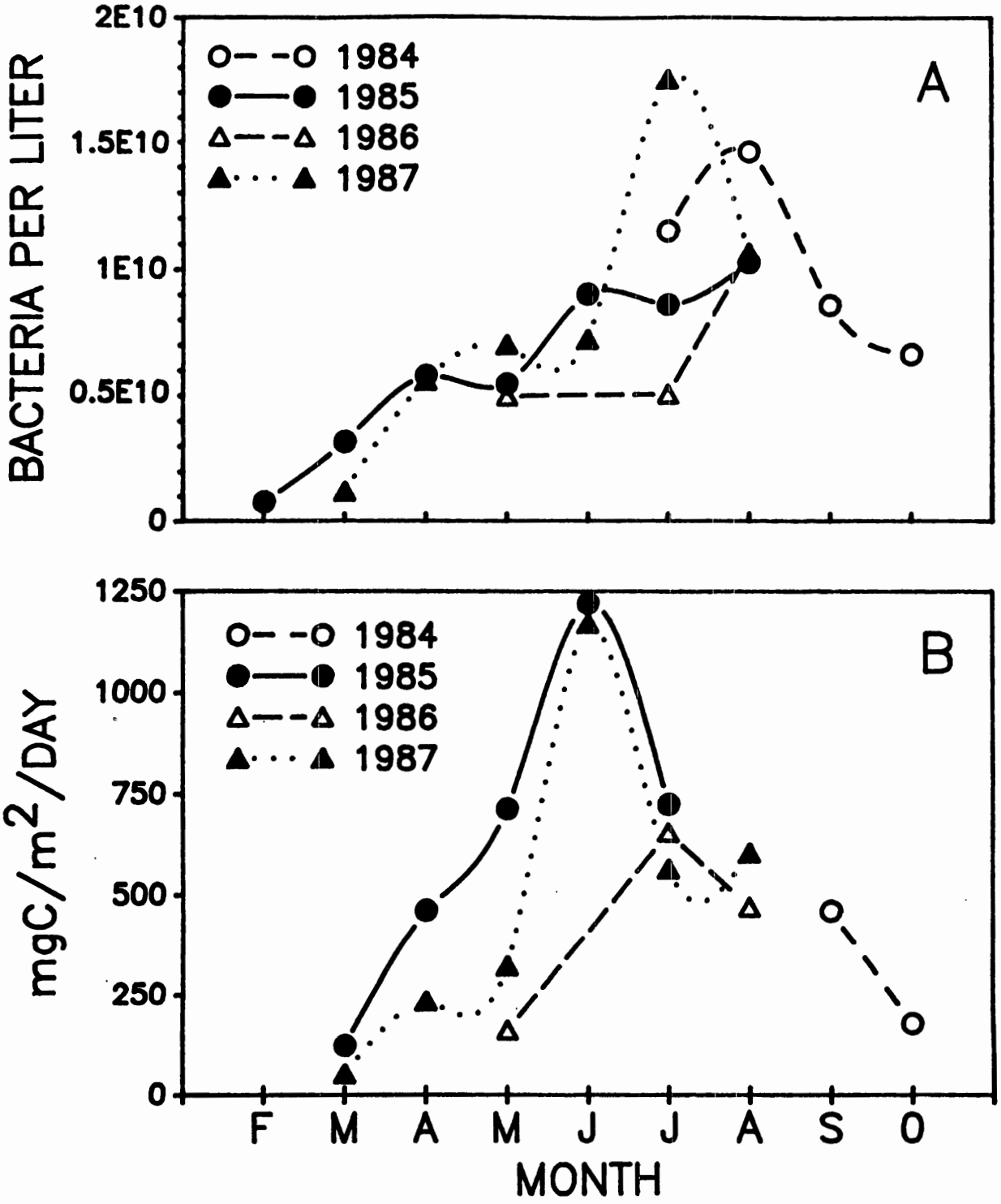

Figure 1. Mean monthly bacterial abundance (A) and bacterial production (B) for the mid-Bay region (Bay Bridge to Patuxent River) during 1984-1987.

Our measurements show that bacterial production in the euphotic zone ranges from about 5% to over 100% of the daily primary production, with levels often surpassing 35% (Figure 2). Bacterial production is very high in Chesapeake Bay, compared to a "global" average of about 15-20% (Cole et al., in press). Bacterial production clearly cannot exceed the rate of primary production for very long, unless there are additional inputs of organic matter for the bacteria to metabolize. In the absence of exogenous inputs of organic matter, bacterial production must be limited to some fraction of primary production, the value of which is set by the conversion efficiencies of the bacteria. Conversion efficiencies for bacterioplankton using natural dissolved organic matter (DOM) are not well characterized (Bjornsen, 1986) so we have resorted to flow analysis models (Fasham, 1985; Ulanowicz, 1986) to estimate the relative importance of internal and external sources of DOM for bacterial growth in the mid-Bay.

Bacterioplankton in most marine systems, including Chesapeake Bay, depend primarily on the flux of DOM to sustain their growth. In estuaries DOM supplies may be internal or external. Internal sources include the release of carbohydrates, proteins and amino acids from phytoplankton and grazers within the euphotic zone (Figure 3). External (exogenous) inputs are those originating from outside the euphotic zone of the mid-Bay area, which include DOM in the freshwater input, or that diffusing upward from the bottom layer and/or sediment (Figures 3,7). Furthermore it is clear from Figure 3 that external inputs of dissolved inorganic nitrogen (DIN) will also be converted into internal inputs of DOM for bacteria. Thus both the fluxes of DIN and DOM must be considered to understand the high levels of bacterial production in the Bay.

To estimate the ranges of bacterial production which can be supported solely by internal sources of DOM, given a specified level of external DIN, we have performed sensitivity analyses of the model foodweb shown in Figure 4. This linear input-output model is described more fully in Ducklow et al., (submitted). To use it, we specify the levels of primary production (1 unit or 100%), bacterial production (varied from 1 to over 50%) and the relative dependency of the phytoplankton on internal (recycled NH_4 or urea) and external DIN (nitrate from rivers or ammonium from the bottom layer). This dependency sets the overall level of grazing on phytoplankton and bacteria. Next physiological parameters like the assimilation efficiencies for zooplankton, conversion efficiencies for bacteria and excretion rates for phytoplankton and zooplankton are varied over reasonable ranges and the combinations of parameter values yielding steady-state solutions for each level of bacterial production and DIN input are identified.

Figure 5 shows the levels of bacterial production (as a percentage of primary production, or PP) which can be supported solely on internal sources of DOM for a given level of DIN input, at steady state. The results show that a maximum bacterial production of about 40% of PP can be supported by internal supplies of DOM. However this is only attained at steady state when DIN inputs are low and PP is dependent mostly on recycled nitrogen. When PP dependence on external

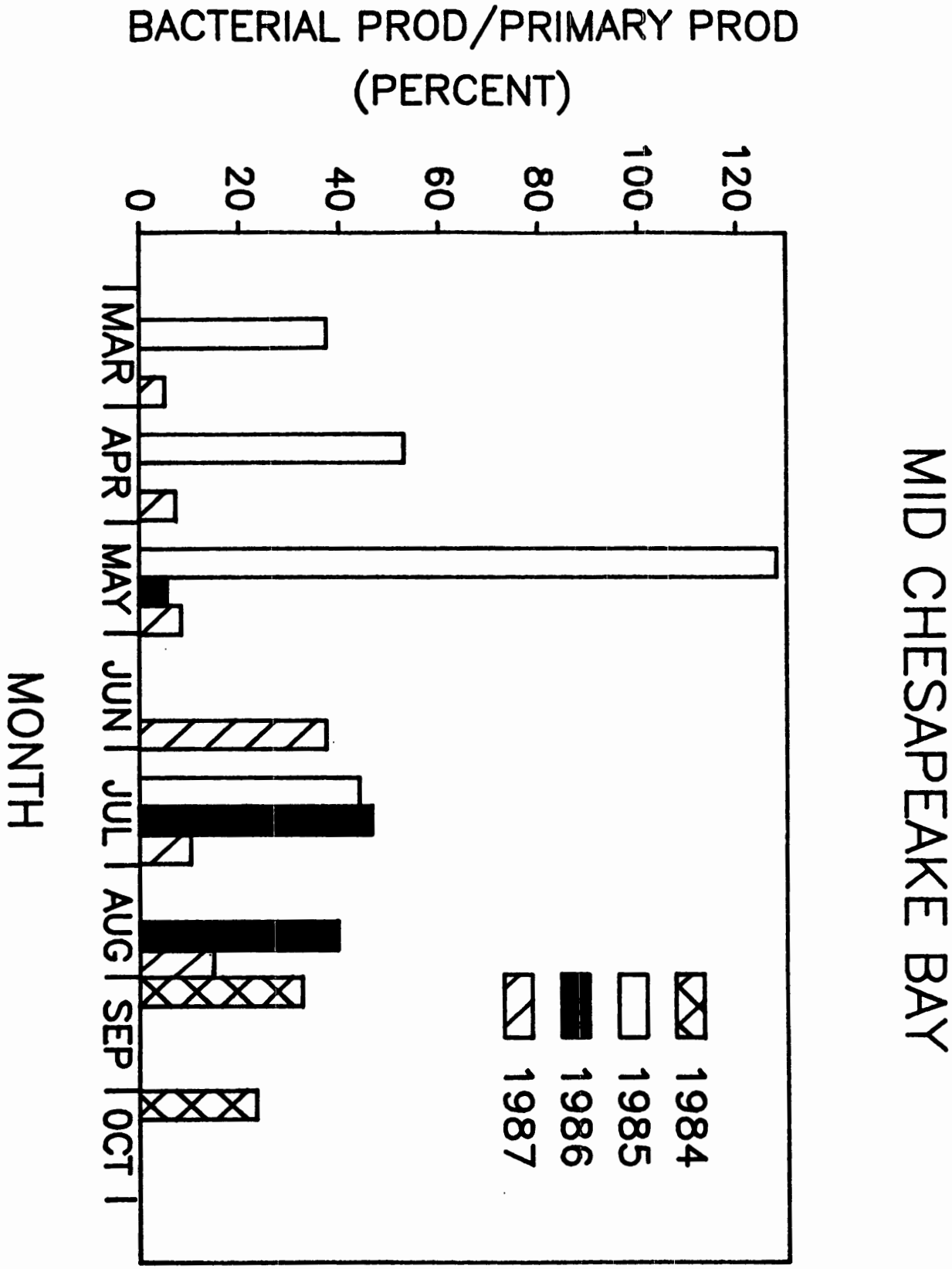

Figure 2. Mean monthly levels of bacterial production, expressed as a percentage of daily primary production in the mid-Bay region during 1984-1987.

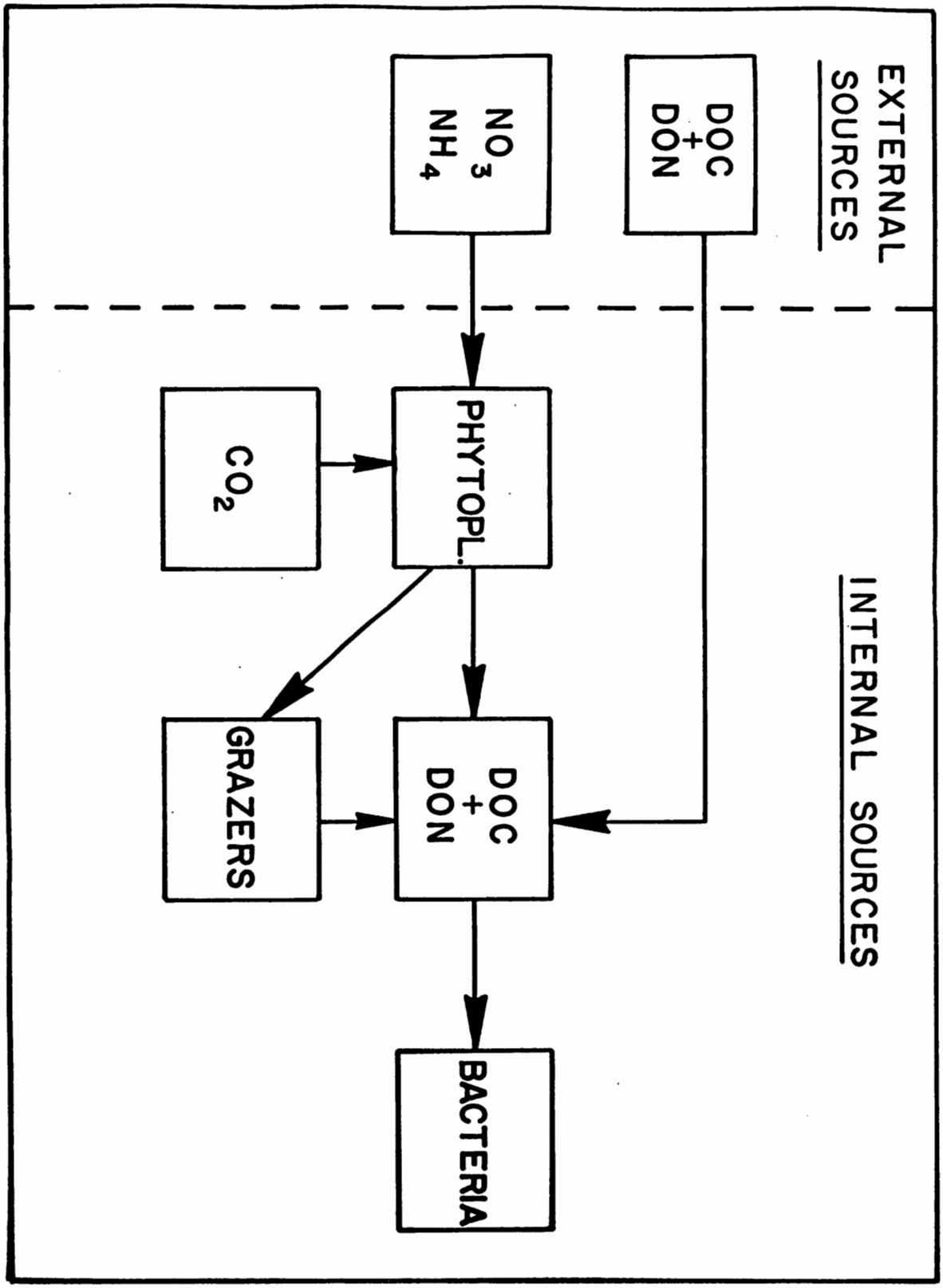

Figure 3. Schematic view of internal and external sources of nutrition for bacterioplankton in the estuarine euphotic zone. All sources generated by in situ processes within the mid-Bay surface layer are internal; all others are external.

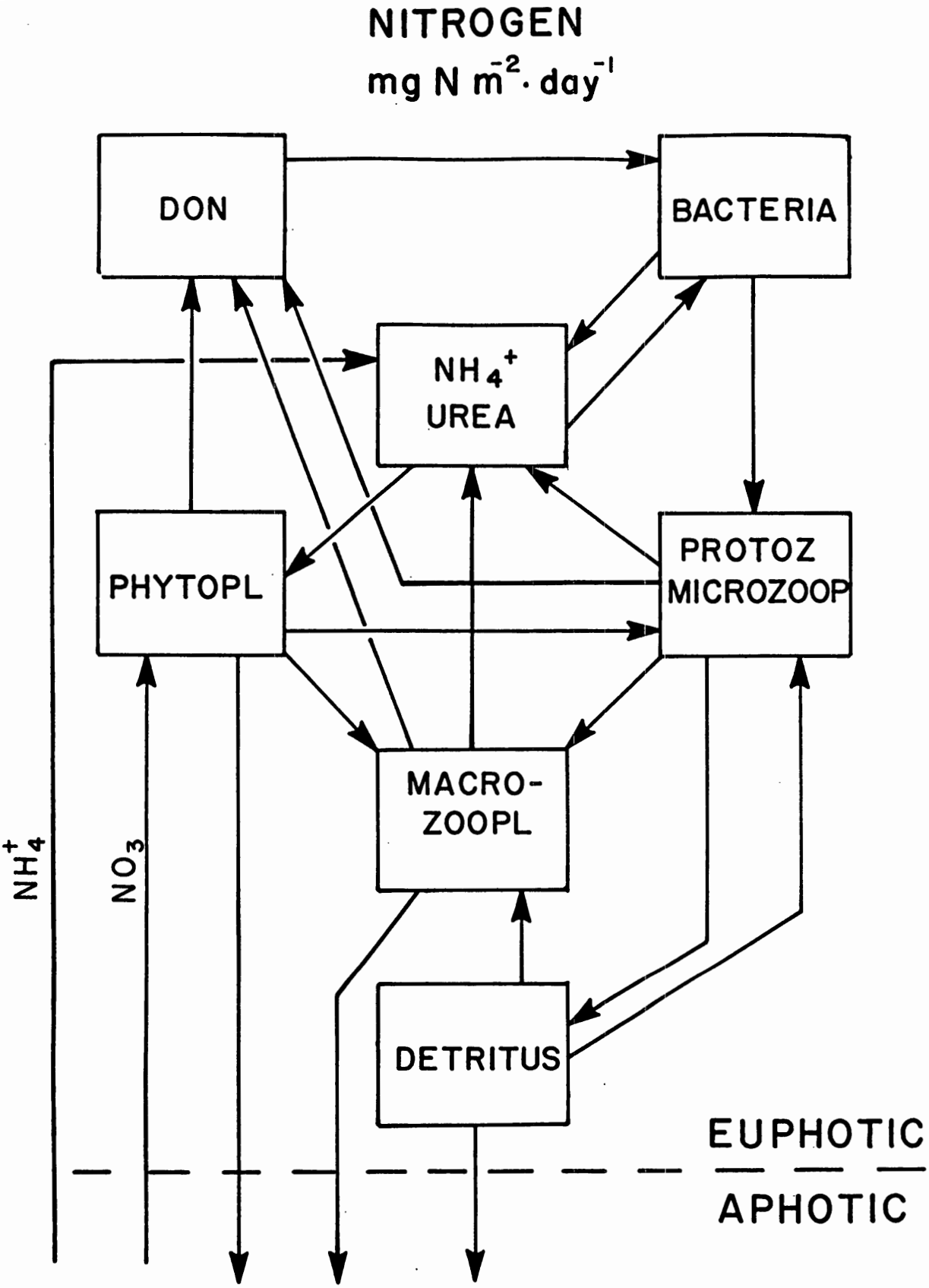

Figure 4. Flows of nitrogen through a generic planktonic foodweb. Fluxes and parameters governing the partitioning of inputs and outputs for each pool are set as described in the text and in Ducklow et al., (1988).

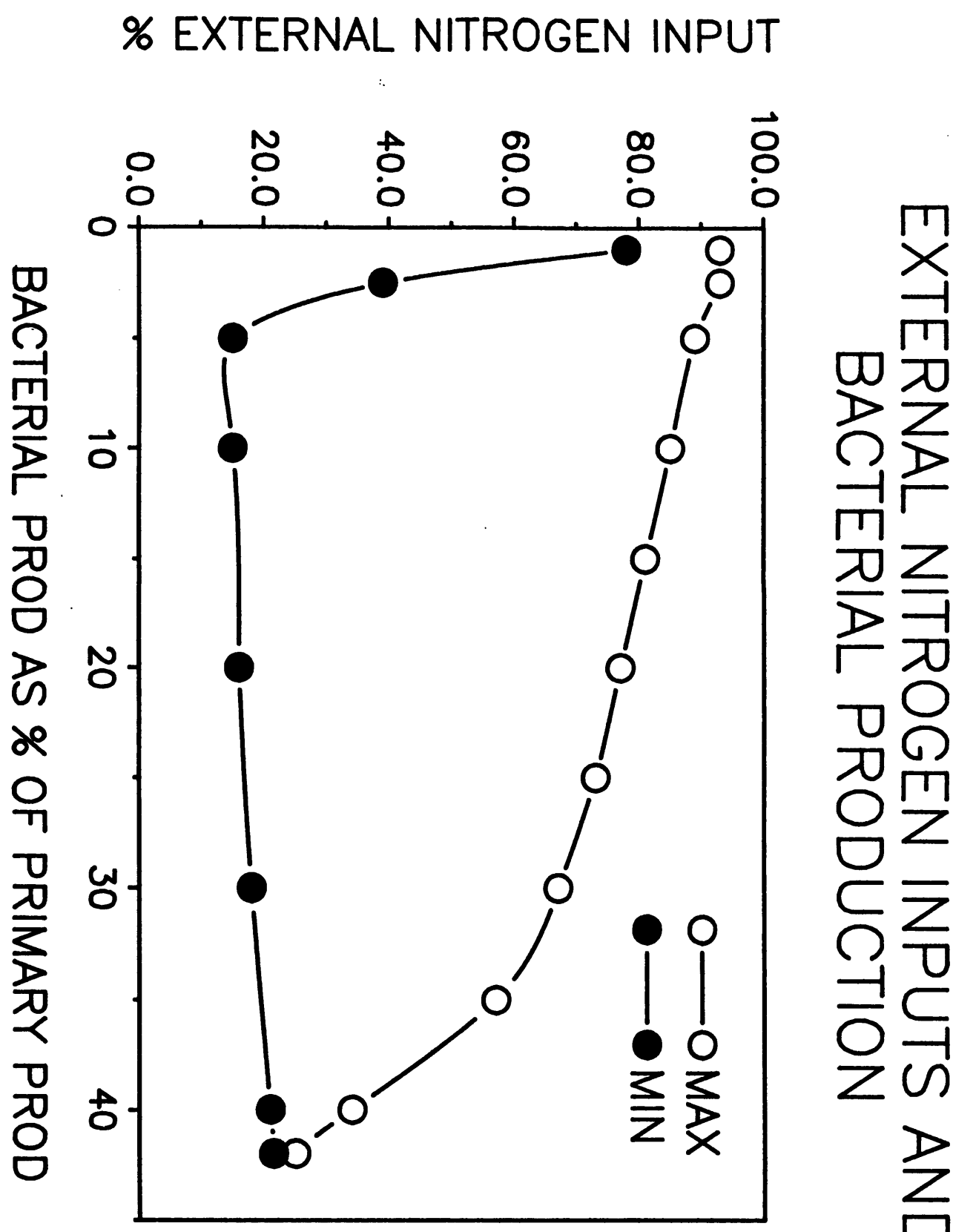

Figure 5. Results of sensitivity analysis of the model shown in Figure 4. This graph describes the relationship between external inputs of inorganic nitrogen and the amount of bacterial production which can be sustained entirely on internal sources of DOM at steady state. Compare maximum levels with Figure 2.

DIN is higher, bacterial production must depend to some extent on external inputs of DOM or decline to maintain the steady state.

It is important to remember that this result only applies to steady state situations when the stocks of all pools are not changing. In addition it should be noted that there are numerous combinations of parameters and fluxes for which permissible solutions can be obtained. This analysis merely points out a crude estimate of the relative levels of bacterial production which can be sustained without external DOM supplies. However, given these caveats, it is striking to compare the model results in Figure 5 with our measurements in Figure 2. About half the time for which we have data, bacterial production is near the maximum sustainable levels, or above them. Thus if we assume that in the mid-Bay region most of the DOM is produced in situ (as is the case for particulate organic matter, Biggs and Flemer, 1972), and further assume normal parameter ranges for DOM release and conversion efficiency, then we see that the fluxes of C and N through the bacteria are about as large as they could be.

What are the characteristics of estuaries in general and Chesapeake Bay in particular which lead to high levels of primary and bacterial production? Clearly PP is high because nutrient inputs are high (Boynton et al., 1982). Malone et al., (in press) have shown how external inputs of DIN interact with the estuarine circulation to boost PP in summer (Figure 6). Large amounts of phytoplankton biomass produced in spring are not consumed and sink into the bottom layer of the mid-Bay. The net upstream transport in the bottom layer tends to retain this material in the region and concentrates it from the larger area over which it was derived. In late spring and summer this biomass is decomposed, consuming oxygen and producing recycled NH_4. Then the wind-driven lateral circulation ("tilting") supplies NH_4 to the surface, triggering higher levels of PP (Malone et al., 1986). We suggest this scenario can also explain high levels of bacterial production. Decomposition of the spring phytoplankton also releases DOM, which is incompletely metabolized in the anoxic bottom layer (Figure 7). During tilts this DOM may be made available to bacterial metabolism in the euphotic zone, boosting bacterial production above the level which could be maintained on internally produced DOM alone.

Much of the above is speculative. Furthermore we have little understanding of the trophic mechanisms and foodweb structures and dynamics responsible for the high levels of DOM flux in the Bay. It does appear clear that bacterial production is very high in Chesapeake Bay. Whether this condition is peculiar to the 20th century Chesapeake Bay, heavily impacted by anthropogenic nutrient inputs, or if it is common to all stratified estuaries remains to be seen. Whatever the answer, we can conclude that the nutrient input to the Bay is flowing primarily through microbial foodwebs, and not efficiently producing oysters and finfish. Further research on microbial processes is needed to discover how this pattern might be reversed.

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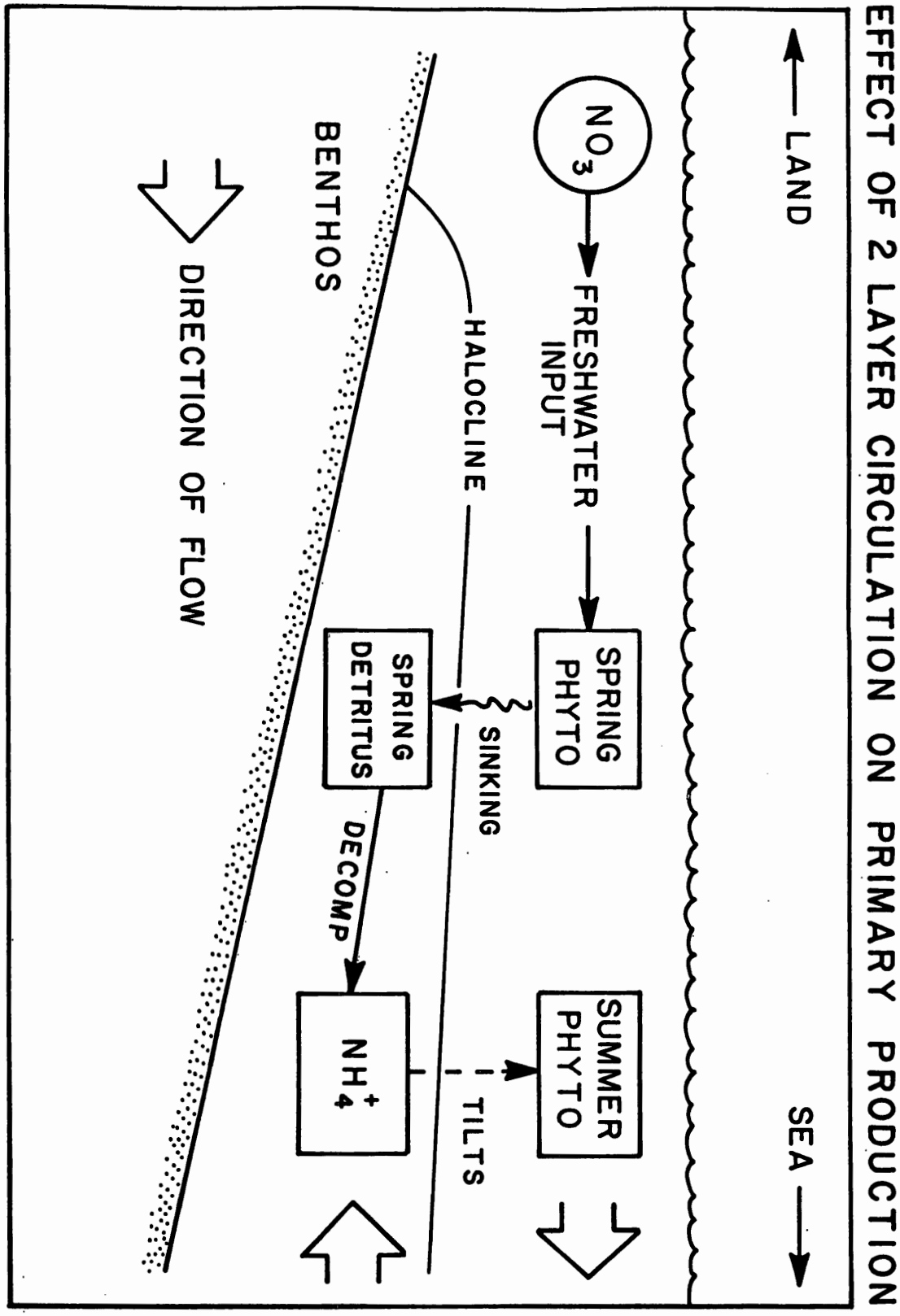

Figure 6. Cartoon view of the interaction between the two-layered estuarine circulation and phytoplankton production in Chesapeake Bay (after Malone et al., 1988).

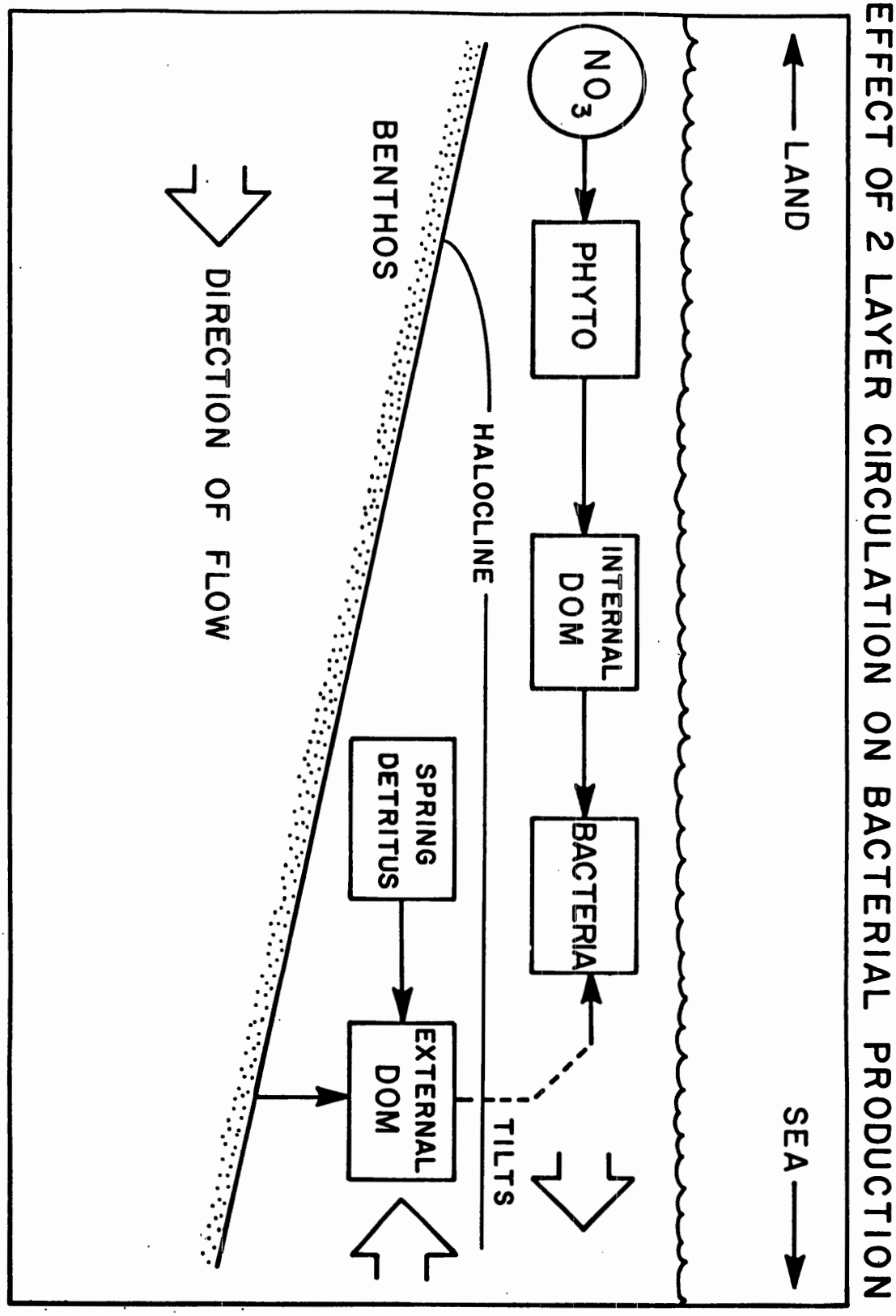

Figure 7. Cartoon view of the interactions between the two-layered estuarine circulation, phytoplankton and bacterial production in Chesapeake Bay.

available their data on phytoplankton and nutrients. Jon Tuttle and Bob Jonas also lent data and discussed bacterial processes. We thank the captains and crews of the CEES research vessels for their support. This research was sponsored by the US EPA, Chesapeake Bay Office, by the University of Maryland Sea Grant Program and NOAA, and by the Horn Point Environmental Laboratory.

REFERENCES

Biggs, R.B., and D.A.Flemer, 1972, The flux of particulate carbon in an estuary, Mar.Biol. 12:11-17.

Bjornsen, P.K., 1986, Bacterioplankton growth yield in continuous seawater cultures. Mar.Ecol. Prog.Ser. 30:191-96.

Boynton, W.R., W.M. Kemp and C.W.Keefe, 1982, A comparative analysis of nutrients and other factors influencing estuarine phytoplankton production. In: Estuarine Comparisons (V.S.Kennedy, Ed.), pp 69-90. Academic Press, New York.

Coffin, R.B., and J.H.Sharp, 1987, Microbial trophodynamics in the Delaware Estuary, Mar.Ecol.Prog.Ser. 41:253-257.

Cole,J.J., M.L.Pace, and S.Findlay, 1988, Bacterial production in fresh and saltwater ecosystems: a cross-system overview, Mar.Ecol.Prog.Ser. in press.

Ducklow, H.W., 1982, Chesapeake Bay nutrient and plankton dynamics. I. Bacterial biomass and production during spring tidal destratification in the York River, Virginia, estuary. Limnol. Oceanogr. 27:651-659.

Ducklow, H.W., 1983, The production and fate of bacteria in the ocean, BioScience 33:494-501.

Ducklow, H.W., 1986, Bacterial biomass in warm core Gulf Stream ring 82B: Mesoscale distributions, temporal changes, and production. Deep Sea Res. 33:1789-1812.

Ducklow, H.W., D.A. Purdie, P.J.LeB.Williams, and J.M.Davies, 1986, Bacterioplankton: A sink for carbon in a coastal plankton community, Science 232:865-67.

Ducklow, H.W., M.J.R.Fasham, and A.F.Vezina, 1987, Flow analysis of open sea plankton networks, In: F.Wulf, Ed., Lecture Notes on Coastal and Estuarine Studies, Vol XX, Springer Verlag, New York.

Fasham, M.J.R. 1985. Flow Analysis of Materials in the Marine Euphotic Zone. Can. Bull. Fish. Aquat. Sci. 213:139-162.

Hobbie, J.E., and J.J. Cole. 1984. Response of a detrital food web to eutrophication. Bull. Mar. Res. 35:357-363.

Malone, T.C., W.M.Kemp, H.W.Ducklow, W.R.Boynton, J.H.Tuttle, and R.B.Jonas, 1986, Lateral variation in the production and fate of

phytoplanktnkton in a partially stratified estuary, Mar Ecol Progr Ser 32:149-160.

Malone, T.C., L.H.Crocker, S.E.Pike, and B.W.Wendler, 1988, Influences of river flow on the dynamics of phytoplankton production in a partially stratified estuary, Mar Ecol Progr Ser (in press).

Nixon, S.W., C.A.Oviatt, J.Frithsen, and B.Sullivan, 1986, Nutrients and the productivity of estuarine and coastal marine ecosystems, J.Limnol.Soc.South Africa 12:43-71.

Pomeroy, L.R., 1974, The ocean's food web: A changing paradigm. BioScience 24:499-504.

Ulanowicz, R.E., 1986, Growth and development: Ecosystems Phenomenology, Springer-Verlag, New York.

Understanding the Estuary: Advances in Chesapeake Bay Research. Proceedings of a Conference. 29-31 March 1988. Baltimore, Maryland. Chesapeake Research Consortium Publication 129. CBP/TRS 24/88.

Abstract only

Planktonic Respiration in Mesohaline Waters of Chesapeake Bay: Seasonal Patterns of Size-Fractionated Oxygen Consumption with Reference to Depletion of Bottom Water Oxygen

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Planktonic oxygen consumption was measured with dark bottle incubations at ambient temperatures in the spring and summer of 1986 and 1987. Mid-Bay water, sampled south of the mouth of the Choptank River, was collected using Niskin bottle casts from surface and subsurface (pycnocline or bottom) depths. Gentle reverse pressure filtration with membrane filters and Nitex screening was used to fractionate sampled water into two or more sizes (<3 ug 64 u, whole) prior to incubation. Dissolved oxygen was measured with polarographic oxygen electrodes standardized against Winkler titrations. Surface water respiration rates showed the clearest seasonal response. Rates for all size fractions rose as spring progressed into summer (March rates, 4 ug O_2 $1^{-1}h^{-1}$; August, 37 ug $1^{-1}h^{-1}$). This seasonal pattern corresponded to contemporaneous increases in temperature, primary production and microbial and zooplankton biomass. It was interesting to note that the increase in whole water rates was most highly correlated with temperature and not parameters that might reflect substrate availability (ie. phytoplankton biomass or production, total suspended solids). Diel cycles in surface respiration rates did exhibit a strong periodicity reflecting a tight coupling to phytoplankton production in May and August. Maximum rates were measured for water collected in the afternoon. Minimum night time rates were 30-50% of the afternoon maxima. The relative role of microbial respiration as a fraction of total was significantly different between spring and summer. Microbial rates dominated summer respiration throughout the water column. In contrast, microbial oxygen consumption was typically less than 30% of whole water rates in the spring. This distinction has implications regarding the relative importance of free microbes in the plankton community and is probably related to the size spectrum of primary producers. A comparison of water column respiration rates and sediment oxygen consumption illustrates the combined importance of both processes in the depletion of bottom water oxygen in late spring. During the period of rapid oxygen depletion in May preliminary oxygen budget indicated that 60 to 70% of total biological oxygen consumption was attributable to planktonic respiration.

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The Relative Significance of Macrophyte Decomposition and Phytoplankton Respiration in the Consumption of Oxygen in the Lower Chesapeake Bay

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INTRODUCTION

The apparent increase in the impact of hypoxic and anoxic events in the Chesapeake Bay has provided impetus for more intensive research into the processes that lead to such phenomena (Tippie 1984; Officer et al. 1984). While anoxic events are observed in other estuaries and are a common occurrence in bodies of water subject to some stratification, the development and extent of anoxic conditions in the Chesapeake Bay is perceived with alarm (Seliger et al. 1985). The increase in the magnitude and duration of these anoxic events is thought to be due to the microbial oxygen demand in the water column and in the sediments (Taft et al. 1980; EPA 1982; Officer et al. 1984; Seliger et al. 1985). High microbial oxygen demand is believed to be due to the high concentration of organic matter in the Bay. The elevated organic matter concentrations in Bay waters is postulated to be a result of the high inputs of nutrients from the drainage basins feeding the Bay. In aquatic environments, elevated nutrient inputs result in high phytoplankton productivities. In addition, accumulated nutrients in the sediments are thought to maintain the high production rates in the Bay through the summer. High rates of primary production in the water column in turn generate high bacterial abundance, productivity and turnover rates (Tuttle et al. 1987).

The role of physical factors in the development of anoxic conditions in the Bay is documented (Seliger et al. 1985). Since the Bay is a partially stratified estuary, areas of the Bay show varying degrees of stratification through the year. Anoxic conditions develop when oxygen consumption in bottom waters exceeds the rate of reoxygenation because the delivery of oxygenated

surface waters to oxygen depleted bottom waters is reduced or interrupted. Stratified conditions in the water column can develop because of density differences in surface and bottom layers caused by temperature or salinity differences. Salinity differences are greatest during spring due to increased river discharge. Increased river discharge during spring is also thought to provide the nutrients that support the spring bloom of phytoplankton in the Bay (Kemp et al. 1987). The stability or stratification of the water column may also be increased during warm periods when density differences caused by salinity are enhanced by warming of surface layers.

In addition to enhancing stratification, warm temperatures and increased salinity also decrease the solubility of oxygen in water. As temperatures and salinities increase, the pool of available dissolved oxygen decreases. The development of anoxic conditions in the Bay is thought to begin in late winter when bottom temperatures start to rise (Tuttle et al. 1987). With the increase in temperatures, the metabolic rates, and consequently abundance and productivity, of autotrophs and heterotrophs also increase. All these factors further increase oxygen consumption rates.

While the rise in water temperature tends to result in stratification events, wind and tides reduce stratification. These physical factors potentially increase mixing of surface and bottom water preventing anoxic conditions from developing. The interplay of climatological, hydrological and geomorphic factors can then determine the occurrence of anoxic conditions in an estuary that is predisposed to anoxia due to the heavy load of organic matter in its waters.

In aquatic environments, oxygen is consumed by biological and chemical processes in the water column and sediments. Oxygen is consumed during autotrophic and heterotrophic respiration. Although the mineralization of organic carbon is thought to be the major oxygen consuming process, microbially-mediated chemical reactions, such as ammonium oxidation or nitrification, and sulfide oxidation also consume oxygen. Sulfide oxidation can also be a spontaneous chemical reaction. Normally, sulfides are found in estuarine sediments but as anoxic conditions develop, the sulfide containing layer begins to move into the water column. If anoxic conditions persist due to stability of the water column, sulfides may reach the pycnocline (Tuttle et al. 1987). Because sulfide generation requires low-molecular weight carbon compounds as substrates for sulfate reducers, this process is also controlled by the supply of organic material in the sediments. The source of this organic matter may be from labile, nutrient-rich phytoplankton material or more refractory, comparatively nutrient-poor macrophyte material.

Extensive marsh areas and submerged aquatic vegetation produce large amounts of litter that decompose in the sediments in certain areas of the Bay. In the middle section of the Bay, 33% of the organic carbon flux is reported to be from submerged macrophytes (Kemp et al. 1984). One important submerged macrophyte in the Chesapeake Bay is the seagrass Zostera marina. Zostera senesces during summer and in the late fall producing substantial wrack deposits. Macrophytes such as Spartina alterniflora found in salt marshes fringing significant areas of the Bay senesce in the fall and are washed off during the spring and summer. In addition, terrestrially derived macrophyte litter, such as deciduous leaf material, may be entrained into the estuary through river discharge.

Thus, macrophyte material provides a substrate for microbes in the sediment. The rate of decomposition of this organic matter is affected by its chemical composition and structure. Since mineralization of the organic carbon in macrophyte material requires oxygen, plant material with higher nitrogen content will decompose faster, develop a large microbial community, and result in higher oxygen consumption rates.

To evaluate the relative significance of macrophyte litter and seston/water column oxygen consumption in the consumption of oxygen in the Chesapeake Bay estuary, field and laboratory experiments were conducted to measure:

a) oxygen consumption associated with decomposing macrophyte litter
b) oxygen consumption in the seston or water column.

In this paper, evidence will be given that the rate of oxygen consumption in macrophyte litter is lower, compared to oxygen consumption due to phyto-plankton and bacterioplankton respiration in the water column. Oxygen consumption rates in macrophyte litter will be shown to be affected by the chemical composition and structure of material. The fractionation of oxygen consumption to size-classes representing phytoplankton and bacterio-plankton and dependence of bacterio-plankton on phytoplankton production will be demonstrated.

METHODS

The study was conducted along the lower 10-mile segment of the York River during the summer of 1986 and 1987. To obtain decomposing samples of macrophyte litter of known age, litterbags containing Zostera marina (green blades), Spartina alterniflora (standing dead leaves and stalks) and Quercus alba or white oak (fallen leaves) material were deployed in June, 1986. These three species were used to determine the effect of plant structure and composition on rates and patterns of oxygen consumption.

Litterbags were deployed at three subtidal sites, Guinea Marshes, Allen's Island, and Mumfort Island (Figure 1). At intervals over three months, three replicate litterbags of each species were retrieved from each site. The remaining litter was cleaned and aliquots incubated for six hours at ambient temperatures in river water which had been filtered through glass fiber filters (GF/C).

To estimate water column or seston oxygen consumption rates, replicated water samples were collected from the three sites in the summer of 1987. In addition, consumption measurements were made at two mid-channel sites: Sarah Creek and Gloucester Pt. Water samples were incubated in the dark at 20°C for 24-48 hours. Seston densities were determined by filtering water samples through GF/C filters and analyzing for organic content using the method of weight loss on ignition.

Bottle effects may increase as the length of incubation increases. Therefore, to determine the effect of incubations time on consumption measurements, consumption rates from the 24-48 hour incubations were compared with measurements in short-term incubations. In addition, consumption rates were compared to net productivity of the water column using the light and dark bottle incubation method. Water samples from the shoal and mid-channel sites

were incubated for 6-8 hours. Size-fractionated incubations of water samples were conducted to determine the partitioning of oxygen consumption between bacterio-plankton(<1 micron) and zoo/phytoplankton (>1 micron).

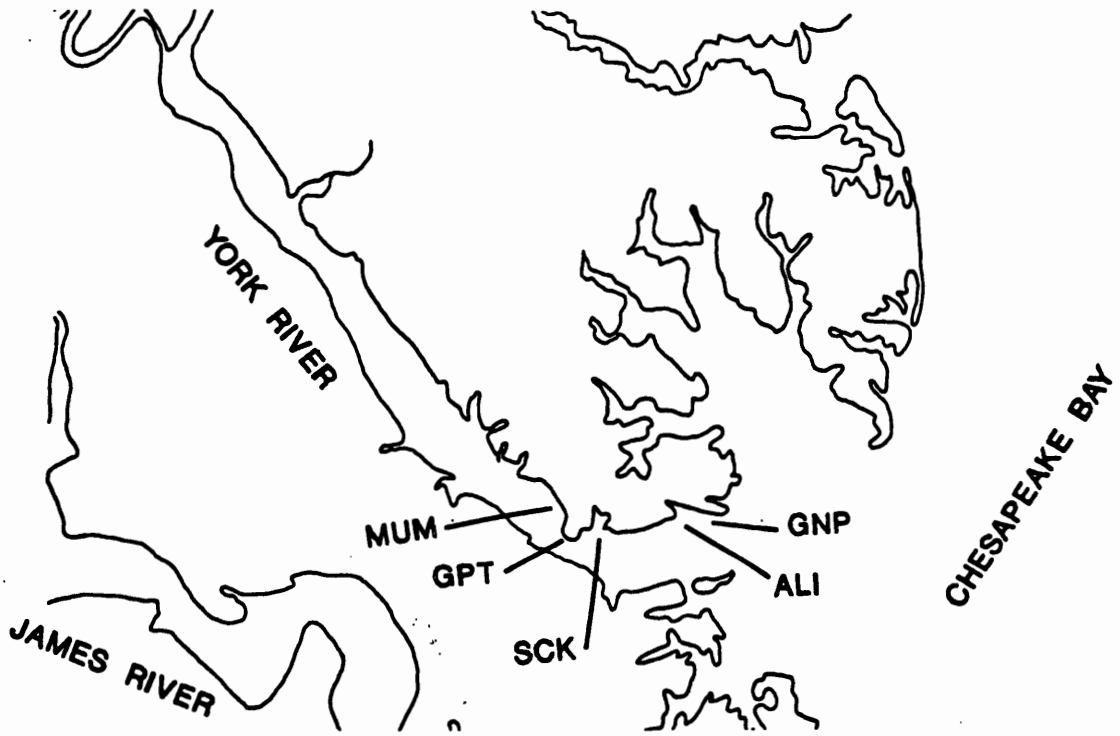

Figure 1. York River sample sites: Guinea Marshes (GNP); Allen's Island (ALI); Sarah Creek (SCK); Gloucester Point (GPT; Mumfort Island (MUM).

A series of light and dark bottle incubations of samples from a transect from the shoals to the channel at Guinea Marshes was also conducted to check the observed differences in consumption and production rates between shoal and channels areas. To further test the relative significance of macrophyte, water column and sediment oxygen consumption, in-situ oxygen metabolism measurements were made using 2.5 l plexiglass chambers in shaded and unshaded conditions.

RESULTS AND DISCUSSION

The results of the incubations of litter from the Guinea Marsh site are summarized in Figure 2. Oxygen consumption rates in Zostera litter (1.5-4 mg O_2/g AFDW/h), increased over the course of decomposition and was the highest rate measured for the three macrophyte species. Consumption rates in Spartina and oak litter were similar to each other (0.1-1.0 mg O_2/g AFDW/h) and were not as variable as Zostera through the course of decomposition. At the Allen's Island and Mumfort Island sites, the results were similar, in that consumption rates for Zostera were higher than that of Spartina and oak. Over the course of the study, oxygen consumption rates for decomposing macrophytes ranged from 0.5 to 4 mg O_2/g AFDW/h.

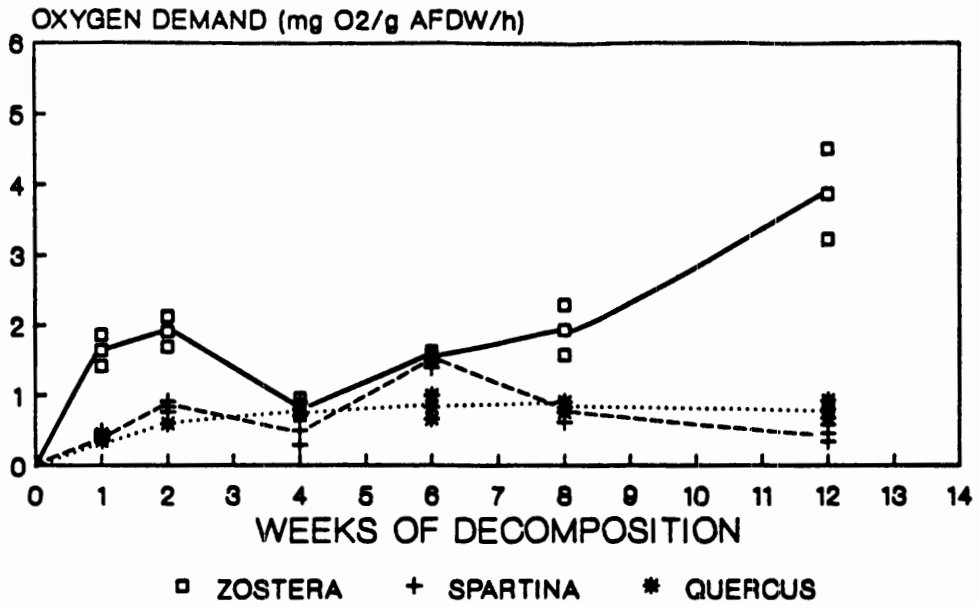

Figure 2. Oxygen consumption rates in Zostera, Spartina and Quercus litter decomposing at the Guinea Marshes site. Points along a line are means with upper and lower standard errors of 3 replicates.

The rate of oxygen consumption is affected by the size and activity of the microbial populations in the sample. When the substrate is optimal for microbial colonization, a larger and more active microbial community will develop. Materials high in structural matter, that require more energy to metabolize, will be colonized by a smaller and more specialized microbial community which possesses the enzyme systems needed to breakdown the more complex refractory organic matter. Materials that contain adequate nitrogen to mineralize the carbon will support a larger microbial community, resulting in higher oxygen consumption rates. Microbial activity and oxygen consumption rates can be stimulated in macrophyte material containing low nitrogen by amendments of nitrogen (Almazan and Boyd 1978). Zostera litter contains less structural material and has a higher nitrogen content than Spartina and oak. The higher oxygen consumption rates measured in Zostera litter compared to Spartina and oak correlate well with observations of faster decomposition rates in Zostera litter compared to Spartina and oak (Lagera et al. in prep.).

Figure 3 summarizes the data from the oxygen consumption measurements of the seston/water column samples. All of the sites showed water column consumption rates which were higher than 4 mg O_2/g AFDW/h, the highest consumption rate in macrophyte litter. These estimates are conservative since the incubations were conducted at 20°C while ambient water temperatures were higher. Weight-based consumption rates did not vary systematically upstream. This

observation contrasts with the consumption measurements calculated on a volume basis. On a volume basis, an increase in consumption rates was seen in the upstream sites. Measurements made from samples collected at Guinea Marshes in June (0.03 mg O_2/l/h) were significantly lower than in samples from Mumfort Island (0.06 mg O_2/l/h). The consumption rates estimated in this study are comparable with rates reported by Kemp et al. (1987) from surface waters of the mid-Chesapeake (0.01, 0.02 and 0.03 mg O_2/l/h in April, May and August, 1986). Similarly, Tuttle et al. (1986) reported consumption rates from cross-Bay transects conducted in 1985 which are comparable (0.01-0.1 mg O_2/l/h).

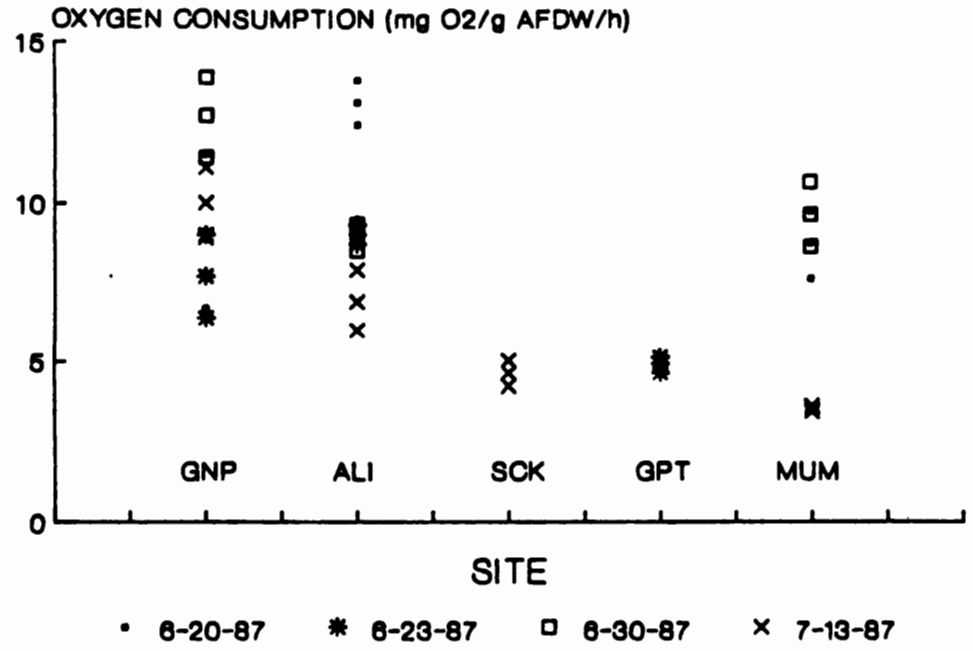

Figure 3. Seston oxygen consumption by site.

The short-term light and dark bottle incubations showed water column consumption rates which were comparable with the measurements made from extended incubations reported above. At all sites, consumption was, on the average, 30% of gross production. Net production was more variable than consumption rates and was higher and more variable upstream. Increased oxygen consumption upstream could be explained by higher concentrations of suspended organic matter. If bacterioplankton abundance and productivity, and consequently, oxygen consumption, is dependent on dissolved and suspended organic matter, consumption rates should be higher in samples containing greater organic matter.

The oxygen consumption measurements for shoal areas on the average were higher than those of the channel sites. The observations of higher consumption rates in shoal areas were supported by the shoal-channel transect study. The consumption and net production rates varied along the transect

from the shoal to channel at Guinea Marshes. Water column consumption was highest in the shoals while net production was highest in the channel. These agree with the observations reported by Tuttle et al. (1987) of higher bacterial abundance and production in the shallow flanks of the mainstem of the Bay. Tuttle et al. (1987) also observed that the concentrations of phytoplankton and labile organic matter were also higher along the flank areas. Increased phytoplankton abundance and productivity supports higher bacterial production. In shoal areas where turbulence is higher, resuspension of sedimented material, including benthic heterotrophs, is higher. Consequently, the proportion of the heterotrophs making up the water column community is greater, resulting in higher consumption rates and lesser net production.

The size fractionated incubations revealed that 50-75% of the oxygen consumption could be attributed to the consumption associated with the >1 micron fraction. This fraction includes zooplankton, phytoplankton, and bacteria associated with particles greater than 1 micron. Of the <1 micron fraction, at least 70% of oxygen consumption could be attributed to particles greater than 0.45 microns. Kemp et al. (1987) estimated that in surface waters in the mesohaline reach of the Chesapeake, 45-70% of the consumption could be attributed to the <3 micron fraction.

Raw and size-fractionated incubations of water samples from red tide eddies or phytoplankton blooms occurring in the York River in September, 1987 revealed higher consumption rates (0.17 mg O_2/l/h) compared to non-red tide waters (0.06 mg O_2/l/h). The proportion of oxygen consumption associated with particles less than 0.45 microns in size, that is the bacterial fraction, was much higher (>80%) than in non-red tide waters (<50%). This data supports the hypothesis that high phytoplankton production supports and enhances bacterial abundance and production. The consumption rates in 1 micron filtered red tide samples were significantly higher than in the incubations of the raw or unfiltered non-red tide samples.

If the hypothesis that bacterial abundance and production in the Chesapeake is dependent on phytoplankton production is correct, blooms of phytoplankton would be expected to have a higher proportion of heterotrophic populations compared to non-bloom waters. Phytoplankton blooms are characterized by large releases of dissolved organic carbon that can fuel heavy growth of bacteria. Since high consumption rates are associated with increased bacterial abundance, consequently, higher consumption rates would be expected in these bloom waters. In addition, the proportion of oxygen consumption attributable to the bacterial fraction (<1 micron) would be greater.

The uncoupling of the phytoplankton-zooplankton link in the Chesapeake Bay due to anoxic conditions is also postulated as a mechanism for the development of the unusually large bacterioplankton populations in the Bay. Anoxic conditions can prevent the normal diel vertical migrations that zooplankton exhibit resulting in increased predation and the extirpation of zooplankton eggs as they sink into anoxic bottom waters The inability of zooplankton to effectively graze on phytoplankton provides bacterioplankton with an larger source of substrate as phytoplankton die and decompose (Roman 1987). Sellner et al. (1987) reported that in August, 1986 microzooplankton assemblages consumed 21-53% of the available phytoplankton carbon in mid-Chesapeake Bay. The balance of the carbon could then provide a substrate for

bacterial metabolism and growth. This may explain the reported rates of carbon deposition in the Bay of about 30-60% of areal phytoplankton production. Ducklow and Peele (1987) reported that on the average 70% of phytoplankton production in the Bay is metabolizable by bacteria.

The effect of grazing and organic enrichment on the consumption of oxygen in the water column is suggested in a set of size-fractionated incubations. A set of bottles was incubated for 42 hours with unfiltered samples, 1 micron filtered samples and 1 micron filtered samples with the GF/C filters incubated in the bottles. This last set of incubations was conducted to determine the effect of physically removing grazing organisms from the water samples but still retaining their metabolic activities within the samples. The death and decomposition of organisms caught on the filters would also provide a rich substrate for microbial growth. Consumption rates in the samples with the GF/C filters included were significantly higher (140%) than in the unfiltered samples while the 1 micron filtered samples showed consumption rates which were less than 50% of the unfiltered samples. Owing to the confounding effects of nutrient enrichment on bacterial abundance and productivity, additional experiments need to be conducted to conclusively quantify the effect of grazers on the consumption of oxygen in the water column.

Water column consumption rates estimated from the in situ incubations using plexiglass chambers were comparable to those obtained by using BOD bottle incubations. The in situ incubations produced a benthic consumption rate of 0.5 g O_2/m^2/day. These are within the range of measurements of benthic consumption previously conducted in the York River. During the summer in 1976, in situ measurements of 0.9-3.4 g O_2/m^2/day were reported (Hyer 1977). Rizzo (1987) reported a mean sediment oxygen consumption rate of 0.8 g O_2/m^2/day for shoal areas of the York River. These values are also comparable with measurements made in the mainstem of the Bay. At the middle reaches of the Bay, benthic consumption rates in April, 1986 were 1.0 g O_2/m^2/day at a 10 m depth and 0.8 g O_2/m^2/day at 25 m depth (Kemp et al. 1987).

In this study, the effect of litter on net production and consumption in the water column and sediment was apparent from the in situ incubations. The presence of litter on the sediment surface increased consumption rates by over 100% while it decreased net production by about 50%. Macrophyte litter on the sediment surface can increase oxygen consumption because of the associated heterotrophs and can decrease net production by shading the benthic flora.

CONCLUSION

A number of observations of relevance to the problem of anoxia in the Chesapeake Bay were made in this study. Oxygen consumption rates in macrophyte litter were shown to be lower than in bacterio- or phyto-plankton respiration. This can be explained by the greater proportion of more labile, readily metabolizable, nutrient-rich substrate in phytoplankton material. Higher rates of water column oxygen consumption rates were also observed in upstream sites where the concentration of suspended organic matter and samples from shoal areas were higher. Bacterial oxygen consumption associated with the higher concentrations of suspended organic matter in upstream sites and over shoal areas explain the higher rates of oxygen consumption.

Heavy blooms of phytoplankton associated with high bacterial populations exhibited higher oxygen consumption rates supporting the view that bacterial populations in the Chesapeake Bay are dependent on phytoplankton production. Preliminary observation on the effect of grazers on oxygen consumption rates suggest further research along these lines be pursued.

The study supports the prevailing opinion that phytoplankton decomposition or water column oxygen consumption is the dominant source of oxygen demand in the Chesapeake Bay compared to macrophyte litter. However, decomposing macrophyte litter may be important in consumption of oxygen in certain cases or areas where litter may accumulate such as shallow embayments. In these areas, macrophyte litter can increase benthic consumption rates. Because macrophyte litter decomposes over a longer period of time (the turnover rate for phytoplankton is much faster than macrophyte litter), over time the cumulative oxygen demand of macrophyte litter could be comparable.

Estimates of water column and benthic consumption rates from this study were used with data in the literature in an order-of-magnitude calculation and sensitivity analysis to evaluate the relative importance of water column and benthic oxygen consumption (including macrophyte litter) in the development of anoxia. This exercise suggests that due to the relatively consistent values derived for oxygen consumption measurements for the water column and sediments, the accuracy of estimates of Bay-wide oxygen consumption may be more dependent on the rigorous calculation of the total volume of water (i.e. depth, hypsographic profiles and cross-sectional areas) used in the estimates.

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LITERATURE CITED

Almazan, G. and C.E. Boyd. 1978. Effects of nitrogen levels on rates of oxygen consumption during decay of aquatic plants. Aquat. Bot. 5:119-126.

Ducklow, H. and E. Peele. 1987. Bacterial carbon pools and fluxes in Chesapeake Bay plankton. In: G.B. Mackiernan, ed. Dissolved oxygen in the Chesapeake Bay. Maryland SeaGrant, College Park, Maryland. pp. 81- 85.

U.S. Environmental Protection Agency. 1982. Chesapeake Bay Program technical studies: as synthesis. U.S. Govt. Printing Office, Washington, D.C. 635 pp.

Hyer, P.V. 1977. Water quality model of York River, Virginia. Special Report No. 146. Virginia Institute of Marine Science, Gloucester Pt., Virginia.

Kemp, W.M., W.R. Boynton, R.R. Twilley, J.C. Stevenson, L.G. Ward. 1984. Influences of submersed vascular plants on ecological processes in upper Chesapeake Bay. In: V.S. Kennedy, ed. The estuary as a filter. Acad. Press. pp. 367-394.

Kemp, W.M., P. Sampou and W.R. Boynton. 1987. Relative roles of benthic versus pelagic oxygen-consuming processes in establishing and maintaining anoxia in Chesapeake Bay. In: G.B. Mackiernan, ed. Dissolved oxygen in the Chesapeake Bay. Maryland SeaGrant, College Park, Maryland. pp. 103-106

Lagera, L.M., J.C. Zieman, L.K. Blum and A.L. Mills, in prep. Decomposition of macrophytes in a temperate estuary: The effect of plant composition on decomposition rates.

Officer, C.B., R.B. Biggs, J.L. Taft, L.E. Cronin, M.A. Tyler and W.R. Boynton. 1984. Chesapeake Bay anoxia: origin, development and significance. Science 223:22-27.

Rizzo, W.M. 1987. The community metabolism and nutrient dynamics of a shoal sediment in a temperate estuary, with emphasis on temporal scales of variability. PhD. Dissertation. Virginia Institute of Marine Science, College of William and Mary, Virginia. 113 pp.

Roman, M. 1987. Physical and biological processes regulating anoxia in Chesapeake Bay: Zooplankton dynamics. In: G.B. Mackiernan, ed. Dissolved oxygen in the Chesapeake Bay. Maryland SeaGrant, College Park, Maryland. pp. 91-95.

Seliger, H.H., J.A. Boggs and W.H. Biggley. 1985. Catastrophic anoxia in the Chesapeake Bay in 1984. Science 228:70-73.

Sellner, K.G., D.C. Brownlee and L.W. Harding Jr. 1987. Implications of microzooplankton grazing on carbon flux and anoxia in Chesapeake Bay. In: G.B. Mackiernan, ed. Dissolved oxygen in the Chesapeake Bay. Maryland SeaGrant, College Park, Maryland. pp. 86-90.

Taft, J.L., W.R. Taylor, E.O. Hartwig and R. Loftus. 1980. Seasonal oxygen depletion in Chesapeake Bay. Estuaries 3:242- 247.

Tippie, V.K. 1984. An environmental characterization of Chesapeake Bay and a framework for action. In: V.S. Kennedy, ed. The estuary as a filter. Acad. Press. pp. 467-487.

Tuttle, J.H., T.C. Malone, R.B. Jonas, H. Ducklow and D. Cargo. 1986. Nutrient, dissolved oxygen dynamics: roles of phytoplankton and microheterotrophs under summer conditions, 1985. Draft Final report to U.S. Environmental Protection Agency, Chesapeake Bay Office, Annapolis, Maryland.

Tuttle, J.H., R.B. Jonas and T.C. Malone. 1987. Origin, development and significance of Chesapeake Bay anoxia. In: S.K. Majumdar, L.W. Hall Jr. and H.M. Austin, eds. Contaminant problems and management of living Chesapeake Bay resources. The Pennsylvania Academy of Science.

Understanding the Estuary: Advances in Chesapeake Bay Research. Proceedings of a Conference. 29-31 March 1988. Baltimore, Maryland. Chesapeake Research Consortium Publication 129. CBP/TRS 24/88.

Ecological Changes in Chesapeake Bay: Are They The Result of Overharvesting the American Oyster, *Crassostrea virginica*?

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ABSTRACT

Standing stocks of the American oyster (Crassostrea virginica) in Chesapeake Bay were as high as 188 x 10^6 Kg dry tissue prior to the major harvests of the late 19th Century. Due to continued overfishing and the effects of disease, the population has declined to a current level of about 1.9 x 10^6 Kg dry tissue. Crassostrea virginica is a suspension feeder, removing suspended organic and inorganic particles >3 μm in diameter with high efficiency. As it only assimilates approximately 70% of the filtered organic material, its dense mucus-bound biodeposits can provide a food resource for benthic organisms. Thus the oyster can form an important link between pelagic and benthic food webs.

It is possible that this decline in oyster biomass, and consequent reduction in their cropping of phytoplankton populations, may be a strong contributing factor to the apparent shift to microbial food webs and resultant summer anoxia in the deeper waters of the Bay. In addition, it is possible that planktonic suspension-feeding organisms, such as copepods and microzooplankton, have increased in abundance in response to this reduction in competition for phytoplankton. Such an increase in zooplankton would be expected to lead to a rise in the biomass of their predators, such as ctenophores and jellyfish. An increase in the oyster population by management and aquaculture could significantly improve water quality by removing large quantities of particulate carbon.

INTRODUCTION

The development during the summer of brief periods of anoxia in the bottom waters of Chesapeake Bay was first reported by Newcombe et al. (1939) in the 1930's. Since then, the duration and extent of such anoxic conditions may have expanded until by the mid-

1980's much of the water beneath the pycnocline is anoxic for extended periods during the summer (Officer et al. 1984). Such a deterioration in water quality in the Bay imposes a stress on the biota, which is usually fatal to benthic invertebrates, including many commercially valuable species. Thus, over the last decade a considerable amount of research has been directed toward understanding the causes and consequences of these changes in water quality (Taft et al. 1980, Officer et al. 1984). The general consensus from these investigations is that an increase in inorganic nutrient inputs, from both point and non-point sources, has led to enhanced levels of primary production throughout the Bay. It is the utilization and respiration of these abnormally high levels of organic material by the benthos and heterotrophic bacteria that reduces the dissolved oxygen concentration in the bottom waters (Kemp and Boynton 1984), and when a strong summer pycnocline prevails, anoxia can become very widespread (Malone et al. 1986, Officer et al. 1984). Therefore, in 1987 legislation to reduce nutrient inputs into the Chesapeake Bay by 40% was enacted which, it is thought, will reduce many of the consequences of eutrophication.

Extant benthic invertebrate populations are now recognized to be extremely important in nutrient recycling and benthic pelagic coupling (Rhoads 1974, Boynton et al. 1980, Dame et al. 1980) and molluscan suspension feeders may even act as a natural control on the adverse effects of eutrophication in estuaries (Cloern 1981, Cohen et al. 1984, Officer et al. 1982). The objective of this paper is to explore the possibility that the effects of nutrient enrichment on the Chesapeake Bay ecosystem have been exacerbated by the decline over the last century in the American oyster population. In an effort to quantify the role of oysters in Chesapeake Bay, I have compared the time it would take the oyster populations at their pre-exploitation densities, and in 1988, to filter the entire volume of the Bay. This method has previously been used to demonstrate the role of extant populations of suspension feeders in other estuaries (e.g. Dame et al. 1980, Biggs and Howell 1984). I have then taken these calculations a stage further by determining what proportion of the current daily primary production the original oyster populations could have removed from the water column. From this information I have calculated what proportion of the carbon that could have been used by the original high densities of oysters is now available for other organisms, such as pelagic suspension feeders.

OYSTER STANDING STOCKS

The American oyster, Crassostrea virginica is a well adapted estuarine species that produces large oyster beds in localities with salinities that range from 5 to 30 ppt (Galtsoff 1964). The oyster also requires firm substrates to form substantial populations; thus it is incapable of living on the softer muds often found in the >9 m deep channels of both the main-stem Chesapeake Bay and upper bay tributaries, except in certain exceptional localities (e.g. at 37-40 m near Point Patience in the Patuxent river [Kennedy and Breisch 1981]). The oyster was once so abundant in Chesapeake Bay that 18th century contemporary accounts suggest that oyster reefs were a significant navigational hazard (Wharton 1957). The subsequent decline in abundance is thought to be primarily due to continuous overexploitation (Kennedy and Breisch 1981) and, secondarily, due to the effects of disease.

Despite the economic importance of the oyster there is little reliable information on oyster biomass in Chesapeake Bay. Instead, I have estimated biomass from information on oyster landings in Maryland and Virginia, although it is widely regarded that oyster landings are significantly under-reported (Krantz and Haven 1982, Stagg 1985). Also, harvest statistics only give information for commercial size oysters (>7 cm) and are subject to other errors that give rise to inconsistences between supposedly identical records (Haven et al. 1978).

The most complete record for oyster landings in Maryland were compiled by Kennedy and Breisch (1981). From their data (Figure 1) I have calculated that 304 x 10^6 Maryland bushels of oysters were harvested between 1870 and 1895. Contemporary surveys of the major oyster growing regions during the late 19th Century show that there was a precipitous decline in oyster stocks and an irreversible change in many oyster beds as a result of the harvest activities, such that many were no longer conducive to oyster larval settlement (for review see Kennedy and Breisch 1981). After these large harvests, landings declined to a fairly stable level of about 3 x 10^6 bushels a year for the next four decades. The stability of these subsequent harvests suggest that this was the annual level of recruitment to the population and the earlier annual harvests of over 10 x 10^6 bushels were of a standing stock that had built up over many years in areas where recruitment occurred infrequently. Similarly, Davis et al. (1976) suggested that in certain localities within the Bay the abundance of oysters depends on a successful recruitment that occurs only every 10-15 years. Thus, if an annual recruitment of 3 x 10^6 bushels is subtracted from the landings for the 25-year period 1870-1895 it indicates that there was a standing stock of 229 x 10^6 bushels of oysters in Maryland prior to 1870. I used a yield factor of 2.64 Kg wet weight of oyster tissue per Maryland bushel (Haven et al. 1978), and that an oyster has an 80% water content (Newell, unpublished data), to calculate that the dry weight biomass of oysters in Maryland prior to the major harvests was 120.5 x 10^6 Kg. Unfortunately, there is no information available on oyster landings in Virginia prior to 1880. Thus, oyster stocks cannot be calculated from reported landings for the period of the major harvest. Instead, I have calculated from National Marine Fisheries Service statistics (Figure 1) that in 1880, 1888, 1890 and 1891 the Virginia landings were, on average, 56% of those in Maryland. Assuming that the fishing pressure on the oyster stocks during this period was the same between the two states, the standing stock in Virginia would be 56% of the calculated value for Maryland i.e. 67.4 x 10^6 Kg dry oyster tissue.

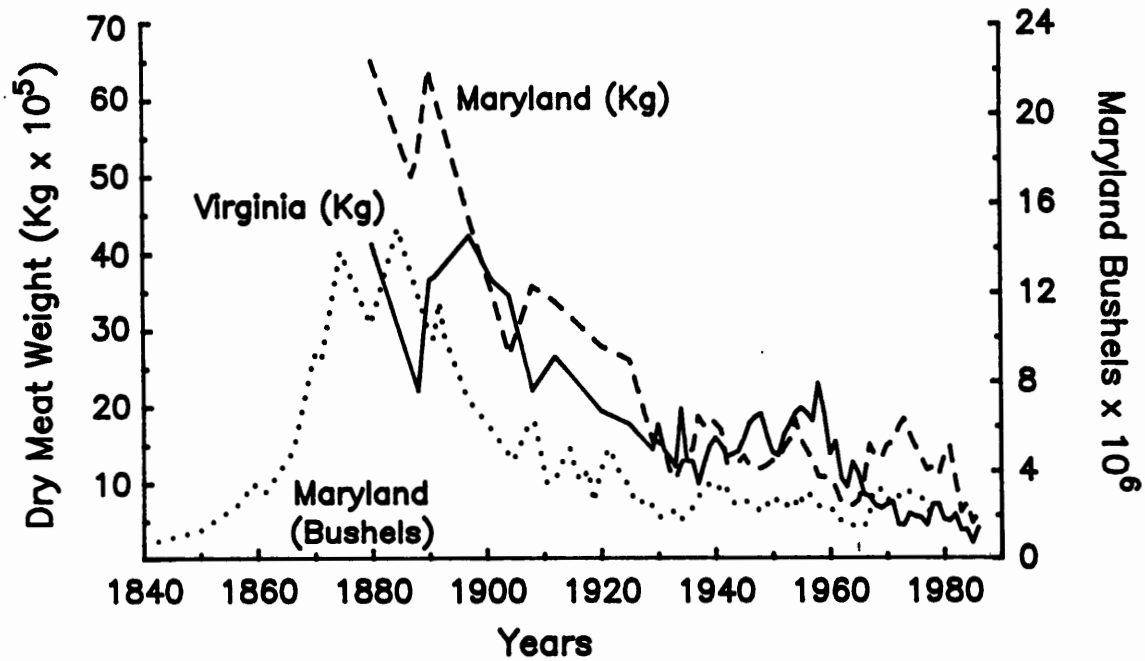

Figure 1. Oyster landings (dry tissue weight) for Maryland (- - -) and Virginia (——) [source: NMFS statistics for oyster landings, converted to dry weight: see text for details] and bushels x 10^6 landed in Maryland (. . . .) (Redrawn from Kennedy and Breisch 1981).

A second method for estimating pre-harvest oyster stocks is to use an estimate of what percentage of the total population was being harvested annually and apply this to the data on landings. In the early 1970's this exploitation rate was extremely high, reaching over 30%, because the oyster stocks were small (Cabraal 1978). In order to obtain a conservative estimate of the oyster biomass I have applied a moderately high exploitation rate of 10% to the peak harvest data of 1880 for Maryland and Virginia (Figure 1) to calculate that in 1880 Maryland had a stock of 65.3 x 10^6 Kg and Virginia 41.4 x 10^6 Kg dry tissue.

By 1975 the oyster stocks in Maryland had declined to 5.26 x 10^6 Kg dry tissue (Cabraal 1978) and in 1988 are about 1.05 Kg dry tissue (pers. comm., Dr. S.J. Jordan, Maryland Department of Natural Resources). The Virginia oyster population in 1988 was estimated to be about 1.5 x 10^6 bushels (0.85 x 10^6 Kg dry tissue) (pers. comm., Dr. R. Mann, Virginia Institute of Marine Science).

ESTIMATION OF THE FEEDING ACTIVITY OF CRASSOSTREA VIRGINICA

The oyster is an active suspension feeder, removing particles > 3 μm with high efficiency (Haven and Morales-Alamo 1970). Subtidal oysters in the Chesapeake Bay feed for over 23 h per day with no diurnal rhythm (Newell, unpublished data). Feeding activity varies seasonally but rates of 5 l . h^{-1} . g^{-1} dry tissue weight are typical for oysters during the summer (Newell, unpublished data). Thus oysters have an average summer filtration rate of 0.115 l . kg^{-1} . d^{-1}.

For Chesapeake Bay, and Maryland's portion of the Bay, the total volumes (71.5 x 10^9 and 27.3 x 10^9 m^3, respectively), and volumes <9m in depth (53.8 x 10^9 and 17.8 x 10^9 m^3, respectively), of the regions with salinities >5 ppt (in which oysters flourish), were calculated from Cronin and Pritchard (1975).

The pre-1870 oyster populations in the Chesapeake Bay could potentially filter the entire water column during the summer in less than 3 to 6 days and the <9m water every 2 to 4.5 days (Table 1). However, with the current oyster stocks, these turnover times have dramatically increased to 325 days and 244 days, respectively (Table 1).

Table 1. Number of days required by the oyster population, prior to the major harvests (pre-1870 and 1880 oyster stock estimates) and today, to filter both the entire water column and the water shallower than 9 m, both in Maryland's portion of the Bay and in the entire Chesapeake Bay.

Date	Oyster Biomass (Kg x 10^6 Dry Tissue Wt)	Volume of Water Filtered (L x 10^{12} d^{-1})	Turnover Time (d) Total Water Column	<9m Zone
		Maryland's Portion of Chesapeake Bay		
Pre-1870	120.5	13.86	2.0	1.3
1880	65.3	7.51	3.6	2.4
1975	5.3	0.61	44.8	29.2
1988	1.1	0.12	227.5	148.3
		Entire Chesapeake Bay		
Pre-1870	188.0	21.62	3.3	2.5
1880	106.8	12.27	5.8	4.4
1988	1.9	0.22	325.0	244.5

THE ROLE OF OYSTERS IN CARBON CYCLING

Harding et al. (1986) measured the daily rate of carbon production by phytoplankton along the main stem of Chesapeake Bay in the spring, summer and fall of 1982. From their data I have calculated an average carbon production of 1.08 $\pm$ 0.25(S.E.) g C . m^{-2} . d^{-1} (range 0.14 - 2.65) for the >5 ppt regions of the Bay where oysters can grow. Malone et al. (1986) reported a slightly higher rate of phytoplankton carbon production of 1.44 g C . m^{-2} . d^{-1} during the summer and early fall of 1984 in the main stem of the Bay near the mouth of the Choptank river.

The total area of Chesapeake Bay with salinities >5 ppt is 10.7 x 10^9 . m^{-2} and the area within Maryland is 3.66 x 10^9 . m^{-2} (Cronin 1971). Thus, using the value of 1.08 $\pm$ 0.25(S.E.) g C . m^{-2} . d^{-1} (Harding et al. 1986), the average total daily planktonic carbon production during 1982, in the entire Bay, and in Maryland's portion, was calculated to be 11.5 g x 10^9 g and 3.95 x10^9 g, respectively. Malone et al. (1984) reported that the euphotic zone ranged in depth from 3 to 6 m and was confined to the mixed layer above the pycnocline. Therefore, in order to convert the areal daily carbon production to carbon per litre I have assumed that phytoplankton production is confined to waters shallower than 9 m waters (This is generally also the maximum depth in which oysters live in the Bay). Thus, there is an average daily carbon production of 2.2 x 10^{-4} g . l^{-1} in the Chesapeake Bay.

The metabolic carbon requirements of an oyster feeding at 5 l . h^{-1} . g^{-1} are 0.537 mg C respired h^{-1}, calculated from its rate of oxygen consumption of 1 ml O_2 . h^{-1} . g^{-1} (Newell and Langdon 1986). Assuming that respiratory carbon demands are 75% of total carbon requirements (Bayne and Newell 1983), the oyster's total carbon requirements are 17.3 mg C . d^{-1}. Using these estimates of carbon requirements, together with those for 1982 phytoplankton carbon production, the pre-1870 oyster stocks in Maryland would have been capable of removing 77% of the 1982 daily carbon production in the <9 m waters (Table 2); about 70% of the filtered carbon would be used to satisfy their metabolic demands while the other 30% would have been egested as compacted biodeposits available to the benthic food web (Table 2). Even the conservative estimate of the original oyster population in Maryland suggests that they could remove 42% of the 1982 daily carbon production from the <9 m water column. Overall, the entire population of oysters that existed in Chesapeake Bay prior to their heavy exploitation by man might have been capable of removing between 23% and 41% of the 1982 phytoplankton carbon production.

By 1975 the population of oysters that existed in Maryland's portion of Chesapeake would have been capable of removing only 3.4% of the 1982 daily carbon production in the <9 m waters. By 1988 the oyster population had declined to such an extent that it is only capable of removing about 0.7% of the daily carbon from Maryland's waters and 0.4% baywide (Table 2).

Unfortunately, it is difficult to assess from my calculations the relative impact of the pre-exploitation oyster populations on benthic pelagic coupling in the main stem of Chesapeake Bay, where about 35% of the area is >9 m deep (Pritchard 1952), compared to its shallower tributaries. Even in the main-stem, however, oyster densities were extremely high on both the western and eastern shores, with particularly extensive populations at the mouths of each tributary, and in Eastern Bay and Tangier Sound (Kennedy and Breisch 1981). Generally, oyster beds are concentrated in areas of high water flow and scour, such as occur along the edges of channels, where water movement is sufficient to bring in food and remove biodeposits (Lund 1957). This water circulation would have enhanced the ability of oysters to filter large amounts of particulate material, even from the main-stem Chesapeake Bay. In addition, the pre-

Table 2. For Maryland's portion of the Bay and the entire Chesapeake Bay, prior to the major period of oyster harvest (pre-1870 and 1880 oyster stock estimates) and today, a) percent of the daily phytoplankton carbon production filtered by oysters; b) weight of carbon filtered by oysters from the <9 m water column; c) the total carbon assimilated; and d) the amount of carbon, in the form of oyster biodeposits, sedimented. All calculations are based on 1982 daily carbon production estimates of 39.5 x 10^8 and 115.6 x 10^8 g carbon for Maryland's portion of the Chesapeake Bay, and the entire Bay, respectively.

Date	% Daily Carbon Production Removed	Carbon Removed <9m Zone (g x 10^8 d^{-1})	Assimilated Carbon (g x 10^8 d^{-1})	Carbon Sedimented (g m^{-2} d^{-1})
		Maryland's portion of Chesapeake Bay		
Pre-1870	77.2	30.5	20.7	0.267
1880	41.7	16.5	11.2	0.145
1975	3.4	1.34	0.9	0.012
1988	0.7	0.26	0.2	0.002
		Entire Chesapeake Bay		
Pre-1870	41.2	47.6	32.2	0.144
1880	23.4	27.0	18.3	0.081
1988	0.4	0.48	0.3	0.002

exploitation oyster reefs were such large accretions of shell and living oysters that they must have enhanced turbulent mixing and created changes in the local water currents. Again this would enable the oyster community to intercept a larger proportion of the water flow, compared to other benthic suspension feeders that can only feed in the region of low flow associated with the benthic boundary layer.

ECOLOGICAL CHANGES IN CHESAPEAKE BAY

In Chesapeake Bay the extant macrobenthic invertebrates are both numerous and diverse, with many communities dominated by suspension feeding molluscs (Holland et al. 1987). Unfortunately, there is no direct data available to indicate whether the biomass of these communities has changed as a consequence of the decrease in the oyster population. However, currently one of the most abundant suspension feeding molluscs in the mesohaline regions is the wedge clam, Rangia cuneata. This species has only become a dominant member of the benthos in the last two decades, and was thought to be extinct on the east coast prior to 1955 (Hopkins and Andrews 1969). In the oligohaline to freshwater reaches of the Potomac river, the Asiatic clam, Corbicula fluminea, which was introduced in the last decade, is so abundant that its feeding activity causes a significant decrease in the phytoplankton population (Cohen et al. 1984). In contrast to the increase in some species of benthic invertebrates it is possible that in some localities the benthic biomass is lower today as a consequence of the reduction in oyster biodeposition. The feces and pseudofeces produced by the oyster contain a large proportion of organic matter (Newell and Jordan 1983) which can form an important energy source for bacteria and benthic invertebrates feeding at the sediment water interface (Haven and Morales-Alamo 1966, 1968; Jordan 1987).

Indeed, Holland et al. (1987) have speculated that benthic communities at some locations within Chesapeake Bay are currently food limited. Also, the original oyster beds provided a major source of firm substrate for other benthic invertebrates, such as mussels, barnacles, and tunicates that cannot live on the generally muddy bottom of the Bay.

Pelagic suspension feeders, such as zooplankton may have increased in abundance as a result of the decline in oyster stocks. Zooplankton, which are relatively short-lived and opportunistic animals, can respond rapidly to an increase in the supply of phytoplankton. Zooplankton and meroplankton in Chesapeake Bay can attain high densities and currently consume nearly 100% of the daily phytoplankton production during August in the mesohaline portion of the Bay (White and Roman 1988). This suggests that during the summer the zooplankton community is a major consumer of the phytoplankton population, a position that could once have been filled by the adult oyster and its planktotrophic larvae.

The major consumers of zooplankton are fish larvae, certain species of adult fishes, ctenophores (Mnemiopsis leidyi), and jellyfish, e.g., the sea nettle (Chrysaora quinquecirrha); the sea nettle also preys on the ctenophore. In a detailed review by Wharton (1957) of the historical literature available on colonial fisheries in Virginia there is little mention of the now-abundant sea nettle. Given the current attention paid to the sea nettle by fisherman and boaters it is plausible to equate the lack of reference in the historical diaries with the idea that sea nettles might not have been very abundant during that period. I speculate that the current high abundance of sea nettles may be due, in part, to an increase in their zooplankton food supply, which is a consequence of a reduction in oyster stocks. However, sea nettles have probably always been present within Chesapeake Bay, perhaps relying on the once abundant oyster larvae as a prey item.

It is currently thought that increased nutrient inputs and reduced light levels are encouraging the growth of small species of phytoplankton in Chesapeake Bay (for review see Verity 1987). There is no information available concerning the size structure of the phytoplankton community in Chesapeake Bay a century ago when oysters were abundant. However, oysters are capable of removing cells <3 μm, albeit with relatively low efficiency (Haven and Morales-Alamo 1970). In contrast, Acartia tonsa, which is usually the dominant pelagic suspension feeder during the summer (Heinle 1966, White and Roman 1988), cannot feed efficiently on cells <7 μm (Richman et al. 1977, Ryther and Sanders 1980). Thus, in systems such as Chesapeake Bay that are dominated by zooplankton herbivores, it is likely that there will be a shift to smaller phytoplankton species simply as a result of selective effects of zooplankton grazing (Richman et al. 1977, Ryther and Sanders 1980).

Oysters, in common with other bivalve molluscs, are efficient suspension feeders and can have a considerable impact on particle concentrations because of their ability to pump large volumes of water across the ciliated surfaces of their gills (Jorgensen 1966). They are also long-lived animals that overwinter and increase their feeding activity in the spring in response to rising water temperature. Thus the pre-harvest oyster population would have been an important consumer of the spring phytoplankton bloom. In contrast, the copepod Acartia tonsa is not able to overwinter as an adult in the Chesapeake Bay, due to low winter temperatures. Other members of the zooplankton population are meroplanktonic e.g., polychaete larvae which in August are second only to copepods in their consumption of phytoplankton (White and Roman 1988). As a consequence of the low zooplankton biomass in the spring a large proportion of the spring phytoplankton bloom currently remains ungrazed. It is the provision of this pulse of carbon to the benthic invertebrate and bacterial communities

beneath a strong pycnocline (when it develops) that is responsible for the anoxic conditions in the bottom waters of the Bay (Kemp and Boynton 1984).

In addition to their role in cropping phytoplankton populations, the pre-exploitation oyster populations may also have been important in controlling turbidity, by reducing both the particulate inorganic material (PIM) and organic material (POM) suspended in the water column. The oyster removes PIM and POM with high efficiency and ejects the undigested material as mucus-bound biodeposits. Thus, the grain size of the PIM is effectively increased, making the PIM more resistant to erosion and resuspension (Haven and Morales-Alamo 1968, Jordan 1987). Increased turbidities in the Bay that have occurred since colonial times have principally been attributed to anthropogenic factors (e.g., land clearing) increasing the rate of terrestrial erosion in the watershed, and a decline in the seagrass communities that not only trap suspended sediments but can also serve to stabilize sediments (Short and Short 1986). However, the decline in the oyster population over the last century may have exacerbated suspended particulate concentrations simply by reducing rates of biodeposition.

Currently the euphotic zone in Chesapeake Bay lies between 3-6 m deep (Malone et al. 1986) and although primary production is high it is generally considered to be light and not nutrient limited (Harding et al. 1986, Verity 1987). Thus phytoplankton production may also have been high prior to 1870 because of increased light penetration, as well as the rapid recycling of nutrients via the oysters metabolic processes (Rhoads 1974, Boynton et al. 1980). My calculations for the amount of carbon required by the pre-1870 oyster population in Maryland (Table 2) indicate that the oysters required about 52% of the 1982 daily carbon production to satisfy their own requirements. These calculations suggest that only 25% of the 1982 carbon production would be directly available, and another 25% indirectly in the form of oyster biodeposits, for all other herbivores and detrivores in Chesapeake Bay. This suggests that the Bay must have been an extremely productive system prior to 1870 in order to support both the oyster population and the other animal species. It is possible that it was even food limited on some occasions.

SUMMARY

From my calculations it is apparent that oyster stocks were once abundant enough to be the dominant species filtering carbon from the water column in Chesapeake Bay. Harding et al. (1984) estimated that phytoplankton doubling times during the summer in Chesapeake Bay are on the order of 0.8 to 4.3 days. This is very similar to the summer filtration times for the <9 m water column of 1.3-4.4 days that I estimated for the original oyster populations (Table 1). From the rationale of Officer et al. (1982), this suggests that the oyster could have exerted a major influence on the phytoplankton population in Chesapeake Bay. I believe that these calculations provide insights into the role that a flourishing oyster population once played in the ecology of Chesapeake Bay. Even if the figures are not absolute, the relative magnitude of the changes associated with the over-exploitation and decline of the oyster stocks cannot be ignored. Consideration of the former trophic role oysters may have played in the Chesapeake Bay ecosystem is instructive since it suggests a valuable biological role that restored oyster populations can play in future water quality restoration efforts. This is in addition to the well recognized importance of the oyster to commercial fisheries. If it is impossible to restore Crassostrea virginica, due to its susceptibility to two major oyster diseases (MSX and Perkinsus) that are present in the Bay, perhaps we should consider introducing a non-native oyster species, such as the Pacific oyster Crassostrea gigas. Officer et al. (1982) make the point that is the introduced species of bivalve molluscs in South San Francisco Bay that are controlling the detrimental effects of eutrophication. Restoring an abundant population of filter

feeding oysters to the shallow waters of Chesapeake Bay and its tributaries might really help lend new meaning to the phrase "Clean up the Bay".

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LITERATURE CITED

Bayne, B.L. and R.C. Newell. 1983. Physiological energetics of marine molluscs. In: The Mollusca, Vol 4, Physiology, Part I. A.S.M. Saleuddin, K.M. Wilbur, eds. Academic Press, New York, p.407-515.

Biggs, R.B. and B.A. Howell. 1984. The estuary as a sediment trap: Alternate approaches to estimating its filtering efficiency. In: The estuary as a filter. V.S. Kennedy, ed. Academic Press, Orlando, Fl. p. 107-129.

Boynton, W.R., W.M. Kemp and C.G. Osbourne. 1980. Nutrient fluxes across the sediment-water interface in the turbid zone of a coastal plain estuary. In: Estuarine Perspectives. V.S. Kennedy, ed. Academic Press, New York. p. 93-109.

Cabraal, R.A. 1978. Systems analysis of the Maryland oyster fishery: production, management, and economics. Ph.D. dissertation, University of Maryland, College Park. 318pp.

Cloern, J.E. 1981. Does the benthos control phytoplankton biomass in South San Francisco Bay? Mar. Ecol. Prog. Ser. 9:191-202.

Cohen, R.R.H., P.V. Dresler, E.J.P. Philips, and R.L. Cory. 1984. The effect of the asiatic clam, Corbicula fluminea on phytoplankton of the Potomac River, Maryland. Limnol. Oceanogr. 29:170-180.

Cronin, W.B. 1971. Volumetric, areal, and tidal statistics of the Chesapeake Bay estuary and its tributaries. Chesapeake Bay Institute. Special Report. No 20.

Cronin W.B. and D.W. Pritchard. 1975. Additional statistics on the dimensions of the Chesapeake Bay and its tributaries: Cross section widths and segment volumes per meter depth. Chesapeake Bay Institute. Special Report. No 42.

Dame, R., R. Zingmark, H. Stevenson and D. Nelson. 1980. Filter feeder coupling between the estuarine water column and benthic subsystems. In: Estuarine Perspectives. V.S. Kennedy, ed. Academic Press. New York. p. 521-526.

Davis, J., D. Haven, K.G. Drobeck and E.A. Dunnington. 1976. Plans for the Management of the Fisheries of the Tidal Potomac River. Univ. of Maryland, Solomons, Md. CEES Ref No.76-123CBL. 38pp.

Galtsoff, P.S. 1964. The American oyster Crassostrea virginica (Gmelin). U.S. Fish Wildl. Serv. Fish. Bull. 64:1-480.

Harding, L.W., B.W. Meeson and T.R. Fisher. 1986. Phytoplankton production in two east coast esturies: Photosynthesis-light functions and patterns of carbon assimilation in Chesapeake and Delaware Bays. Estuar. Coastal Shelf Sci. 23:773-806.

Haven, D., and R. Morales-Alamo. 1966. Aspects of biodeposition by oysters and other invertebrate filter-feeders. Limnol. Oceanogr. 11:487-498.

Haven, D., and R. Morales-Alamo. 1968. Occurrence and transport of faecal pellets in suspension. Sed. Geo. 2:141-151.

Haven, D., and R. Morales-Alamo. 1970. Filtration of particles from suspension by the American oyster Crassostrea virginica. Biol. Bull. 139:248-264.

Haven, D.S., W.J. Hargis and P.C. Kendall. 1978. The oyster industry of Virginia: its status, problems and promise. VIMS Spec. Papers Mar. Science No. 4 1024pp.

Heinle, D.R. 1966. The production of a calanoid copepod, Acartia tonsa, in the Patuxent estuary. Ches. Sci. 7:59-74.

Holland, A.F., A.T. Shaughnessy and M.H. Hiegel. 1987. Long-term variation in mesohaline Chesapeake Bay benthos: Spatial and temporal patterns. Estuaries 10:227-245.

Hopkins, S.H., and J.D. Andrews. 1969. Rangia cuneata on the east coast: Thousand mile range extension, or resurgence? Science 167:868-869.

Jordan, S.J. 1987. Sedimentation and remineralization associated with biodeposition by the American oyster Crassostrea virginica (Gmelin). Ph.D. Dissertation Thesis. University of Maryland, College Park.

Jorgensen, C.B. 1966. The Biology of Suspension Feeding. Pergamon Press, London. 357pp.

Kennedy. V.S. and L.L. Breisch. 1981. Maryland's Oysters: Research and Management. Maryland Sea Grant, University of Maryland, College Park, MD. 286pp.

Kemp, W.P. and W.R. Boynton. 1984. Spatial and temporal coupling of nutrient inputs to estuarine primary production: the role of particulate transport and decomposition. Bull. Mar. Sci. 35:242-247.

Krantz, G.E., and D.S. Haven. 1982. A study of the present state of oyster statistics in Chesapeake Bay and suggested remedial measures. In: Report of Workshop on Chesapeake Bay Fisheries Statistics. L.E. Cronin, ed., UMCEES-CBL. p.44-52.

Lund, E.J. 1957. Self silting by the oyster and its significance for sedimentation geology. Publ. Inst. Mar. Sci. Univ. Texas 4:320-327.

Malone, T.C., W.M. Kemp, H.W. Ducklow, W.R. Boynton, J.H. Tuttle and R.B. Jonas. 1986. Lateral variation in the production and fate of phytoplankton in a partially stratified estuary. Mar. Ecol Prog. Ser. 32:149-160.

Newcombe, C.L., W.A. Horne, and B.B. Sheperd. 1939. Studies on the physics and chemistry of estuarine waters in Chesapeake Bay. J. Mar. Res. 2:87-116.

Newell, R.I.E. and S.J. Jordan. 1983. Preferential ingestion of organic material by the American oyster Crassostrea virginica. Mar. Ecol. Prog. Ser. 13:47-53.

Newell, R.I.E., and C.J. Langdon. 1986. Digestion and absorption of refractory carbon from the plant Spartina alterniflora by the oyster Crassostrea virginica. Mar. Ecol. Prog. Ser. 34:105-115.

Officer, C.B., T.J. Smayda and R. Mann. 1982. Benthic filter feeding: A natural eutrophication control. Mar. Ecol. Prog. Ser. 9:203-210.

Officer, C.B., R.B. Biggs, J.L. Taft, L.E. Cronin, M.A. Tyler, W.R. Boynton. 1984. Chesapeake Bay Anoxia: Origin, development and significance. Science 223:22-27.

Pritchard, D.W. 1952. Salinity distribution and circulation in the Chesapeake Bay estuarine system. J. mar. Res. 11:106-123.

Rhoads, D. 1974. Organism-sediment relations on the muddy sea floor. Oceanogr. Mar. Biol. Ann. Rev. 12:263-300.

Richman, S., D.R. Heinle and R.Huff. 1977. Grazing by adult estuarine calanoid copepods of the Chesapeake Bay. Mar. Biol. 42:69-84.

Ryther, J.H. and J.G. Sanders. 1980. Experimental evidence of zooplankton control of the species composition and size distribution of marine phytoplankton. Mar. Ecol. Prog Ser. 3:279-283.

Short, F.T., and C.A. Short. 1986. The seagrass filter: Purfication of estuarine and coastal waters. In: The estuary as a filter. V.S. Kennedy, ed. Academic Press, Orlando, Fl. p. 395-413.

Stagg, C. 1985. An evaluation of the information available for managing Chesapeake Bay Fisheries: Preliminary stock assessments. Chesapeake Bay Commission, Md. p. 11-43.

Taft, J.L., W.R. Taylor, E.D. Hartwig, and E.D. Loftus. 1980. Seasonal oxygen depletion in Chesapeake Bay. Estuaries 3:242-247.

Verity, P.G. 1987. Factors driving changes in the pelagic trophic structure of estuaries, with implications for the Chesapeake Bay. In: Perspectives on the Chesapeake Bay: Advances in Estuarine sciences. M.P. Lynch, ed. Chesapeake Research Consortium, Gloucester Point, Va. p. 33-56

Wharton, J. 1957. The Bounty of the Chesapeake. Fishing in Colonial Virginia. Univ. Press of Virginia, Charlottesville, Va. 78pp.

White, J.R. and M.R. Roman. 1988. Grazing and egg production by Chesapeake Bay zooplankton in spring and summer. Abstract In: Understanding the Estuary: Advances in Chesapeake Bay Research. M. Lynch, ed. Chesapeake Research Consortium, Gloucester Point, Va.

Understanding the Estuary: Advances in Chesapeake Bay Research. Proceedings of a Conference. 29-31 March 1988. Baltimore, Maryland. Chesapeake Research Consortium Publication 129. CBP/TRS 24/88.

Oxygen Fluctuations and Fish Population Dynamics in a Chesapeake Bay Oyster Bed

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INTRODUCTION

Dissolved oxygen concentrations below 4 mg/L or 50% saturation are physiologically stressful to many fish species (4 mg/L = 50% saturation at approximately 27.5°C). Below these concentrations, larval survival, development and growth are negatively affected (Siefert and Spoor 1974; Rogers et al. 1982) and adult behaviors change (Magnuson et al. 1985; Kramer 1987). At nonlethal concentrations, low dissolved oxygen can decrease growth rates, feeding rates and reproductive activities and lead to changes in swimming behavior and habitat use that increase the risk of predation (reviewed in Kramer 1987).

Deep mesohaline areas of the mainstem Chesapeake Bay become anoxic during summer months. Wind-driven cross-Bay tilts of the pycnocline can bring anoxic or severely hypoxic waters onto the shallow flanks of the Bay (Carter et al. 1978; Malone et al. 1986). Although low dissolved oxygen conditions in the Bay are thought to negatively impact both finfish populations and shallow water epibenthic communities (Holland et al. 1980; Kemp and Boynton 1981; Coutant 1985; Price et al. 1985), few studies have directly addressed this hypothesis.

The current study was designed to establish the relationship between low and fluctuating oxygen levels in Chesapeake Bay oyster bars and recruitment and population structure of a benthic fish, Gobiosoma

bosci, the naked goby. The naked goby is a potentially important component of the Bay trophic structure. Our data indicate that periods of low dissolved oxygen concentrations, probably associated with tilting of the pycnocline, lead to decreases in naked goby populations and may cause mortality of recent recruits in moderate-depth and deep areas of oyster bars. Although no intrusions of anoxic water appear to have occurred at the study site during summer 1987, dissolved oxygen concentrations as low as 0.2 mg/L were recorded. Such concentrations are sufficiently low to cause 100% mortality of naked goby larvae in less than two hours (Saksena and Joseph 1972).

STUDY ORGANISM

The naked goby is a small benthic species that is abundant in oyster beds and seagrass habitats in estuaries and tributaries along the east and Gulf Coasts of the United States and Mexico. Naked gobies live in and among oyster shells and other hard substrates which they use both for shelter and as sites for egg attachment. Eggs are guarded by the male for several days, after which time larvae undergo a month-long planktonic stage (Breitburg unpublished data) before settling to the benthos. Settlement generally occurs at 9-12 mm total length (TL) (Breitburg unpublished data).

Tolerance of the naked goby to oxygen stress has been assessed only for the larval stage. Saksena and Joseph (1972) found that 50% of newly hatched naked goby larvae die within 24 h at 1.30 mg/L dissolved oxygen. Dissolved oxygen concentrations of 0.86 mg/L were lethal to 50% of larvae within 4 h.

There are no estimates of population size of benthic naked gobies within the Chesapeake Bay, probably because of the difficulty of remote sampling of cryptic benthic fishes in structurally complex habitats such as oyster beds. Our data from the Flag Ponds oyster bar indicate that densities of 10-50 individuals/m^2 of suitable habitat are not unusual. In contrast to benthic populations, planktonic naked goby larvae have been extensively sampled in the Chesapeake Bay and its tributaries. During the summer, naked goby larvae are typically first or second in abundance among fish larvae in Chesapeake Bay tributaries and nearshore waters (Dovel 1971; Wakefield 1977; Gallagher and Currence 1982). Because of their abundances, both benthic and pelagic stages of the naked goby are potentially important as predators and prey. For example, naked goby larvae are, at least at times, the most important fish prey of juvenile striped bass (Wass and Wright 1969; Markle and Grant 1970).

METHODS

Field studies were conducted in the Flag Ponds oyster bar near Camp Conoy, MD on the western shore of the Chesapeake Bay during summer 1987. The oyster bar ranges from approximately 2-6 m in depth and varies in bottom topography from low rock ledges and outcroppings to a continuous cover of unconsolidated oyster shell. Permanent stations were established in 2, 5 and 6 m of water. Rock outcroppings are more abundant at the 2 m site than at the deeper sites.

Hydrolab model 2040 DataSondes were used to monitor dissolved oxygen, temperature and salinity at 15 min intervals within 10 cm of the substrate at the 5-m site. Two DataSondes were rotated to obtain as continuous a data record as possible. Deployment was limited to $\leq$4 d to minimize the effects of fouling on measurements. There were 7 dates on which we obtained readings $\leq$15 min apart from both a DataSonde that had been deployed for several days and a newly calibrated DataSonde. Salinity readings on old and new DataSondes were within 0.6 ppt and dissolved oxygen readings were within 0.5 mg/L for six of the seven data pairs. Differences in dissolved oxygen concentrations of 0.5 mg/L may not be due to the effect of fouling or differences between the two DataSondes. Dissolved oxygen fluctuations of 0.5 mg/L were often recorded on a single DataSonde during successive 15-min readings.

Surface and bottom dissolved oxygen and temperature at all three sites were also monitored periodically with a Yellow Springs Instruments (YSI) model 57 dissolved oxygen meter. Concurrent conductivity measurements were taken with either a YSI model 33 salinity-conductivity-temperature meter or a Beckman model RS5-3 salinometer. Ten of 13 YSI bottom dissolved oxygen readings at the 5 m site were within 0.6 mg/L of the DataSonde measurement taken during the same 15 min interval (range of differences = 0.1 - 1.2 mg/L).

Benthic fish were collected by two methods: recruitment trays and suction samples. The biases of the two methods are probably different but are unknown. Both types of sampling were conducted by divers using scuba. Therefore we were able to supplement sample data with visual observations. Replicate samples were taken approximately 10 m apart along a line roughly centered at each permanent station and parallel to shore.

Recruitment trays were 0.35 m^2 (67 x 51.5 x 6 cm deep) plastic trays filled with approximately 12 L of oyster shell overlaying 4 L of sand. Trays were deployed for 10 - 29 d. Escape of fish during retrieval was minimized by first covering trays with 1-mm mesh netting and then placing each covered tray in a nylon drawstring bag. Naked gobies swim down among the oyster shell rather than move laterally when disturbed. Recruitment trays did not work well at the shallow site. Shallow trays were often overturned by water motion. Those shallow trays we were able to retrieve may underestimate goby populations because the shells in these trays were often obviously disturbed and, in some cases, part of the trays contents had been lost.

Suction samples were taken by collecting all fish contained within a 0.26 m^2 metal cylinder (20 cm high by 57 cm diameter) pushed several cm into the substrate (where possible). Water was pumped by a 5 hp gasoline engine through flexible reinforced hose. The end of the hose was attached to a PVC pipe with a cloth collection bag at its end. A narrower PVC pipe was attached at an angle to the wider pipe. The pumped water flowed across the opening of the narrow pipe thus drawing benthic organisms up and into the cloth collection bag.

RESULTS

Dissolved Oxygen

From 22 June to 21 September dissolved oxygen concentrations at the 5-m site fell to or below 4 mg/L on 51 of the 66 d for which we have measurements (Figure 1). On 17 days, oxygen levels fell below 2 mg/L. In contrast, daily maximum dissolved oxygen concentrations during the same period were always above 4 mg/L even on days with daily minimums as low as 0.2 mg/L dissolved oxygen. Diel fluctuations of up to 8.5 mg/L were recorded.

Two periods of extremely low minimum dissolved oxygen concentrations occurred. On 25-27 July and again on 2-3 August dissolved oxygen concentrations dropped to 0.2 mg/L (Figures 1 & 2a). The elevated salinities accompanying the low dissolved oxygen concentrations during these episodes suggest that the severe hypoxia resulted from cross-Bay tilts of sub-pycnocline waters advecting deep water onto the shallow flanks of the Bay. On other days daily minimum dissolved oxygen concentrations could occur during daylight or dark and fluctuations in dissolved oxygen could be accompanied by or be independent of fluctuations in salinity (Figures 2b & c).

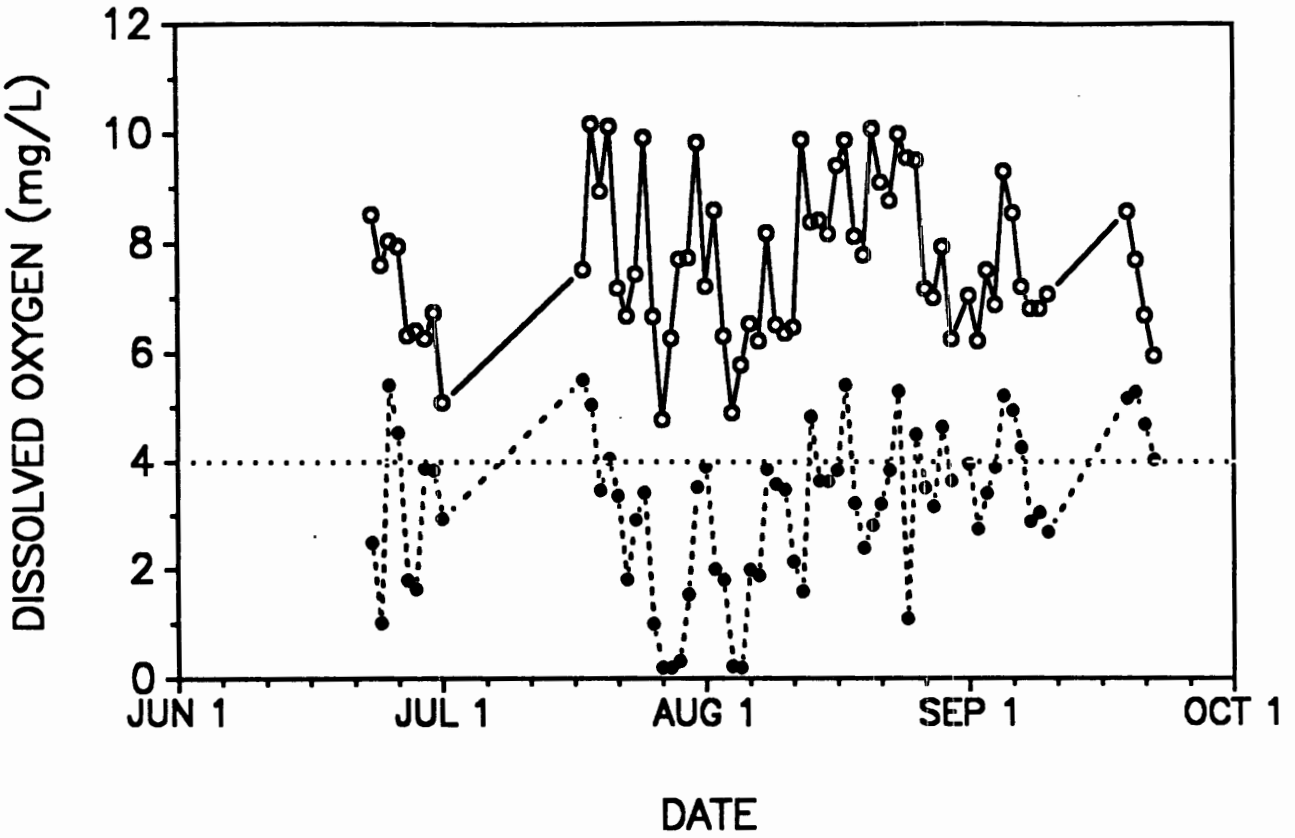
The relationship between dissolved oxygen and depth varied with stratification of the water column. On days when surface and bottom salinities a the 5-m site differed by $\geq$0.5 ppt, bottom dissolved oxygen averaged 1.9 mg/L higher at 2 m than at 5 m (n=6, range = 0.0 - 3.1). On days with smaller surface-to-bottom salinity differences, dissolved oxygen at 2 m averaged only 0.8 mg/L higher than at 5 m (n=9, range=0.0-1.6). Dissolved oxygen and salinity at the 5- and 6-m sites were generally the same.

Recruitment and Population Fluctuations

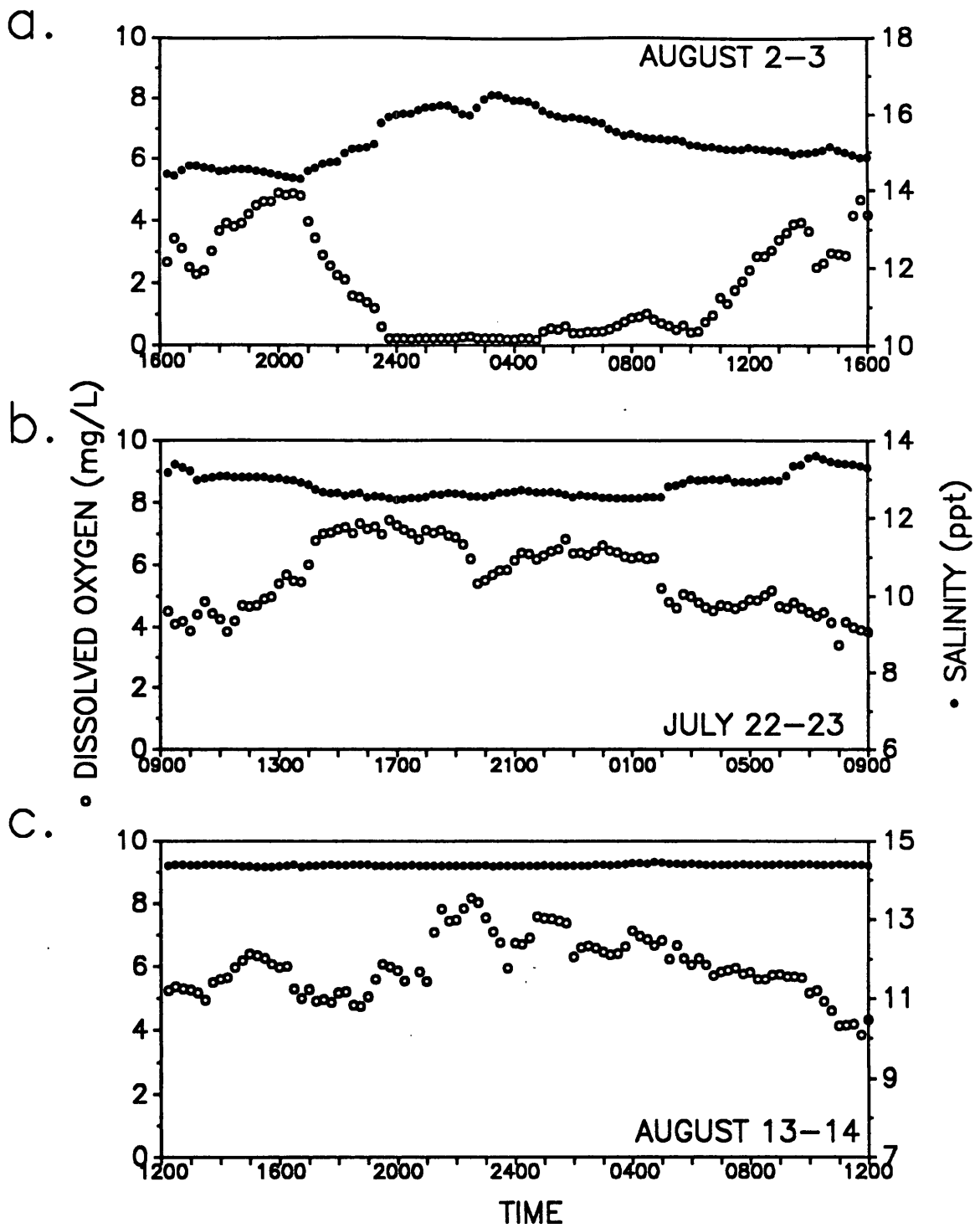
Recruitment of naked gobies began during early July and although we collected no naked gobies <18 mm TL after August 7, settlement stage demersal larvae were seen at the study site through late September (Breitburg in preparation). Schools of demersal larvae containing several hundred individuals were generally observed around high-relief structures, especially at a 0.6-m high research platform at the 5-m site. However, on 29 July no larvae were seen and on 4 August only a single demersal larva was found.

Prior to the late July and early August episodes of severe hypoxia, large numbers of recently settled juveniles were found in suction samples from all three depths, with higher densities at the two deeper sites (Figure 3). Two days after the second low dissolved oxygen episode, only 4 recruits were taken in suction samples at the two deep sites combined, as compared to 41 recruits at the shallow site (n=5 samples/depth). Only at the shallow site were recruits <18 mm TL collected. Several weeks after the second low dissolved oxygen episode, young-of-year gobies were again more abundant at the deeper sites than at the shallow site.

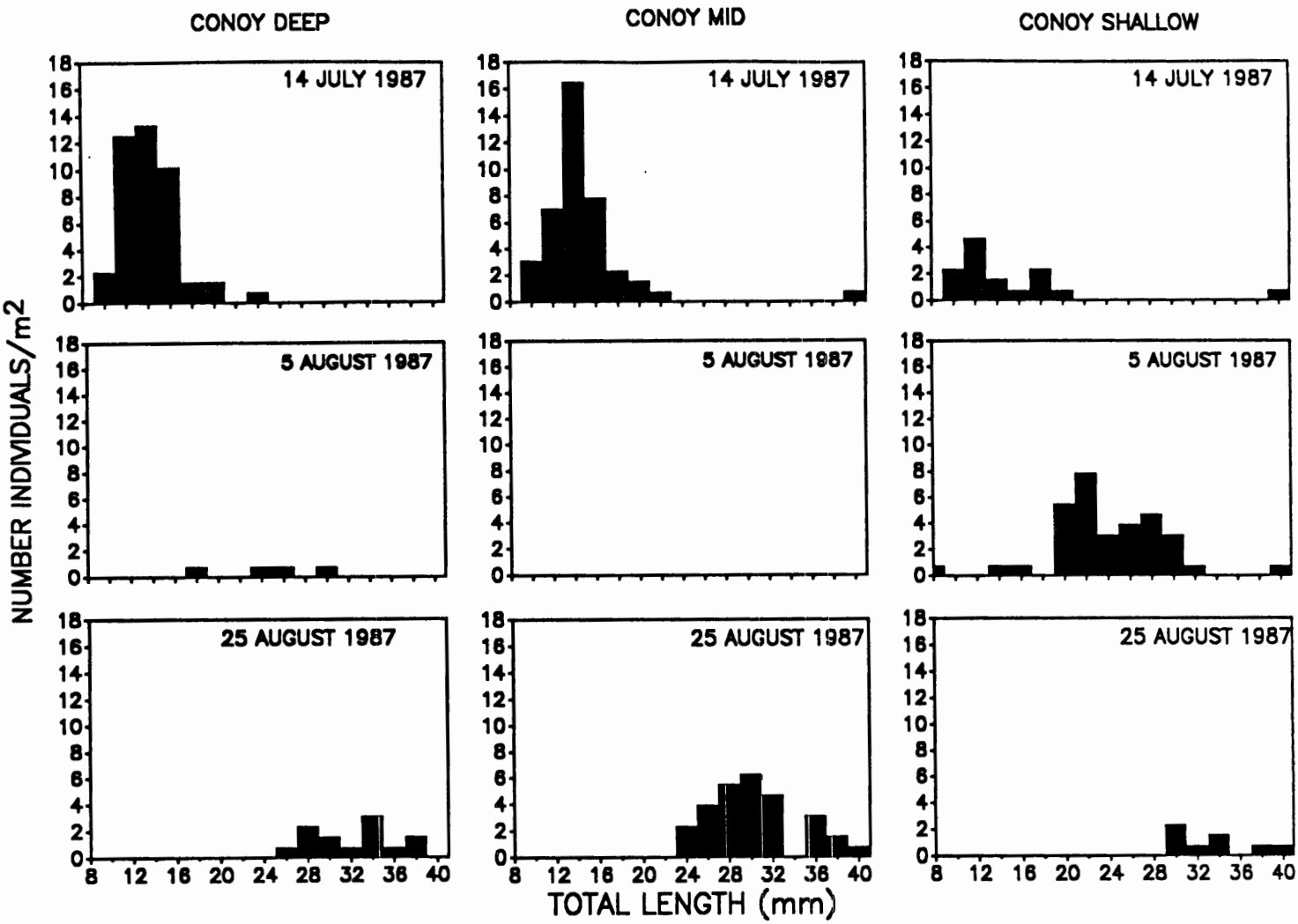
Our recruitment tray data are less complete for the 2- and 6-m sites and did not show as dramatic a population decline as did suction samples (Figure 4). Nevertheless, 4 d prior to the first low dissolved oxygen episode an average of 8.3 recruits <23 mm TL were



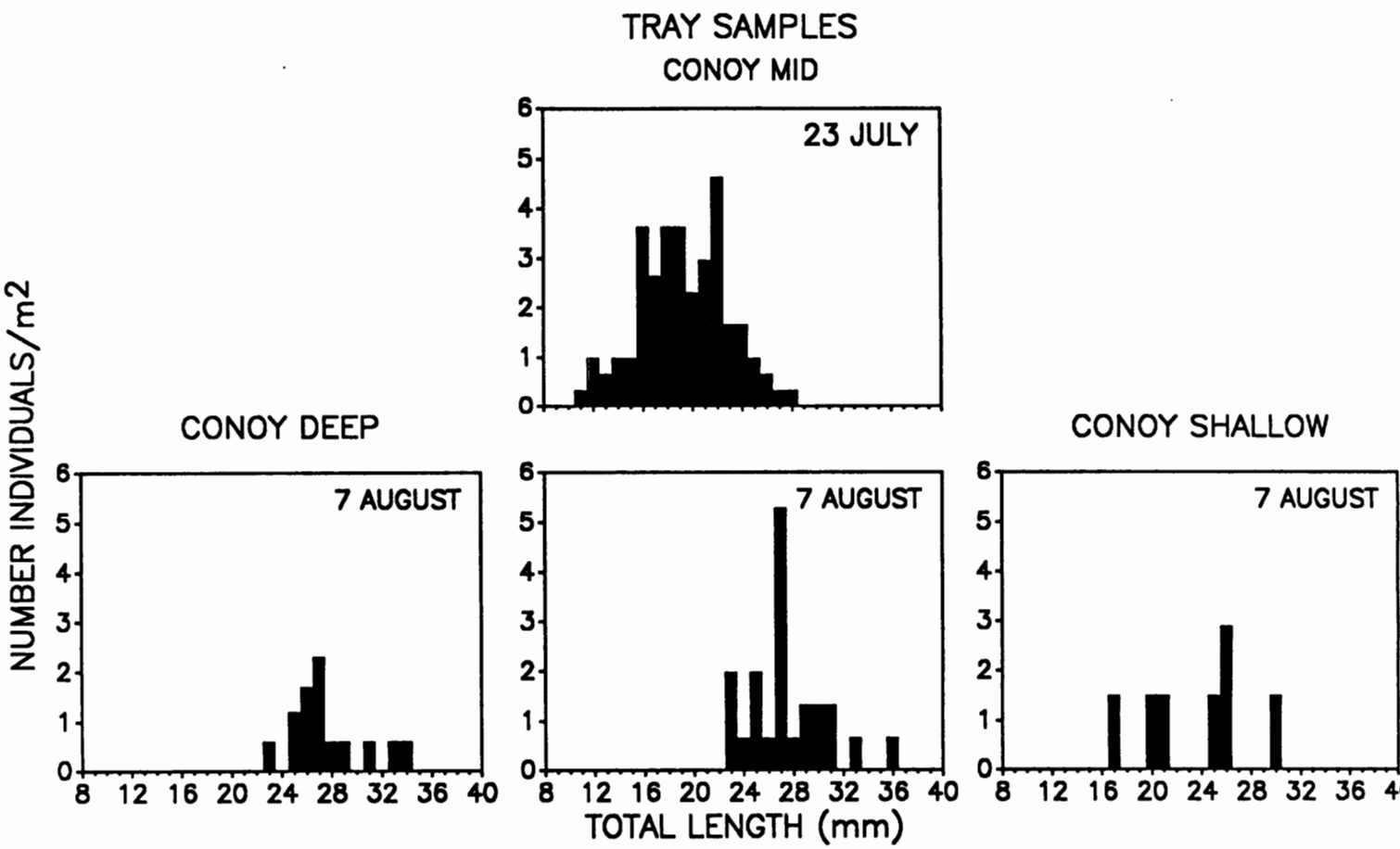
1. Daily maximum and minimum dissolved oxygen concentrations recorded with Hydrolab DataSondes at 5 m at Camp Conoy.



2. Examples of diel fluctuations in dissolved oxygen and salinity. Note that starting time differs among graphs. a. Severe hypoxia episode probably associated with a cross-Bay tilt of the pycnocline. b. Example of maximum dissolved oxygen concentration occurring during daylight and associated with fluctuations in salinity. c. Example of maximum dissolved oxygen concentrations occurring during the night and not associated with fluctuations in salinity.



3. Size distributions and densities of gobies collected in suction samples. N = 5 trays per date and site.



4. Size distributions and densities of gobies collected with recruitment trays. N = 5 trays per date and site except: Conoy shallow 9 July = 3 trays, Conoy shallow 7 August = 2 trays, Conoy mid 23 July = 10 trays.

collected in each recruitment tray at the 5-m site. Two weeks later, no juveniles <23 mm TL were found at either of the deep sites. As with suction samples, the smallest juveniles collected in recruitment trays following the low dissolved oxygen episodes were from the shallow site.

DISCUSSION

Our data indicate that shallow water habitats on the flanks of the Bay frequently experience oxygen levels that are physiologically stressful and sometimes experience oxygen levels lethal to fish. Although we recorded dissolved oxygen levels as low as 0.2 mg/L, we found no evidence that anoxic waters intruded into, or developed at the site monitored during our study. However, during years of heavier streamflow the volume of anoxic water in the Bay should be greater and anoxia may occur at the site.

Susceptibility to hypoxic conditions may depend on age or size of fish. Both tray and suction sample data indicated that oxygen levels that occurred during late July and early August of 1987 probably caused mortality at least of small juvenile naked gobies. At the 5-m site no juveniles <23 mm TL were found in recruitment trays following the episodes of severe hypoxia. Just two weeks prior juveniles as small as 11 mm TL were found at the same site. Our data do not allow us to determine whether low oxygen concentrations were a direct cause of mortality or whether young fish suffered increased predation when they attempted to escape the hypoxic waters. Because daily fluctuations in dissolved oxygen were often large and oxygen levels exceeded 4 mg/L during some part of each day, both timing and duration of low dissolved oxygen levels are likely to influence their effects.

Our suction sample data suggest that large juvenile and adult naked gobies migrate inshore in response to severe drops in dissolved oxygen. Densities of naked gobies in 5 and 6 m of water were markedly lower following the two episodes of 0.2 mg/L dissolved oxygen concentrations than prior to the episodes. In contrast, goby densities at the shallow site increased during the same period. Before the abundances of sumberged macrophtyes declined, vegetated areas may have served as important shoreline refuges when fish were forced out of oyster bars during intrusions of hypoxic or anoxic water. Our data also suggest a migration back to the deeper areas of the oyster bar during the weeks following the low dissolved oxygen episodes.

Goby densities in recruitment trays at the 5-m site did not decline as dramatically as did densities in suction samples from the natural substrate. Differences in microhabitat between trays and natural sediments, such as differences in the microbial biota or the amount of organic matter, may have lessened the impact of low dissolved oxygen concentrations on fish in trays. Trays were also sampled two days later than suction samples were taken and therefore some back-migration may have occurred.

Although this paper has focused on the naked goby, several other benthic fish species are abundant in Chesapeake Bay oyster bars.

Numbers of these fishes changed in a similar manner as described above for the naked goby following the periods of severe hypoxia (Breitburg unpublished data). As a group, these oyster bed species may serve as a model for determining the impacts of low and fluctuating oxygen concentrations on benthos-dependent Chesapeake Bay fishes.

ACKNOWLEDGMENTS

The others included in the "we" in this paper and to whom I am extremely grateful are Brian Albright, who was fundamental to all aspects of this work, Tim Thoman, Bill Yates, Kevin Junghans, Bridget Disharoon, Kevin Braun and Fritz Reidel. Kevin Sellner and Jim Sanders reviewed an earlier draft of this manuscript. I also thank Jacques van Montfrans for sharing the design of this suction sampler. The work is a result of research sponsored by NOAA Office of Sea Grant, U. S. Department of Commerce, under Grant No. NA85AA-D-SG042 to the Virginia Graduate Marine Science Consortium and Virginia Sea Grant College Program and Grant No. R/DO-1B from Maryland Sea Grant College. The U. S. Government is authorized to produce and distribute reprints for governmental purposes notwithstanding any copyright notation that may appear hereon.

LITERATURE CITED

Carter, H. H., R. J. Regier, E. W. Schiemer, and J. A. Michael. 1978. The summertime vertical distribution of dissolved oxygen at the Calvert Cliffs Generating Station; A physical interpretation. CBI Ref. No. 78-1, Spec. Rept. 60, The Johns Hopkins University, Shady Side, Maryland. 95 pp.

Coutant, C. C. 1985. Striped bass, temperature and dissolved oxygen: a speculative hypothesis for environmental risk. Trans. Am. Fish. Soc. 114:31-62.

Dovel, W. 1971. Fish eggs and larvae of the upper Chesapeake Bay. Nat. Res. Inst., Univ. of Maryland Contrib. No. 460. 71 pp.

Gallagher, R. P. and L. E. Currence. 1982. Ichthyoplankton and macroplankton studies in the vicinity of Calvert Cliffs Nuclear Power Plant, 1981. Prepared by The Academy of Natural Sciences, Philadelphia.

Holland, A. F., N. K. Mountford, M. H. Hiegel, K. R. Kaumeyer and J. A. Mihursky. 1980. Influence of predation on infaunal abundance in upper Chesapeake Bay. U.S.A. Mar. Biol. 57:221- 235.

Kemp, W. M. and W. R. Boynton. 1981. External and internal factors regulating metabolism rates of an estuarine benthic community. Oecologia (Berl.) 51:19-27.

Kramer, D. L. 1987. Dissolved oxygen and fish behavior. Env. Biol. Fish. 18:81-92.

Malone, T. C., W. M. Kemp, H. W. Ducklow, W. R. Boynton, J. H. Tuttle, and R. B. Jonas. 1986. Lateral variation in the production

and fate of phytoplankton in a partially stratified estuary. Mar. Ecol. Prog. Ser. 32:149-160.

Markle, D. F. and G. C. Grant. 1970. The summer food habits of young-of-year striped bass in three Virginia rivers. Ches. Sci. 11:50-54.

Magnuson, J. J., A. L. Beckel, K. Mills and S. B. Brandt. 1985. Surviving winter hypoxia: behavioral adaptations of fishes in a northern Wisconsin winterkill lake. Env. Biol. Fish. 14:241- 250.

Price, K. S, D. A. Flemer, J. L. Taft, G. B. Mackiernan, W. Nehlsen, R. B. Biggs, N. H. Burger, and D. A. Blaylock. 1985. Nutrient enrichment of Chesapeake Bay and its impact on the habitat of striped bass: a speculative hypothesis. Trans. Am. Fish. Soc. 14:97-106.

Rogers, B. A., D. T. Westin and S. B. Saila. 1982. Development of techniques and methodology for the laboratory culture of striped bass, Morone saxatilis (Waldbaum). U. S. Environmental Protection Agency, Narragansett,Rhode Island. 264 pp.

Saksena, V. P. and E. B. Joseph. 1972. Dissolved oxygen requirements of newly hatched larvae of the striped blenny (Chasmodes bosquianus), the naked goby (Gobiosoma bosci) and the skilletfish (Gobiesox strumosus). Ches. Sci. 13:23-28.

Siefert, R. E. and W. A. Spoor. 1974. Effects of reduced oxygen on embryos and larvae of the white sucker, coho salmon, brook trout and walleye. Pages 487-495 in J. H. S. Blaxter, ed. The Early Life History of Fish. Springer-Verlag, New York.

Wakefield, W. W., L. F. Berseth and S. L. Zeger. 1977. Ichthyoplankton entrainment studies. Pages 16-1 - 16-145 in Morgantown Station and the Potomac estuary: a 316 environmental demonstration. Vol. III. Prepared by the Academy of Natural Sciences, Philadelphia.

Wass, M. L. and T. D. Wright. 1969. Coastal wetlands of Virginia. Interim report to the Governor and General Assembly. Spec. Rep. Applied Mar. Sci. and Ocean. Eng. No. 10.

Understanding the Estuary: Advances in Chesapeake Bay Research. Proceedings of a Conference. 29-31 March 1988. Baltimore, Maryland. Chesapeake Research Consortium Publication 129. CBP/TRS 24/88.

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

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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

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	-

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-

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

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 1980 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 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		OBS.DO
		FREE CO2	TOTAL CO3	
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		OBS.DO
		FREE CO2	TOTAL CO3	
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		OBS.DO
		FREE CO2	TOTAL CO3	
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	CB4.3C	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

BIBLIOGRAPHY

Fair,G.M.;Geyer,J.G. Water supply and waste-water disposal. New York, NY:John Wiley and Sons; 1963: Chapter 28.

Harvey,H.W. The chemistry and fertility of sea waters.Cambridge, UK: Cambridge University Press; 1963: Chapter 10.

Kinne,O.,editor. Marine ecology. vol 1,part 3. New York, NY: Wiley-Interscience;1972: Chapter 9.

Understanding the Estuary: Advances in Chesapeake Bay Research. Proceedings of a Conference. 29-31 March 1988. Baltimore, Maryland. Chesapeake Research Consortium Publication 129. CBP/TRS 24/88.

Long Term Pattern of Anoxia in the Chesapeake Bay

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Introduction

There has been a dramatic decline in harvests and in the standing stocks of commercially important fish and shellfish species since the 1950's. Since hyper-eutrophication has been implicated as a controlling factor in the losses of fish and shellfish species in other aquatic ecosystems, and nutrients in effluents delivered into the bay have increased significantly since 1950 (Heinle et.al., 1980; Flemer et.al.,1983), it has been proposed (Mackiernan et.al., 1983) that increases in nutrients have likewise been responsible for the species declines that have occurred during this same period. While detailed mechanisms relating nutrient increases to decimation of specific food chains have not been described, it has been tacitly assumed that anoxia (which has been associated with mortalities of fish, shellfish and crustaceans), and overfishing, have been the major causes of the continued decline of fish, crabs and shellfish in the bay. The increased anoxia hypothesis as the link between increased nutrients and biota loss has been reinforced by reports that volumes of anoxic waters in the bay since 1950 have increased annually until in 1980 they were 15 times higher than in the 1950's (Mackiernan et.al., 1983; Officer et.al., 1984). This nutrient scenario for the bay assumes the following sequential processes, each of which is implicitly preceded by the adjective "increased" (Holland et.al., 1977):

nutrients --> primary and secondary production --> organic deposition --> benthic and water column respiration --> hypoxia in bottom waters --> anoxia in bottom waters in the presence of strong density stratification--> mortalities

When anoxia in the Chesapeake Bay first occurs it appears in bottom waters in a

deep hole just below the Bay Bridge, 38°58'N, Figure 1. This is the first place in the bay where stratification occurs in spring and high salinity waters, trapped in this hole, spend the longest times below the pycnocline without exchange with surface waters. In a paper describing intrusions of summer anoxic waters from the bay into the Chester River, Eastern Bay, and the Choptank and Potomac Rivers during 1984 , an empirical formula was used for calculating the cumulative decreases of dissolved oxygen concentrations in bottom waters at the Bay Bridge (Seliger and Boggs, 1985). The monthly losses in dissolved oxygen were calculated by summing for each month the product of a respiration rate and a weighting factor, the monthly Bulk Richardson Number. The Bulk Richardson Numbers (Bowden, 1978), were proportional to streamflow and inversely related to the vertical advective mixing (exchange of oxygen-depleted bottom waters with oxygen rich surface waters across the pycnocline). The severe summer anoxia in the upper Chesapeake Bay in 1984 (Seliger et.al., 1985) appeared to be consistent with the conclusions of Officer et.al. (1984) and with Environmental Protection Agency reports (Heinle et.al., 1980; Flemer et.al., 1983; Mackiernan et.al., 1983), that there had been a 15-fold increase in anoxia in the bay since 1950. The conclusions of Officer et.al., (1984), that "...benthic respiration rather than water column stratification has been the controlling factor in the historical increase [1950-1980] in anoxia", appeared to be necessary in order to explain the increased anoxia in the absence of any increased trend of annual streamflows, i.e., streamflow-induced stratification (inhibition of vertical advective mixing) was the same in the 1980' as in the 1950's

Several pieces of information caused us to question whether anoxia had increased since 1950: In 1985 low Susquehanna River spring streamflow resulted in negligible anoxia even at the Bay Bridge. Since Taft et.al., (1980) had emphasized water column respiration in oxygen depletion of bottom waters, nutrient deliveries in 1985 should have produced some anoxia, at least more than in 1965. In the summers of 1936 and 1937, in the central bay (38°18'N), there was extensive stratification of both salinity and oxygen with anoxia at the bottom (Newcombe and Horne, 1938). Likewise in the summers of 1951, 1952 and 1953, anoxia in the central bay (38°10' - 38°20'N) was severe enough to kill crabs in crab pots at depths of 7 meters (Carpenter and Cargo, 1957) . Thus there appeared to have been extensive anoxia in the bay much earlier than the 1980's.

We therefore calculated summer volumes of anoxic bay waters from all data set on bay-wide measurements of vertical profiles of salinities and concentrations of dissolved oxygen and at the same time compared them with spring Susquehanna River streamflows. In the present paper we have referenced and analyzed all of the data (1950-1985) of bay-wide concentrations of summer dissolved oxygen and salinities, and all of the corresponding spring streamflow volumes from the Susquehanna River. The data base is mainly from the Chesapeake Bay Institute Reports. It also includes our own surveys of salinities and dissolved oxygen concentrations made during 1983, 1984 and 1985, and those of Jonas et.al., (1985), for the summer of 1985. Our analysis showed that for the 36 year period from 1950 to 1985:

I. The annual total volumes of anoxic bottom waters in the Chesapeake Bay showed no statistically significant trend of increase with time.

II. The annual total volumes of summer anoxic waters were linearly related to spring Susquehanna River streamflow volumes with a Student's t- ratio significant at greater than the 99.99% Confidence Level and a Correlation Coefficient of 0.92.

These conclusions take on additional significance since they relate to whether we understand all of the factors that have decimated fish and shellfish populations in what has been the most productive estuary in the United States. A question is raised as to whether the presently proposed federal and multi-state bay program: to restore the biota of the bay to their previous high production by reducing nutrients in effluents to the bay, rests on a firm scientific foundation.

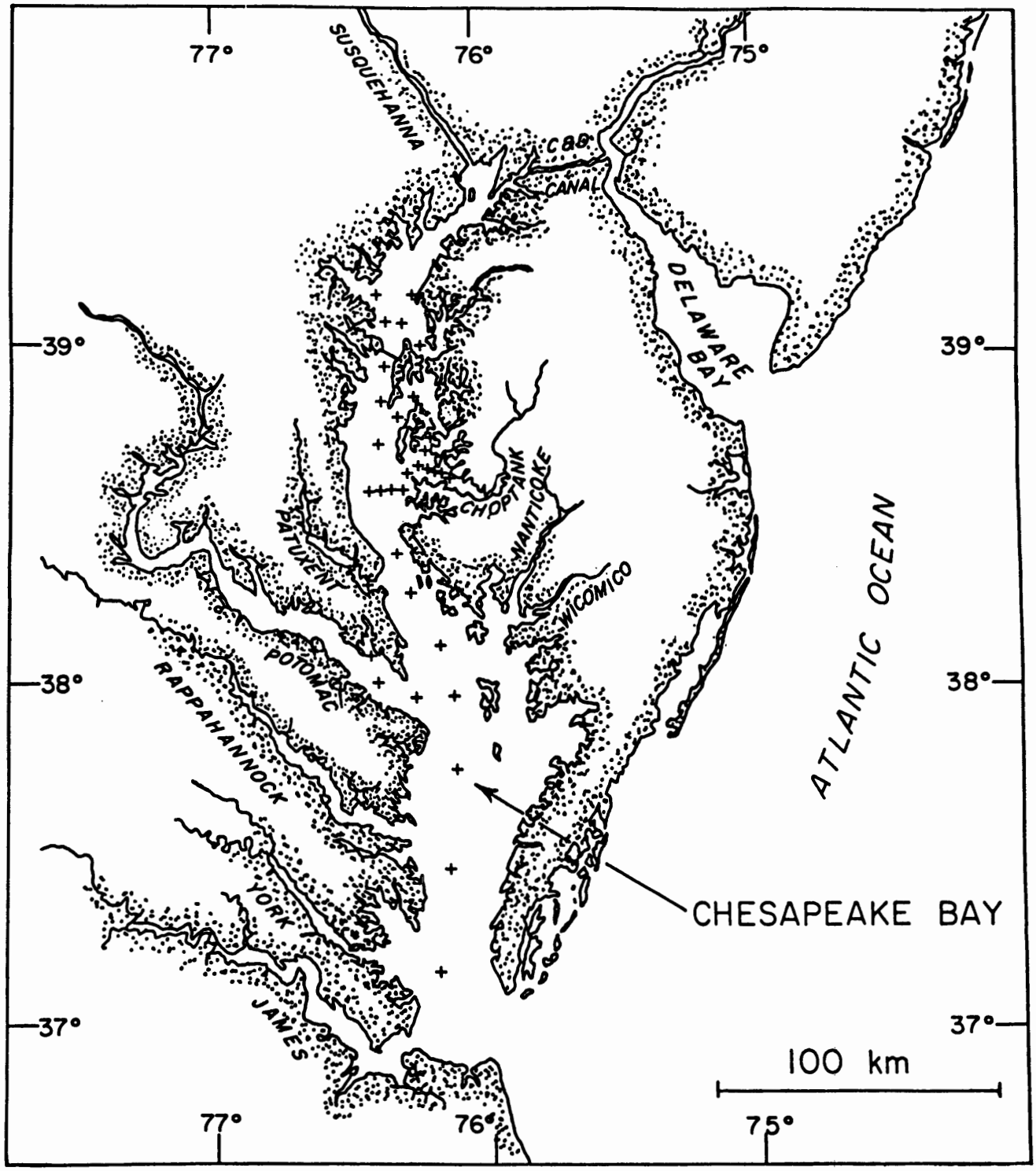


Figure 1. Chart of the Chesapeake Bay. The central region of the bay comprising the Choptank, Patuxent and Potomac tributaries is arbitrarily defined as from below the mouth of Eastern Bay (38°45'N) to below the mouth of the Potomac River (37°45'N). The Bay Bridge crosses the northern bay at 39°00'N, just north of the steep bathymetric discontinuity in the bay.

Calculations of Anoxic Volumes:

Isopleths of D.O. concentrations did not permit extrapolation to values lower than 1 ppm. The depth corresponding to the 1 ppm D.O. concentration in an annual data set was linearly interpolated from the axial station profiling data. The minimum interval for the depth profile at any station was 2-4 meters. Therefore the vertical resolution of interpolation of the 1 ppm value was ±1m, corresponding to an uncertainty in the calculated volume of anoxic waters of $\pm 1 \times 10^9$ m^3. Two- dimensional isopleths representing the 1 ppm D.O. concentration axially throughout the central bay were constructed on X-Z templates with units of nautical miles and meters respectively. The volumes for each nautical mile segment below the 1 ppm isopleth were determined from Chesapeake Bay bathymetry data (Cronin and Pritchard, 1975) and summed to calculate the total volume of anoxic waters (≤ 1 ppm) for each data set. The calculations covered an 80 nautical mile axial region of the northern and central bay between 39°00'N, the Bay Bridge and 37°40'N, the mouth of the Rappahannock River (see Seliger et.al., 1985).

ANALYSIS OF THE DATA

The complete data set beginning with 1950, the first year in which bay-wide measurements were made by the Chesapeake Bay Institute, is summarized in Table 1. The data comprise sixteen years during which summer vertical profiles of water salinities and D.O. concentrations were measured. These include our own surveys for 1983, 1984 and 1985 as well as data from Jonas et.al., (1985) for 1985. Each of the four decades since 1950 is represented by summer measurements made in at least three years. The data include the highest and lowest volumes of anoxic waters and the highest and lowest Susquehanna River spring streamflows measured for the Chesapeake Bay. As shown in Figure 2A and 2B analyses of the annual spring streamflow volumes entering the bay from the Susquehanna River and of the annual summer volumes of anoxic bottom waters, show no statistically significant slopes (increases or decreases in either streamflow or anoxia) over the period 1950-1985. The Regression Coefficients were 0.11 and 0.06 respectively.

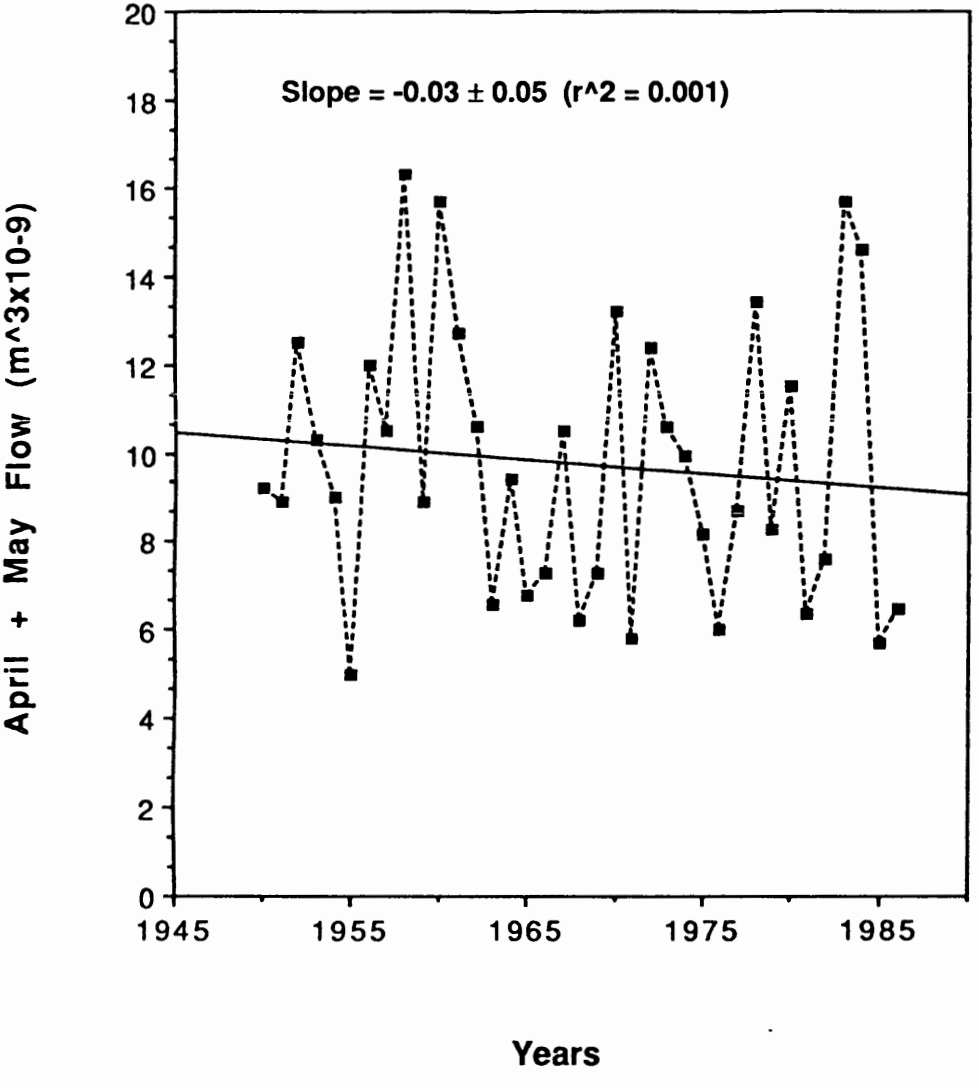
The conclusions of a 15-fold increase in anoxia since 1950 appear to be unfounded. A comparison may have been made between low anoxia years in the 1950's and high anoxia years in the 1980's. Two years in the 1950's, 1952 and 1958, for which extremely high summer anoxia in the Chesapeake Bay had been reported by the Chesapeake Bay Institute (Hires et.al., 1963) and an intermediate year, 1965, for which zero anoxia had been reported (Whaley et.al., 1966), were absent from the data considered in Officer et.al. (1984). The volume of anoxic waters measured for the summer of 1958 remains the highest ever recorded in the Chesapeake Bay. The low volume of anoxic waters measured in 1979 and the absence of anoxia in 1985 are further indications that the proposed trend of a 15-fold increase in anoxia is not supported by the data.

Since 60-90% of the nutrients delivered into the Chesapeake Bay are from non-point sources (Mackieman et.al., 1983), nutrient deliveries from the watershed will also correlate with streamflow. The dimensional analysis for the production of anoxia in bottom waters provides a means for resolving the possible effects on anoxia of increasing benthic and bottom water respiration. Assume that deliveries of annually increasing concentrations of nutrients during the 36 years from 1950-1985 had resulted in annually increased rates of respiration in benthic sediments and in the water column. Streamflow-induced stratification has remained essentially constant over this same period. Therefore in the presence of the same inhibition of vertical mixing increased respiration would have resulted in increased rates of oxygen loss and consequently increased extents and volumes of anoxic bottom waters during summer. Low streamflow years, producing weak stratification, which would have resulted in negligible anoxia in the 1950's, should now result in significant

Table 1. Calculated volumes of summer anoxic waters, volumes of late-spring Susquehanna River streamflows and sources of data on concentrations of D.O. during summer in the Chesapeake Bay for the period 1950-1985.

Year	Reference Number	(1) Calculated Anoxic Volume ($M^3 \times 10^{-9}$)	(2) April + May Susquehanna R. Streamflow ($M^3 \times 10^{-9}$)	Cruise Dates	Total Axial Bay Stations Occupied
1950	3, 9	1.4	9.2	14-19 Jul	17
1952	4, 9	4.7	12.5	15 Jul-6 Aug	17
1957	5, 9	1.8	10.5	23-26 Jul	12
1958	6, 9	8.5	16.3	6-22 Aug	17
1959	7, 9	2.9	8.9	6-17 Jul	17
1961	8, 9	4.4	12.7	19 Jul-1 Aug	17
1962	10	2.7	10.6	24 Jul-7 Aug	17
1965	11	<0.8	6.8	3-9 Aug	12
1969	12	2.7	7.3	5-8 Aug	12
1970	12	5.0	13.2	12-15 Aug	12
1978	13	5.6	13.4	18-20 Sept	14
1979	13	1.9	8.3	9-12 Jul	14
1980	13	3.8	11.5	23 Jul-2 Aug	14
1983	14	5.0	15.7	19-22 Jul	10
1984	14	6.6	14.6	22 Aug	10
1985	14, 15	<0.8	5.7	Mar-Aug	Bay Bridge Area

Table References: 1) Cronin, W.B. and D.W. Pritchard (1975). Additional Statistics on the Dimensions of the Chesapeake Bay and its Tubutaries. Ches. Bay Inst. Special RPT. #42, Ref. 75-3; 2) Estimated Streamflow Entering Chesapeake Bay - Monthly Summary Reports. U.S. Geological Survey, Towson, Maryland; 3) Ches. Bay Inst. Data RPT. #4, (1950); 4) Ches. Bay Inst. Data RPT. #18, Ref. 54-5, (1954); 5) Ches. Bay Inst. Data RPT. #33, Ref. 62-1, (1962); 6) Ches. Bay Inst. Data RPT. #38, Ref. 62-6, (1962); 7) Ches. Bay Inst. Data RPT. #41, Ref. 62--9, (1962); 8) Ches. Bay Inst. Data RPT. #47, Ref. 62-15, (1962); 9) Hires, R.I., et. al., (1963). Atlas of the Distribtion of Dissolved Oxygen and pH in Ches. Bay 1949-1961, Graph. Sum. RPT #3, Ref. 63-4; 10) Ches. Bay Inst. Data RPT #48, Ref. 62-16, (1962); 11) Whaley, R.C., et. al., (1966). Ches. Bay Inst. Spec. RPT #12, Ref 66-4; 12) Taylor, W.R. et.al., (1974). Ches. Bay Inst. RPT #38, Ref 74-6; 13) Cronin, W.B., et. al., (1982). Ches. Bay Inst. Open File RPT #28; 14) Unpublished Data - This Study; 15) Jonas, R.B., et. al., (1985). EOS Trans., Am. Geophys. Union 66:51, 1319.



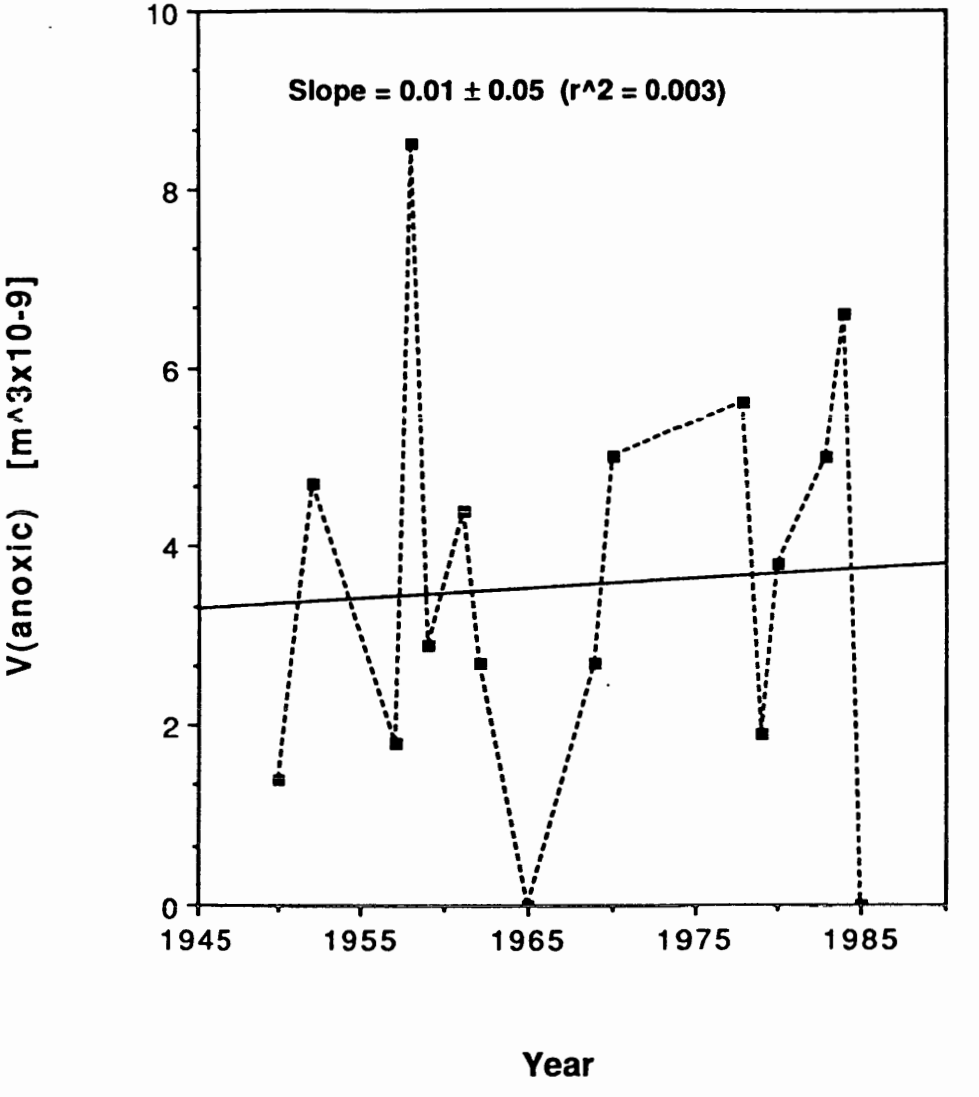


Figure 2. A. Annual April+May streamflow volumes from the Susquehanna River for the period 1950-1986, showing the large annual climatic variability characteristic of this geographical region and the absence of any long term trend in streamflow. B. Annual volumes of anoxic waters in the Chesapeake Bay during summer, for the period 1950-1985, showing large annual variability and the absence of any trend of increase in anoxia.

anoxia. In order to test this hypothesis we assumed a linear model for the relationship between annual volumes of summer anoxic waters and annual volumes of spring Susquehanna River streamflow for the entire 36 year period. If nutrient-dependent respiration had increased over this period, the co-variance between annual anoxic volumes and annual spring streamflow volumes should be low, resulting in a non-significant Student's t test and a low Correlation Coefficient for the linear regression (Edwards, 1979). On the other hand, if for some reason respiration had not changed significantly, a least squares linear regression of annual anoxic volumes upon annual spring streamflow volumes should be highly statistically significant by a Student's t test and consequently have a high Correlation Coefficient.

The 16 years of data in Table 1 includes 6 years where summer D.O. data were available only for July or September, when anoxic volumes might not have reached or might have receded from their maximum values. In order to eliminate any possibility of bias in the selection of data all of the data in Table 1 were included in the calculations. Fig. 3A is a superposition of the data of Figure 2A and 2B upon the same time axis. It implies a direct correlation between calculated anoxic volumes and spring streamflow volumes for the period 1950-1985. Fig. 3B shows the linear regression of these annual anoxic volumes against annual spring streamflow volumes and a least squares fit of the data. The regression coefficient ± Standard Error was 0.67 ± 0.077. The Student t ratio, 8.8 for 15 degrees of freedom, was extremely significant at greater than the 99.99% confidence level (Sokal and Rholf, 1969; Rohlf and Sokal, 1969). The Correlation Coefficient was 0.92, indicating that 85% of the total variance of annual anoxic volumes in the Chesapeake Bay can be accounted for by regression of anoxic volumes on streamflow volumes. None of the residuals was statistically significant. It might be expected that if the linear regression were restricted to the years when D.O. measurements were made in August, when anoxia might be presumed to be maximum, the Correlation Coefficient would be even higher. A separate linear regression of these 10 sets of data had a Correlation Coefficient of 0.96, with annual streamflow volumes accounting for 92% of the total variance in annual anoxic volumes.

A second feature of the linear regression analysis in Figure 3B was the April+May Susquehanna River streamflow threshold for anoxia of $(5.8\pm1.0)\times10^9$ m^3. The mean annual April+May Susquehanna River streamflow was $(9.3 \pm 3.3)\times10^9$ m^3 (SD; n=37). Thus summer anoxia appears to be a normal characteristic of the bay, occurring on the average in 7 out of 10 years. Its occurrence in bay bottom waters of the central bay was reported as early as 1936-1937 (Newcombe and Horne, 1938). The years with the two highest volumes (1958, 1984) and two lowest volumes (1965, 1985) of anoxic waters in the Chesapeake Bay, had spring streamflow volumes within the ±95% Confidence Limits of the annual streamflows.

Persistence of Stratification in the Upper Bay

The persistence and the downstream extents of stratification during summer are important features of the cumulative decreases in concentrations of D.O. in bottom waters leading to anoxia as they progress upstream below the pycnocline.

The Chesapeake Bay is unusual among all of the east coast estuaries in that Susquehanna and Potomac River streamflows during spring exceed the total volume of the upper bay, as shown in Figure 4. The net result is that large spring streamflow pulses produce sharply defined buoyant surface plumes in the upper bay, with large bulk stratifications, $\Delta S \approx 13$ °/oo. Haloclines at depths of 10-12 m, separating the southward-flowing surface plumes from the net northward-flowing bottom waters, are confined to a few meters in thickness. Salinity gradients can be as as high as $\Delta S/\Delta z \approx 3\text{-}6$ °/oo m^{-1}, resulting in essentially complete inhibition of mixing of surface waters with bottom waters.

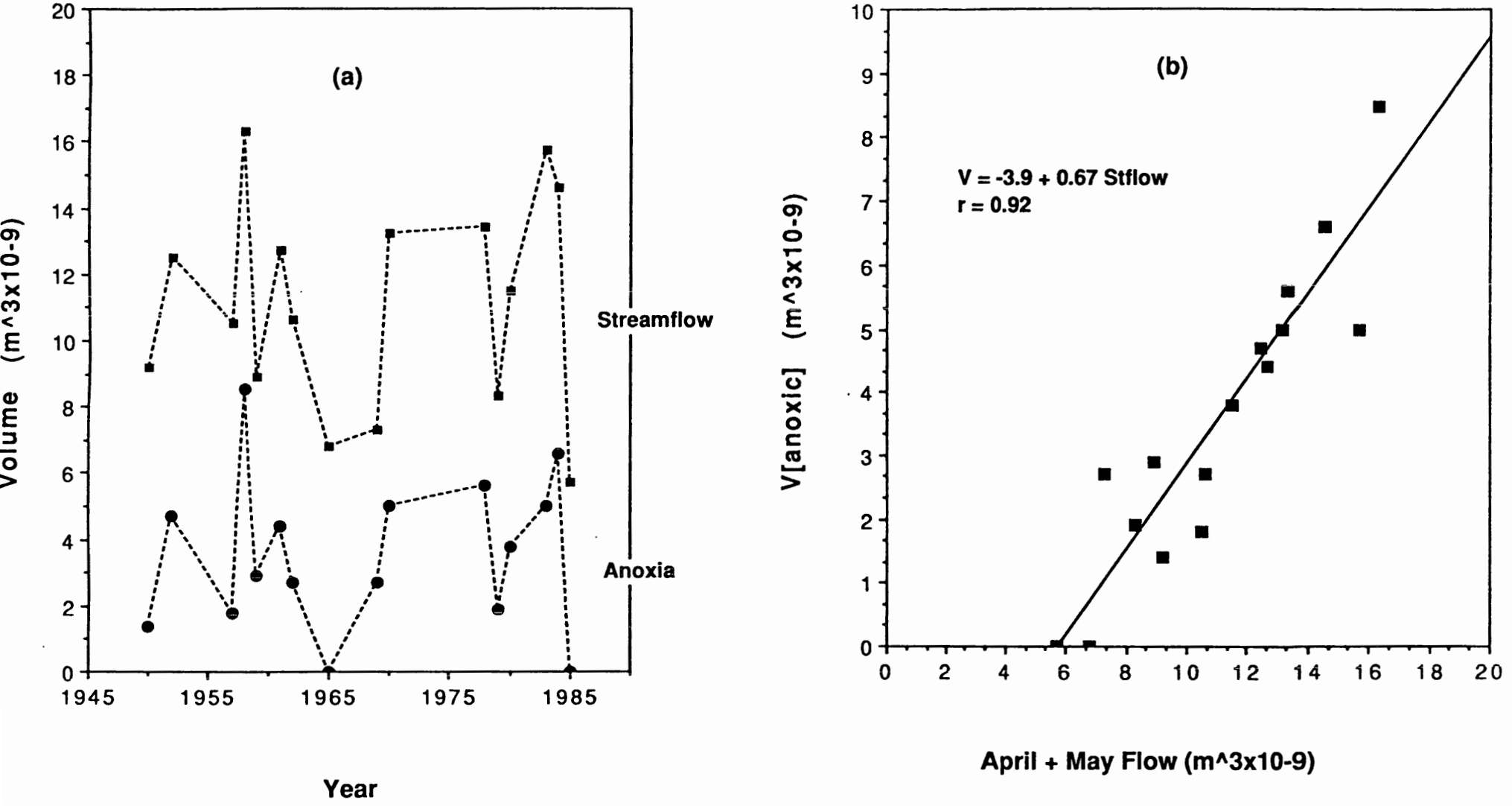


Figure 3. A. Annual volumes of April+May Susquehanna River streamflows superimposed upon annual summer volumes of anoxic bay waters over the period 1950-1985 for all years during which bay-wide surveys of concentrations of D.O. were made. B. Regression of the annual summer volumes of anoxic bay waters upon annual volumes of April+May Susquehanna River streamflows.

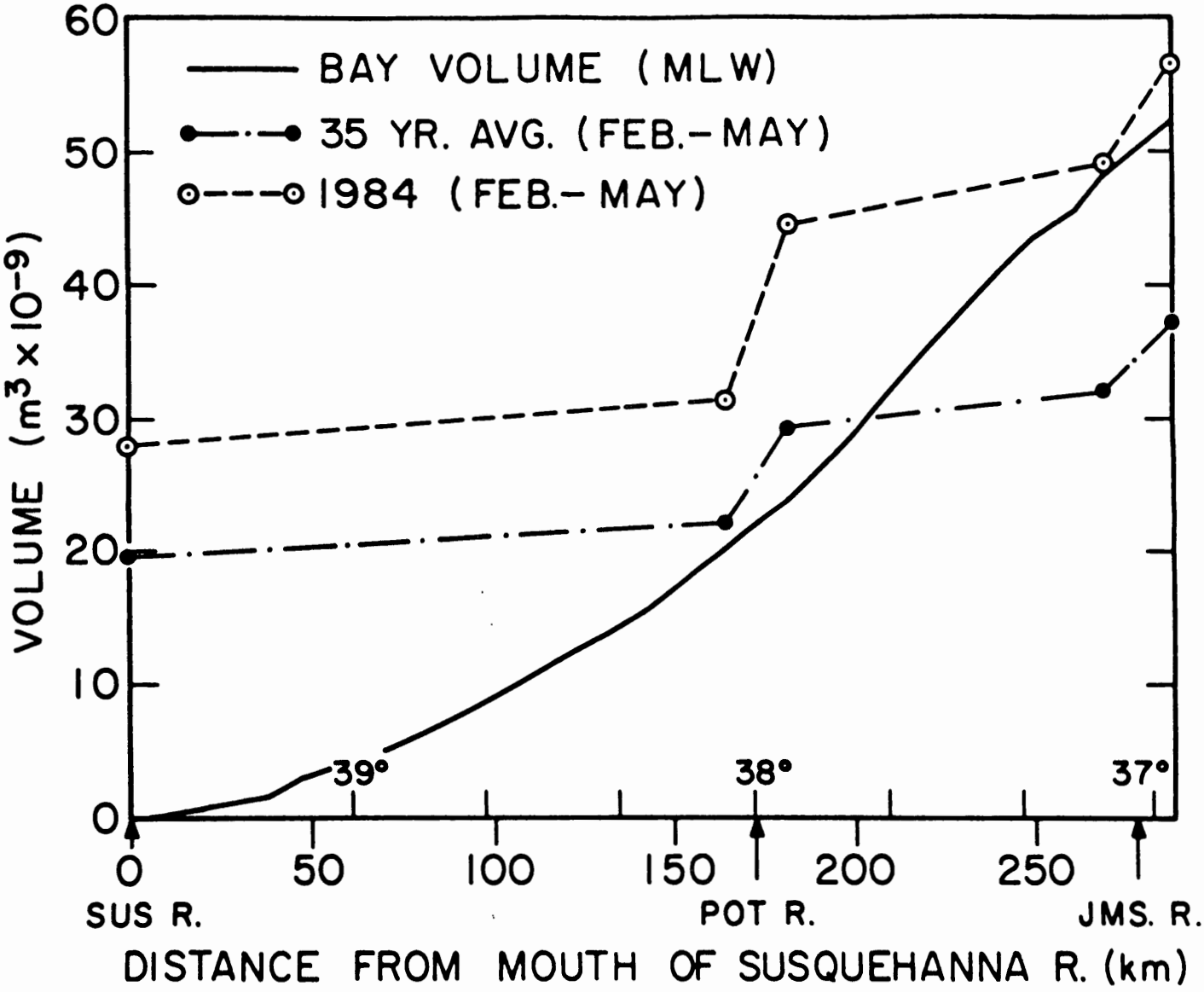


Figure 4. (Solid Heavy Line) Cumulative mean low water volume of the Chesapeake Bay from the mouth of the Susquehanna River to the mouth of the bay. (Dash-dot Line) Mean of 1950-1985 February-May streamflow volumes into the bay. (Dashed Line) 1984 February-May streamflow volume. The data indicate that spring streamflow volumes normally exceed the total volume of the northern and central regions of the bay.

The striking aspect of stratification in the upper Chesapeake Bay is that stratification set up in late spring is qualitatively maintained until the fall overturn by insolation and continued (although lower) streamflow. The precise axial locations of the plume fronts and the vertical density gradients will vary with daily and fortnightly phases of the tidal cycles, but there is not sufficient tidal or wind frictional turbulent power in the upper bay to dissipate the pycnocline completely. The variations are small compared with the total areal extents and gradients of the stratification. Even in the hot summer months streamflow accounts for 90% of the density stratification. In the presence of significant density stratification during the summer, anoxia is reached within a relatively short time in waters submerging below the pycnocline and moving upstream by reverse flow. Since bottom waters further upstream are already anoxic it appears as if there is a seasonal downstream progression of anoxia. Total anoxic volumes measured in August are higher than if measured in June or July, implying that water column respiration rates are highest in the hottest period of summer. If measurements are made in September, subsequent to the beginning of the fall overturn, anoxic volumes may be lower.

The persistence of late spring stratification into summer in the upper bay has been verified from data taken for other reasons during our own research cruises over the 15 year period from 1970 through 1984 (Loftus et.al. 1972; Subba Rao et.al. 1972; Seliger et.al. 1975; 1979; 1981; 1982; 1985; Tyler and Seliger 1978; 1981). Axial transects in different regions of the bay included vertical profiles of water column salinities (densities). During this period approximately 100 cruises of 3-4 day durations were made during spring-summer, as often as biweekly. Vertical profiles of salinities along axial transects were made in different regions of the bay at different phases of the tidal cycles during the 3-4 days. In a number of cases hourly vertical profiling was carried out at anchored stations over complete tidal cycles. In 1983 and 1984, in addition to biweekly 3-4 day cruises, hourly vertical profiles at an anchored station were made continuously over 2-3 week periods. In the years 1971, 1976, 1981 and 1985, low Susquehanna River spring streamflow produced minimal stratification in the upper bay. In 1972, Tropical Storm Agnes disrupted the stratification pattern in June. In all of the other years vertical profiles of salinity distributions in the upper bay showed the persistence of stratification from spring into the summer months. Therefore, despite diel and fortnightly relaxations and intensifications of stratification during flood and ebb, and spring and neap, tides respectively, two-dimensional, X-Z plots of isopleths of salinities (densities) measured at intervals of days to several weeks can be assumed to form a continuous series of snapshots of persistent stratification, at least in the upper bay (Tyler and Seliger 1978; 1981; Seliger et.al. 1981).

Dissipation of Stratification

As the buoyant surface waters delivered by a streamflow pulse flow southward with time, the depth of the halocline and the magnitude of the stratification are continuously decreased by the cumulative effects of tidal turbulent mixing, until at some latitude the halocline intersects the surface, defining the plume front. Following unusually high streamflow pulses the plume can temporarily extend throughout the bay and out into coastal waters. Normally in the lower bay stratification has decreased sufficiently to allow partial vertical mixing of the water column. This effect is strongest during the fortnightly spring tides (Haas 1981). Therefore bottom waters in the southern bay may exhibit temporary periods of hypoxia, but are not subject to summer anoxia. Stratification is dissipated completely during the fall overturn, when colder air temperatures produce density instabilities in surface waters. This partially to well-mixed condition remains throughout the winter until ice melting and rainfall in the watershed of the Susquehanna River again result in high spring streamflow pulses.

Possible Sources of Biota Loss Supplemental to Anoxia

Increased nutrients may affect bay biota in more esoteric ways than anoxia, by stimulating growth of previously poorly competitive food chains at the expense of desirable species.

Increased rates of sedimentation in the tributaries of the bay and in the bay proper have resulted in significant losses of productive shoals in the tributaries and along the shores of the bay. Because of their shallow depths, these previously productive habitats were minimally affected by the deeper anoxic waters below the pycnocline, even during years of severe anoxia. These areas performed essential functions, as reserves for continued productivity and as seed areas for future years, when other, deeper, habitats below the bay pycnocline suffered mortalities during periods of severe anoxia (Seliger and Boggs 1986). Since at present only habitats at intermediate depths remain viable, severe summer anoxic episodes, which have always resulted in mortalities in intermediate depth habitats, now affect a much greater fraction of the total population and are relatively more dramatic.

During the spring streamflow pulses, sediment carried into the bay from the watershed of the Susquehanna R. most likely resulted in some light limitation of photosynthesis in the upper bay. However at the present time the large sediment loads carried by spring streamflow pulses so attenuate downwelling sunlight in the tributaries and in the northern bay proper that the system becomes strongly light-limited with linear light attenuation coefficients (400-700 nm) as high as 4 m^{-1} . This results in a downstream displacement of primary production and nutrient utilization, to latitudes in the Virginia portion of the bay where sufficient light for photosynthesis is transmitted to the upper meters of the water column. A similarly scaled phenomenon occurs in the tributaries. However the upstream tributary locations of spawning of anadromous fish and therefore of their larvae, are governed mainly by salinity and temperature. Thus because of water turbidity due to sediment deliveries, the spatial distributions of the larvae in the tributaries may now be mismatched with their phytoplankton-herbivore food sources .

Increased sediment loading of tributary streamflows may result in mortalities of submerged aquatic vegetation in two ways: a) there can be significant absorption of light by suspended sediment to below the compensation point for photosynthesis, and b) sediment deposition on the aquatic vegetation now results in the excessive growth of epiphytic food chains which absorb light. Loss of aquatic vegetation may adversely affect food chains dependent upon this trophic level.

Discussion

1958 and 1984 represented the maximum measured volumes of spring streamflow and anoxic waters. 1965 and 1985 spring streamflows were the lowest on record and measured anoxic volumes were essentially zero. There were marked similarities between the salinity isopleth distributions and the D.O. isopleths respectively, for 1958 and 1984, consistent with streamflow-driven stratification as the controlling factor for anoxia of bay bottom waters and therefore with the conclusion that there has been no trend in annual anoxic volumes.

Anecdotal descriptions of extensive local summer mortalities of blue crabs (Callinectes sapidus), oysters (Crassostrea virginica) and major fish kills (menhaden, white perch, silversides,...) due to low oxygen in the Chesapeake Bay and in its tributaries have characterized all decades since 1950 (Md. DNR Ann. Sum. Reported Fish Kills). There are no indications of recent increases of mortalities, which might be expected had there been a 15-fold increase in anoxia.

The 16 years of survey data on D.O. concentrations analyzed in this paper include our own surveys in 1983 and 1984, and our own and those of others in 1985. These represent the most complete set of measurements available for the Chesapeake Bay. In two

cases the absences of co-variance between variables were obvious from the graphical presentations. These were the plot of annual volumes of spring streamflows versus calendar year, Fig.1, in which there is no trend of increasing streamflows, and the plot of annual volumes of anoxic waters versus calendar year, Fig. 2, in which there is no trend of increasing anoxia.The least squares linear regression used to analyze the dependence of anoxia upon streamflow is a powerful statistical technique for determining how well experimental data fit a proposed model. The extremely high statistical significance of the regression analysis of annual volumes of anoxic waters on annual volumes of late-spring Susquehanna River streamflows, Fig. 3B, was consistent with a model in which annual benthic respiration had not increased and in which streamflow-induced stratification was the controlling factor in anoxia.

In summary, we find no evidence for a 15-fold increase in anoxia in the Chesapeake Bay since 1950, which has been used as the basis for a federal and multi-state program for restoring the biota of the bay by reducing nutrient inputs. It is likely that increased sedimentation has played a significant although indirect role in the modification of species' success in the bay. An intensive research effort, aimed at understanding the factors that have caused the dramatic declines in major commercial species of fish and shellfish since 1950, would appear to be necessary before taking actions to restore these species to their previous levels.

Boicourt, W.C., A. Numerical Model of the Salinity Distribution in Upper Chesapeake Bay, Chesapeake Bay Institute, Johns Hopkins University, Technical Report 54, Ref. 69-7, (1969).

Bowden, K.F. in Estuarine Transport Processes, B. Kjerfve, Ed.University of South Carolina Press, Columbus (1978). pp. 11-36.

Brandt,A., C.C. Saraban, H.H. Seliger, M.A. Tyler, in Marine Interfaces Ecohydrodynamics, Ed. J.C.J. Nihoul, Elsevier Science Pub. (1986),pp. 361-384.

Carpenter, J.H. and D.G. Cargo, Oxygen Requirement and Mortality of the Blue Crab in the Chesapeake Bay, Chesapeake Bay Institute, Johns Hopkins University, Technical Report 13, Ref. 57-2 (1957).

Cronin, W.B. and D.W. Pritchard, Additional Statistics on the Dimensions of the Chesapeake Bay and its Tributaries: Cross-Section Widths and Segment Volumes Per Meter Depth. Chesapeake Bay Institute, Johns Hopkins University, Special Report 42, Ref. 75-3 (1975).

Cronin, W.B., W.R. Taylor, M. Mallonee, Suspended Sediment Distribution and Related Data Sept. 1978-Oct. 1980, Chesapeake Bay Institute, Johns Hopkins University, Open File Report 28 (1982).

Edwards, A.L. "Multiple Regression and the Analysis of Variance and Co-variance." W.H. Freeman and Co. San Francisco (1979) 212 pages.

Flemer, D.A., G.B. Mackiernan, W. Nehlsen, V.K. Tippie, R.B. Biggs, D. Blaylock, N.H. Berger, L.C. Davidson, D. Haberman, K.S. Price and J.L. Taft, Chesapeake Bay: A Profile of Environmental Change (Environmental Protection Agency, Washington, D.C. 1983) 120 pp. and Append.

Haas, L.W., Estuarine Coastal Marine Science 5, 485 (1981).

Heinle, D.R., C. D'Elia, J. Taft, J.S. Wilson, M. Cole-Jones, A.B. Caplins and L.E. Cronin, Historical Review of Water Quality and Climatic Data from Chesapeake Bay with Emphasis on Effects of Enrichment (Chesapeake Bay Program Report, Environmental Protection Agency, Washington, D.C., 1980).

Hires, R.I., E.D. Stroup, R.C. Seitz, Chesapeake Bay Institute, Johns Hopkins University, Graphical Summary Report 3, Ref. 63-4, (1963).

Holland, A.F., N.K. Mountford, J.A. Mihursky, Chesapeake Science 18, 370 (1977).

Jonas, R.B., J.H. Tuttle, D.C. Cargo, T.C. Malone, H. W. Ducklow, EOS Transactions, American Geophysical Union 66(51), 1319 (1985).

Kemp, W.M. and W.R. Boynton, Oecologia (Berlin) 51, 19 (1981).

Loftus, M.E., D.V. Subba Rao, H.H. Seliger, Chesapeake Science 13, 282 (1972).

Mackierman, G.B., D.A. Flemer, W. Nehlsen, V.K. Tippie, State of the Bay in Chesapeake Bay: A Framework for Action (Environmental Protection Agency, Chesapeake Bay Program, Annapolis, MD, 1983).

Md. Dept. of Natural Resources, Tidewater Fisheries Administration Annual Summaries of Reported Fish Kills in Maryland Tidal Waters (1965-Present).

Newcombe, C.L. and W.A. Horne, Science 88, 80 (1938).

Officer,C.B., R.B. Biggs, J.L. Taft, L.E. Cronin, M.A. Tyler and W.R. Boynton, Science 223, 22 (1984).

Rohlf, F.J. and R.R. Sokal, Statistical Tables, W.H. Freeman and Co. Publishers, San Francisco (1969).

Seliger, H.H. and J.A. Boggs, Coastal Ocean Pollution Assessment News 3, (4), 47 (1986).

Seliger, H.H. Bulletin of Marine Science 37, (2) 775 (1985).

Seliger, H.H., J.A. Boggs, R.B. Rivkin, W.H. Biggley and K.R.H. Aspden. Marine Biology 71, 57 (1982).

Seliger, H.H., J.A. Boggs, W.H. Biggley, Science 228, 70 (1985).

Seliger, H.H., K. R. McKinley, W.H. Biggley, R.B. Rivkin and K.R.H. Aspden. Marine Biology 61, 119 (1981).

Seliger, H.H., M.E. Loftus, S.V. Subba Rao, in Proc. lst International Conference Toxic Dinoflagellate Blooms, Ed. V.R. LoCicero, Mass. Sci. Technol. Found. Wakefield, Mass. (1975). pp. 181-205.

Seliger,H.H., M.A. Tyler, K.R. McKinley, 2nd International Congress on Toxic Dinoflagellate Blooms, Eds. D.L. Taylor and H.H.Seliger, Elsevier, N.Y. (1979) pp 239-248.

Sokal, R.R. and F.J. Rohlf, Biometry, W.H. Freeman and Co. Publishers, San Francisco (1969). 776 pp.

Subba Rao, D.V., M.E. Loftus, H.H. Seliger, Ann. Report Chesapeake Research Consortium, May (1972) pp. 515-532.

Taft, J.L., W.R. Taylor, E.O. Hartwig, R.E. Loftus, Estuaries 3, 242 (1980).

Tyler, M.A. and H.H. Seliger, Limnol. Oceanogr. 23, 337 (1978).

Tyler, M.A. and H.H. Seliger, Limnol. Oceanogr. 26, 310 (1981).

Whaley, R.C., J.H. Carpenter, R.L. Baker, Chesapeake Bay Institute, Johns Hopkins University, Special Report 12, Ref. 66-4 (1966).

Understanding the Estuary: Advances in Chesapeake Bay Research. Proceedings of a Conference. 29-31 March 1988. Baltimore, Maryland. Chesapeake Research Consortium Publication 129. CBP/TRS 24/88.

Insights to the Chesapeake Bay's Eutrophication Process through Modeling

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INTRODUCTION

The Chesapeake Bay, the largest and one of the most productive estuaries in the country, has long been the subject of much research and study. Concern over the Bay's well being led Congress, in 1975 to direct the United States Environmental Protection Agency (EPA) to initiate a comprehensive study of Bay water quality and living resources. As a result, efforts to understand and protect the Bay have intensified, with an ever increasing level of public awareness and understanding of Bay water quality problems.

The initial phase of the EPA Chesapeake Bay Program study, completed in 1983, concluded that the Bay's water quality and living resources have indeed deteriorated as a result of pollution. Nutrient enrichment was cited as one of several factors contributing to the Bay's decline.

Lacking, however, was a tool to provide the cause-effect link between nutrient loading and Bay water quality. In order that these complex interrelationships could be better understood and quantified, and to provide a tool to assist in evaluating the effectiveness of alternative nutrient control strategies, the Chesapeake Bay Program embarked on a comprehensive water quality modeling strategy for the Bay in 1985. The initial phase of the strategy, the development of a large segment, three dimensional steady state eutrophication model, was completed in 1987. The final phase of work, the development of a three dimensional, time variable eutrophication model with an active sediment layer, was initiated in October, 1988 and scheduled for completion by March, 1991.

The insights to the Bay's eutrophication process gained through

the use of the steady state model will be discussed in this paper. Those insights have been used to identify areas needing additional research and monitoring, and to focus the subsequent time variable eutrophication modeling effort on the most important processes affecting Bay algal and dissolved oxygen concentrations.

STEADY STATE MODEL DESCRIPTION

The steady state model of the Chesapeake Bay is a large segment, three dimensional, coupled hydrodynamic/water quality model designed to assess the effect of nutrient inputs on phytoplankton growth and dissolved oxygen levels. A complete description of the model, including its calibration and sensitivity testing is given by HydroQual, Inc. (1987).

The entire Chesapeake Bay and its major tributaries are modeled using the horizontal segmentation shown in Figure 1. The main Bay is segmented into five vertical layers with a coarse scale (10 km.) in the horizontal plane. The major tributaries are segmented into two vertical layers with similar coarse horizontal resolution. The hydrodynamic model provides the circulation and transport which drive the water quality model. Eleven state variables (salinity, phytoplankton, the various forms of nitrogen and phosphorus, and dissolved oxygen) are incorporated in the water quality analysis. Figure 2 presents the principal kinetic interactions for the nutrient and dissolved oxygen cycles. The major inputs to the model are freshwater flows and water quality at the fall line, point and nonpoint source loads below the fall line, atmospheric nutrient loads, and Bay bottom sediment nutrient loads (both sink and source) and oxygen demand.

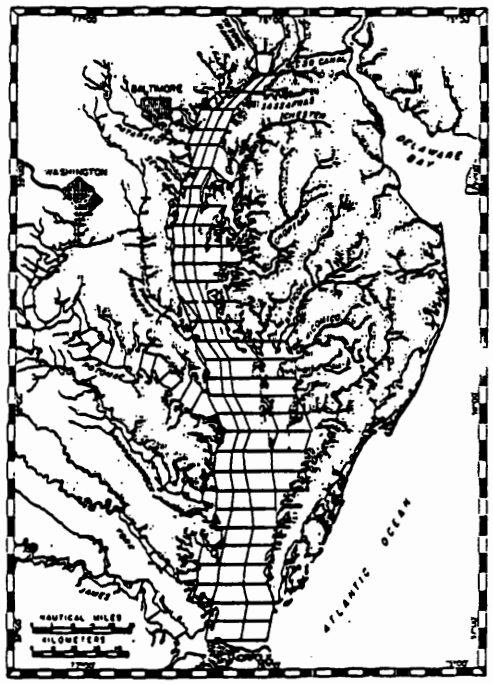

FIGURE 1. Model Segmentation

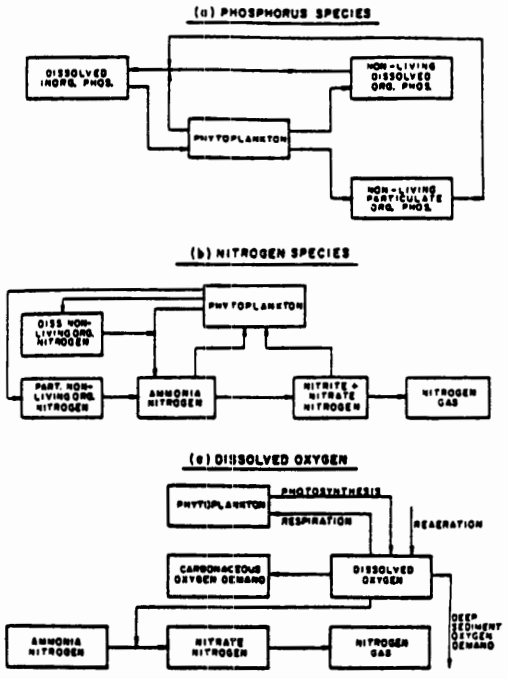

FIGURE 2. Kinetic Structure of Principal Water Quality Constituents

The water quality model has been calibrated to summer average conditions for 1965, 1984, and 1985, which are characterized by different freshwater flows, degree of vertical stratification, and nutrient inputs from point and nonpoint sources. A comparison of the observed and model computed values of the key hydrodynamic and water quality variables shows that both the hydrodynamic and water quality models realistically portray the important features of the Bay. Statistical comparisions of between observed and model predicted values show that for dissolved oxygen, the median relative error is less than 10%, with a median absolute error less than 0.5 mg/l. For total phosphorus, the median relative error was less than 20%, with a median absolute error less than 0.01 mg/l. The median relative error for total nitrogen was less than 15% with a median absolute error less than 0.10 mg/l. Chlorophyll a had the highest median relative error with values of 50% in 1965 and 1985, and 25% in 1984. The median absolute errors for chlorophyll a were 5 ug/l, 2.5 ug/l and 3.8 ug/l for 1965, 1984 and 1985 respectively. The magnitude of the error in the chlorophyll prediction reflects the patchiness found in chloropyyll data reflecting localized growth which cannot be simulated with a large scale model such as is being used for the Bay. The model does, however, reproduce the overall trend of the chlorophyll data.

CONCLUSIONS FROM MODEL CALIBRATION AND SENSITIVITY ANALYSIS

The process of model calibration, in itself, provides much valuable information regarding Bay water quality interactions. During and following model calibration, the response of the Bay to changes in model rates, processes and inputs were evaluated with a sensitivity analysis. The insights gained and major conclusion reached as a result of model calibration and sensitivity analysis are as follows:

1. The change in the dissolved oxygen concentrations in the bottom waters of the Bay occurring between 1965 and 1985 (increase of hypoxia and anoxia area), is due to the combined effect of increased oxygen demand and nutrient fluxes from the bottom sediment, together with phytoplankton respiration and bacterial oxidation.

2. Based on observed data and model results, phosphorus is potentially the more limiting nutrient in the upper Bay while nitrogen is potentially the more limiting nutrient in the lower Bay.

3. A nutrient budget calculated by the model indicates that the Bay bottom sediments were the largest source of dissolved inorganic phosphorus (DIP) and ammonia to the Bay during the summers of 1984 and 1985. During that time it is estimated that the bottom sediments contributed about 65% of the DIP load and 45-57% of the ammonia load to the Bay.

4. Bay water quality is controlled largely by the bottom sediment oxygen demand (SOD) and nutrient flux rates, as well as the degree of vertical stratification. The vertical stratification is primarily a function of freshwater flow. Higher summertime flows coincident with low wind velocity such as observed

during 1984, increases stratification thereby magnifying the effects of the bottom sediment oxygen demand and nutrient release with the result being lower dissolved oxygen concentrations in bottom waters and higher chlorophyll concentrations in surface waters.

5. If SOD and sediment nutrient flux rates remain as observed during the summer of 1984 and 1985, the model indicates that neither point source controls nor fall line reductions have any significant effect on main Bay water quality. This does not imply that point source controls are not a viable management strategy, since controls will reduce the amount of nutrients and organic matter deposited to the bottom sediments. Therefore, it is expected that SOD and nutrient release rates will be reduced in time. A methodology has been developed which relates projected changes in sediment oxygen demand and sediment nutrient release rates to reductions in point and nonpoint source leads. This "sediment methodology" is used in conjunction with the calibrated model to make projections of the effects of point and nonpoint source control strategies.

RELATIONSHIP OF SPRING BLOOM TO SUMMER WATER QUALITY

One of the interesting differences between the two primary years (1984 and 1985) used for model calibration involves the observed dissolved organic nitrogen (DON) concentrations. The summer 1985 dissolved organic nitrogen concentrations are approximately twice those observed during 1984. Figure 3 shows the monthly water column average DON concentration for station CB4.3C, which is located in the center channel of the Bay approximately halfway between the Potomac River and Baltimore. The summer months, June through August have been shaded for emphasis. All other monitoring stations located between the Potomac River and Baltimore exhibit similar temporal patterns of DON concentrations. Since chlorophyll-a levels during the summer of 1984 and 1985 are similar, it was not apparent why the dissolved organic nitrogen levels were so different. The following hypotheses were offered as possible explanations:

1. Higher zooplankton populations and grazing in 1985 could return more nitrogen to the organic pool.

2. Lower nutrient recycle rates due to lower bacterial populations associated with slightly lower algal biomass in 1985 might result in more dissolved organic nitrogen accumulating in the water column.

3. The spring algal bloom in 1985, which died off in May, could have released dissolved organic nitrogen from cell lysing and breakdown of detrital phytoplankton cells. A spring algal bloom did not occur during 1984 to the extent that one occurred during 1985.

Although it was believed that the spring bloom hypothesis had the most merit, it was not possible, at the time of model calibration, to conclude which, if any of the hypotheses were correct. In order to

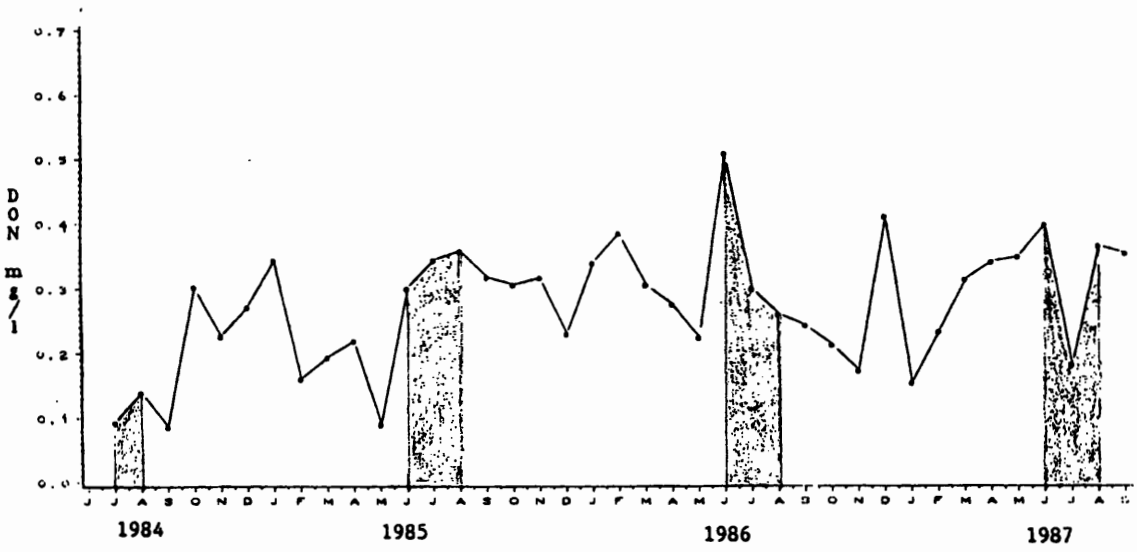

FIGURE 3. Station CB4.3C Monthly Water Column Average DON Concentration

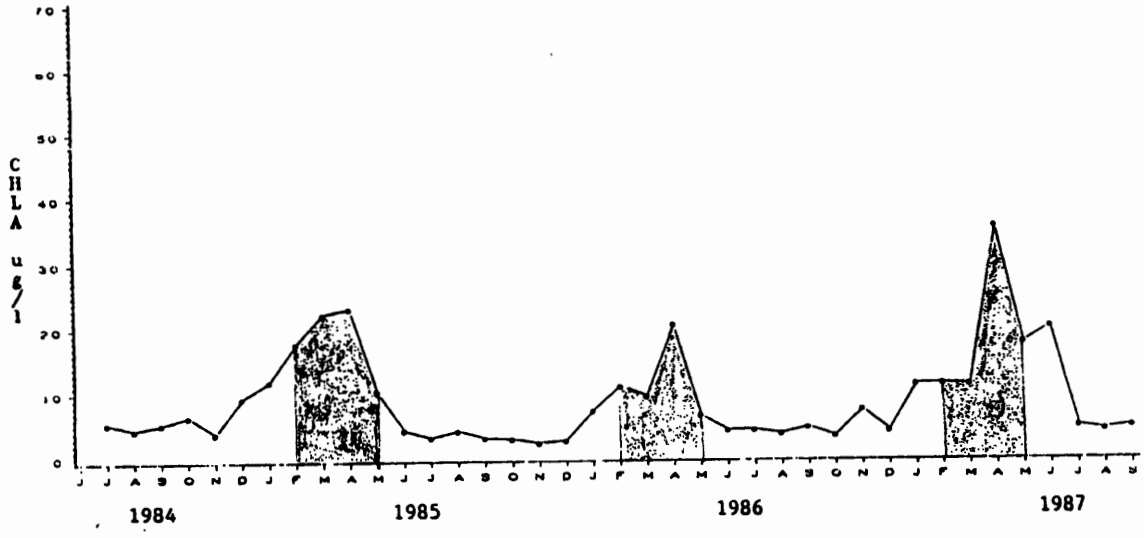

FIGURE 4. Station CB4.3C Monthly Water Column Average Chlorophyll-a Concentration

reproduce the observed summer 1985 dissolved organic nitrogen concentrations, an additional source of dissolved organic nitrogen was required. This source was provided in the form of an equivalent sediment flux of dissolved organic nitrogen, although it may not actually be present as a sediment flux, depending on which hypothesis one subscribes to. This sediment source of dissolved organic nitrogen was not required to obtain a reasonable calibration to the summer 1984 data.

Following completion of model calibration, additional model sensitivity testing showed that the sediment flux of DON used in the 1985 calibration had a significant effect on the results of control scenario projections. Considering the significant effect of the sediment flux of DON, additional testing of the spring bloom hypothesis was conducted. Water quality data, from 1984 to 1987 collected by the States of Maryland and Virginia as part of the Chesapeake Bay Monitoring Network, was evaluated. Data are not available prior to July, 1984 when the monitoring program was instituted. As shown in Figure 3, the summer concentrations of DON for 1986 and 1987, in addition to 1985 are about twice the concentration observed during the summer of 1984. In addition, for 1985, 1986, and 1987 data, there appears to be a pattern of a winter peak with late winter decline followed by a summer peak. Although the 1987 data show a double peak, one in winter and one in the spring, with subsequent decline followed by a summer increase, the basic pattern is the same as 1985 and 1986.

When the chlorophyll'a' data is considered, it is apparent that 1985, 1986 and 1987 all had significant spring phytoplankton blooms, which subsequently died off in late spring. Figure 4 shows the monthly water column average chlorophyll-a concentration for the representative Station CB4.3C. The spring period, February through May has been highlighted for emphasis. Each of these years, as previously discussed, also exhibited increasing DON concentrations from spring to summer. Therefore, the spring bloom/high summer DON hypothesis appears to have some validity.

But why were the 1984 summer DON levels so much lower than 1985 through 1987? To help answer this question, phytoplankton data collected during the spring, 1984 was obtained from Dr. Larry Harding, Chesapeake Bay Institute, Johns Hopkins University. This data confirms the fact that chlorophyll-a levels during the spring of 1984 were on the order of 5-10 ug/l, significantly less than the 20 ug/l - 35 ug/l level observed during the spring of 1985, 1986 and 1987. The lower chlorophyll-a values during the spring of 1984 appear to be the result of the much higher freshwater flows during 1984, which increase turbidity and decrease detention time.

The implications of the sediment flux of dissolved organic nitrogen have been determined to be significant. When model projections are made regarding the impact of nutrient control strategies, using the calibrated model for 1985, nitrogen control, in addition to even a modest degree of phosphorus control is shown to have little effect. Figure 5 shows the model prediction of chlorophyll-a for a phosphorus only control strategy and a phosphorus plus nitrogen control strategy compared to 1985 existing conditions. As can be seen, nitrogen control in addition to phosphorus control produces little benefit north of the Potomac River (km 110). Model projections of dissolved

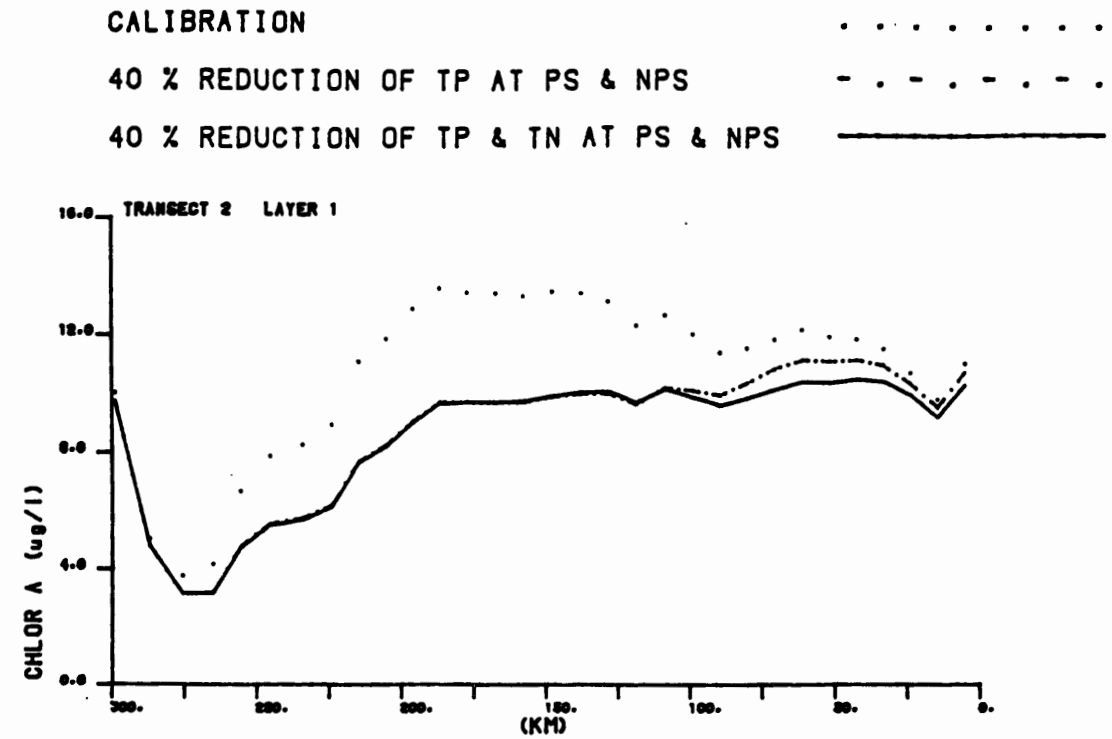

FIGURE 5. 1985 Calibration Model Projections of Chlorophyll-a Along Center Transect of Chesapeake Bay

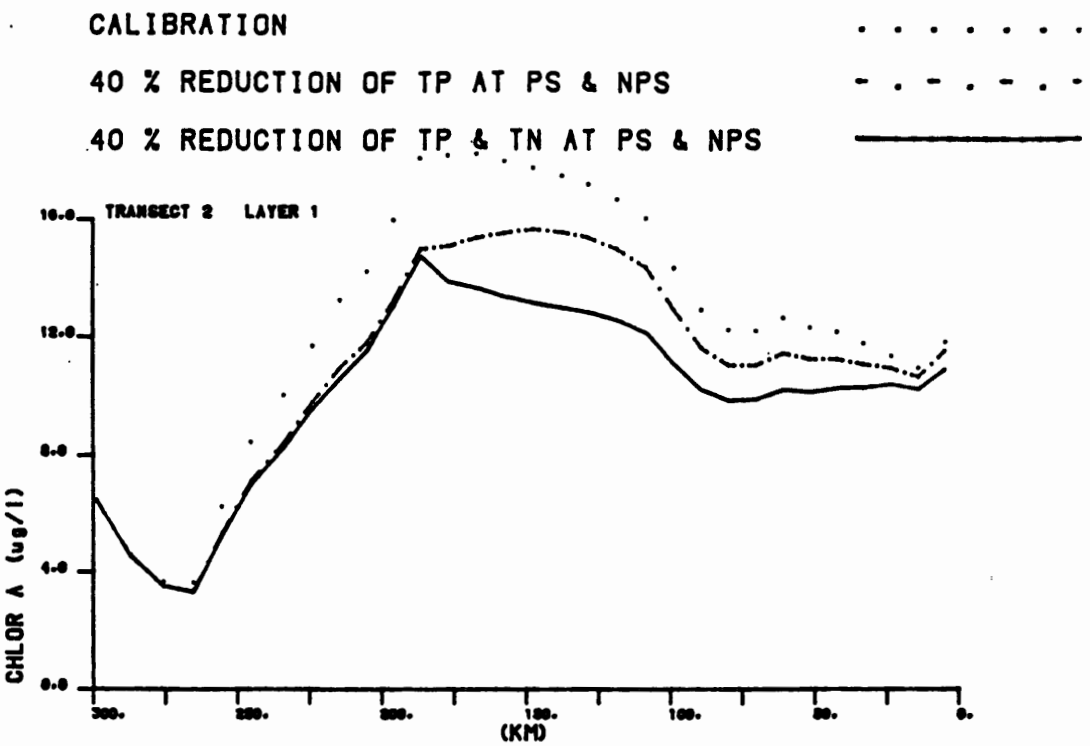

FIGURE 6. 1984 Calibration Model Projections of Chlorophyll-a Along Center Transect of Chesapeake Bay

oxygen likewise show a negligible improvement resulting from nitrogen controls. The reason this happens is that the sediment flux of DON, which is maintained as a constant flux in the projections, supplies a sufficient nitrogen load which, when coupled with even a small amount of phosphorus control, produces a phosphorus limited condition in the area of the Bay north of the Potomac River. It should be noted the observed data as well as the model used for projection clearly show that during the summer of 1985, the area of the Bay between the Potomac River and Baltimore was in fact nitrogen limited. Therefore, this entire area has shifted from a nitrogen limited condition to a phosphorus limited condition with just a small degree of phosphorus control. Additional significant removal of nitrogen is not enough to overcome the effects of the sediment flux of DON and produce nitrogen limitation in this area.

Using the calibrated model for 1984, which did not have a sediment source of DON, one finds that nitrogen control strategies are effective as far north as the Bay bridge (km200). This is also the same area in which both the observed data and the model indicates as nitrogen limited during summer 1984 conditions. Figure 6 shows the model prediction of chlorophyll-a for phosphorus control only and phosphorus plus nitrogen control scenarios. As can be seen, control of nitrogen plus phosphorus results in about twice the improvement achieved by a phosphorus only strategy.

To more directly test the sensitivity of the 1985 calibrated model projections to the sediment source of DON, this source was removed and the projections rerun. Not surprisingly, when the sediment DON flux was set to zero, the control of nitrogen in addition to phosphorus, produced very significant improvements in chlorophyll-a (additional

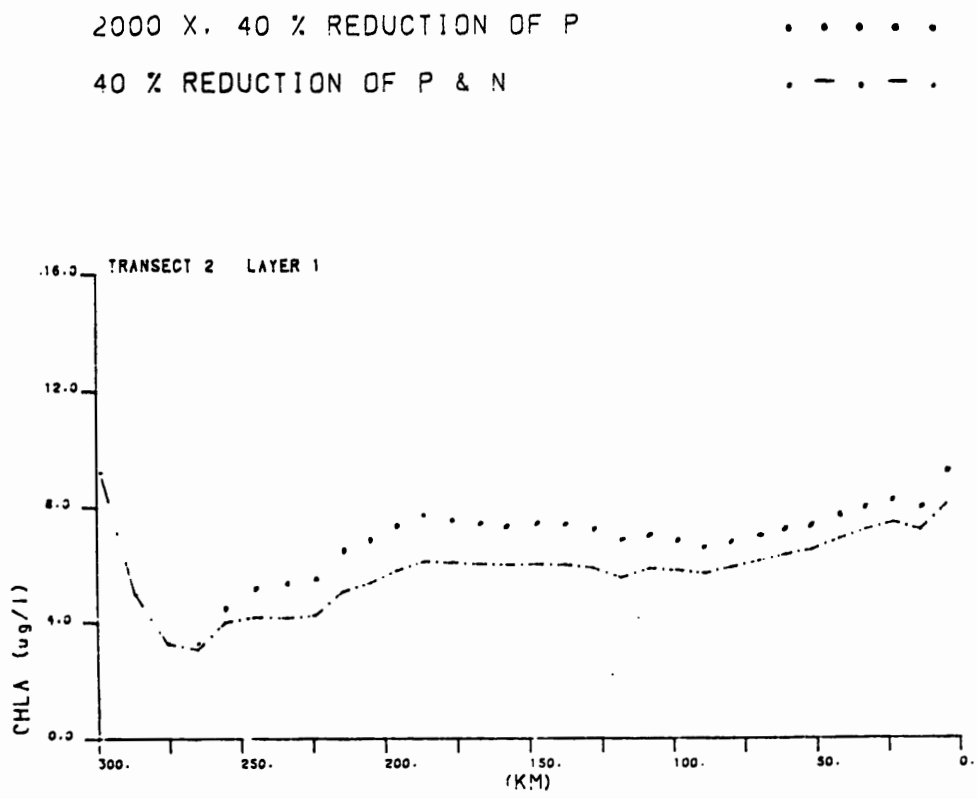

FIGURE 7. Model Projections (1985 Circulation) of Chlorophyll-a with Sediment Flux of DON Removed

reduction of 1-1.5 ug/l) and dissolved oxygen (additional increase of about 1.2 mg/l). Figure 7 shows a model prediction of chlorophyll-a in which control of phosphorus only is compared to control of phosphorus plus nitrogen. As can be seen, the incremental effect of nitrogen control now extends up the Bay as far north as Baltimore (km250). This is contrasted to the previous model projection which included a sediment flux of DON. In that previous projection, nitrogen control had little impact north of the Potomac River (km110). Therefore, it can be concluded that the sediment flux of DON has a very significant impact on determining which nutrient and to what degree that nutrient controls algal growth in the Bay between the Potomac River (km110) and Baltimore (km250). Based on the hypothesis that the spring bloom of phytoplankton significantly controls the level of DON in the water column during the summer it appears that control of the spring bloom may be an important factor in determining the effectiveness of nutrient controls, especially nitrogen, during the summer.

CONCLUSIONS

The steady state eutrophication model of Chesapeake Bay even with the inherent limitations imposed by a steady state analysis, has provided many useful insights to the Chesapeake Bay's water quality problems related to nutrient enrichment. The importance of the Bay bottom sediments and the potential importance of the spring bloom has been highlighted. This information has been used to focus both the Bay monitoring program for 1988 and the time variable eutrophication modeling project which the EPA Chesapeake Bay Program initiated in the fall of 1987.

ACKNOWLEDGMENTS

The assistance of Lewis Linker and Chao-Hsi Chang of the Chesapeake Bay Liaison Office and Joanie Blackwood of Computer Sciences Corporation in evaluating and plotting data is gratefully acknowledged. The support provided by Robert Reynolds of Computer Sciences Corporation in conducting the model simulations is likewise appreciated. Recognition and appreciation are also extended to Barbara Edwards and Josephine Watson, EPA Region 3 for patiently typing and editing this paper.

REFERENCES

Harding, Lawrence W., Jr. Personal Communication of Data, 1988.

Hydroscience, Inc. "The Chesapeake Bay Waste Load Allocation Study", Prepared for the Maryland Department of Natural Resources, 1975.

HydroQual, Inc. "A Steady-State Coupled Hydrodynamic/Water Quality Model of the Eutrophication and Anoxia Process in Chesapeake Bay" Prepared for the U.S. E.P.A. Chesapeake Bay Program Under Contract to Battelle Ocean Sciences, August, 1987.

HydroQual, Inc. "Nutrient Limitation in Chesapeake Bay" Technical Memo, January, 1988.

Linker, L.C. and Chang, C. "Variation in Chesapeake Bay Nutrient Limitation: An Analysis of Chesapeake Bay Monitoring Data" U.S. E.P.A. Chesapeake Bay Liaison Office, February, 1988.

U.S. E.P.A. "Chesapeake Bay: A Framework for Action" Region 3, September, 1983.

U.S. E.P.A. "Chesapeake Bay Program: Findings and Recommendations" Region 3, September, 1983.

Understanding the Estuary: Advances in Chesapeake Bay Research. Proceedings of a Conference. 29-31 March 1988. Baltimore, Maryland. Chesapeake Research Consortium Publication 129. CBP/TRS 24/88.

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.

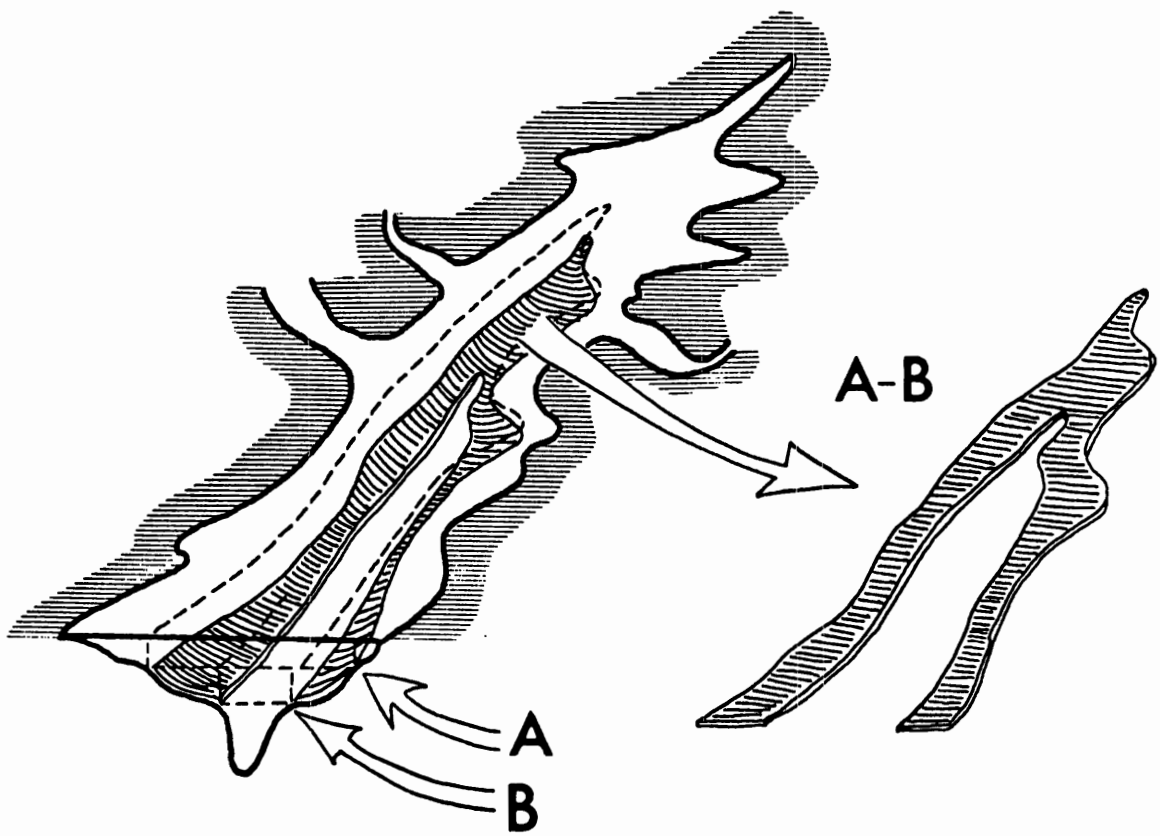


Figure 1: Schematic representation of incremental habitat improvement. "A" scenario: unfavorable conditions extend high in the water column. "B" scenario: unfavorable conditions are reduced and are found only at greater depth. "A-B" differential: represents an increment of Bay bottom which is now potential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

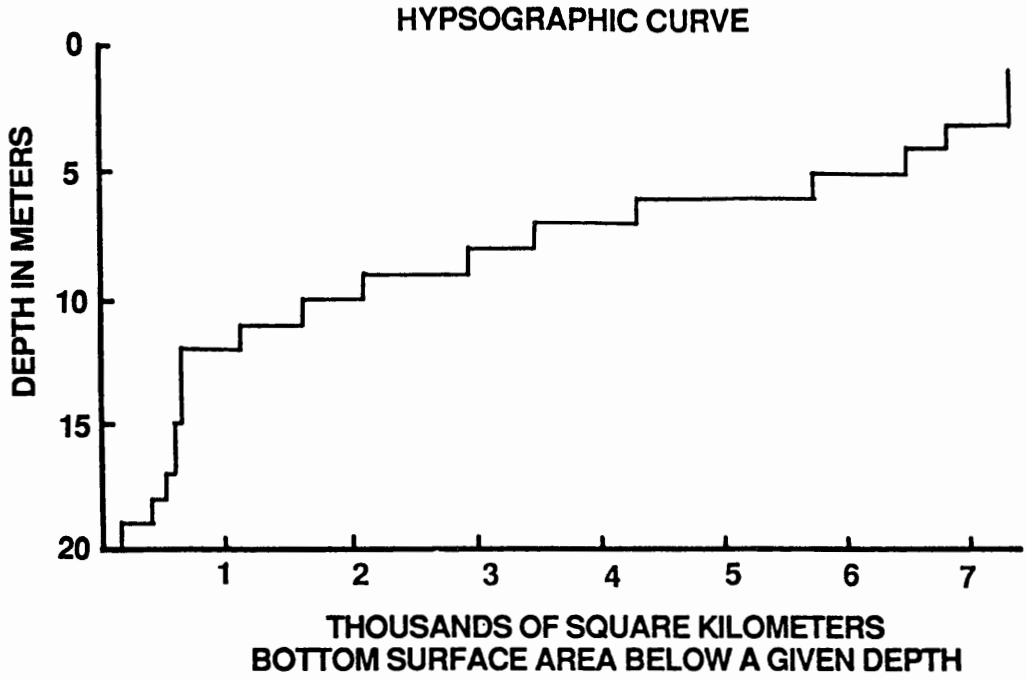


Figure 2

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 599). 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.

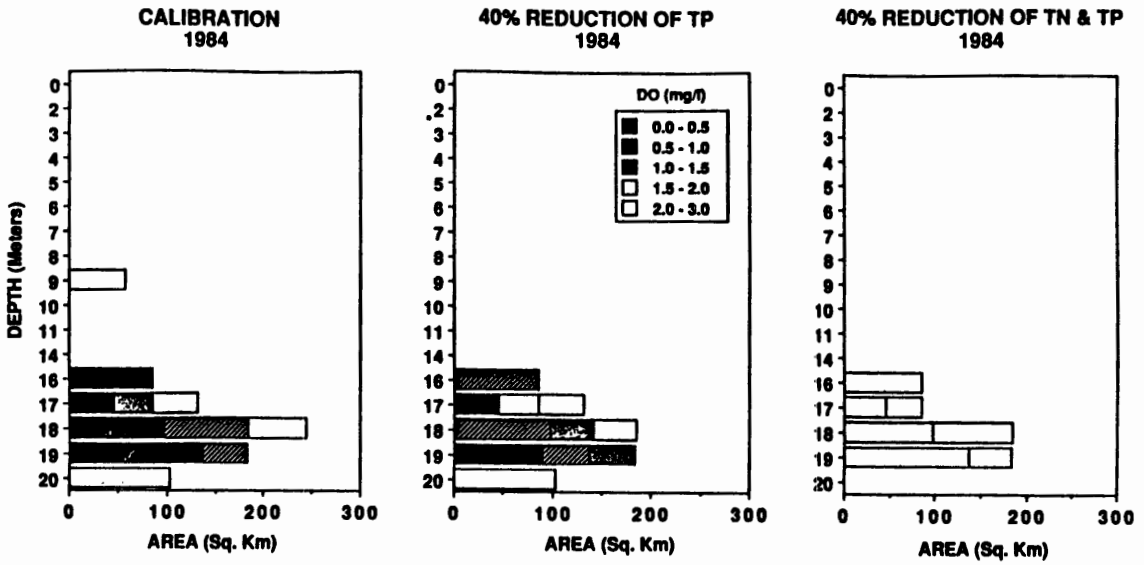


Figure 3

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 7.4×10^4 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 9.4×10^4 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 ≤ 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.

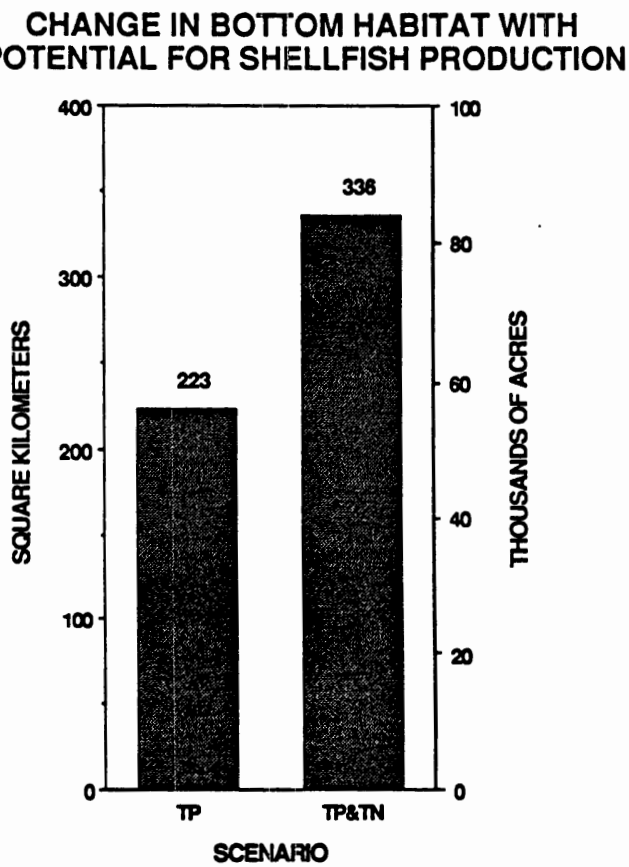


Figure 4:
Increases in bottom area with potential for shellfish production projected for two management scenarios; 40% reduction in total phosphorus loading (TP) and 40% reduction in both total nitrogen and total phosphorus (TP+TN).

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.

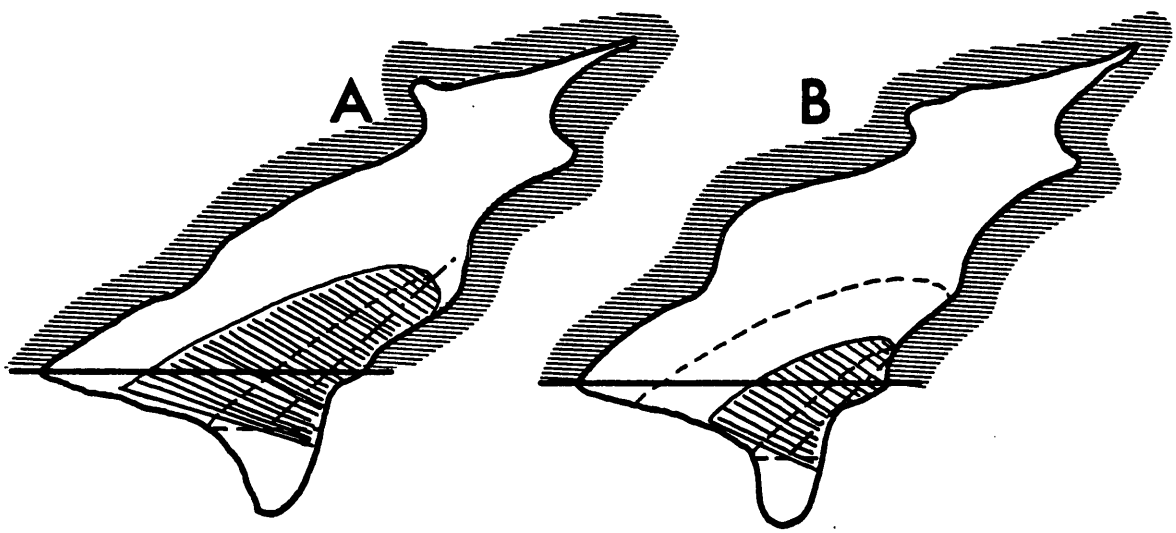


Figure 5: Schematic representation for pycnocline tilting events in Chesapeake Bay; "A" scenario in which unfavorable conditions reach high in the water column and tilting exposes large "shelf" areas to low oxygen; "B" scenario, following an incremental improvement, shows unfavorable conditions are lower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

HABITAT IMPROVEMENTS DURING PYCNOCLINE TILTING EPISODES

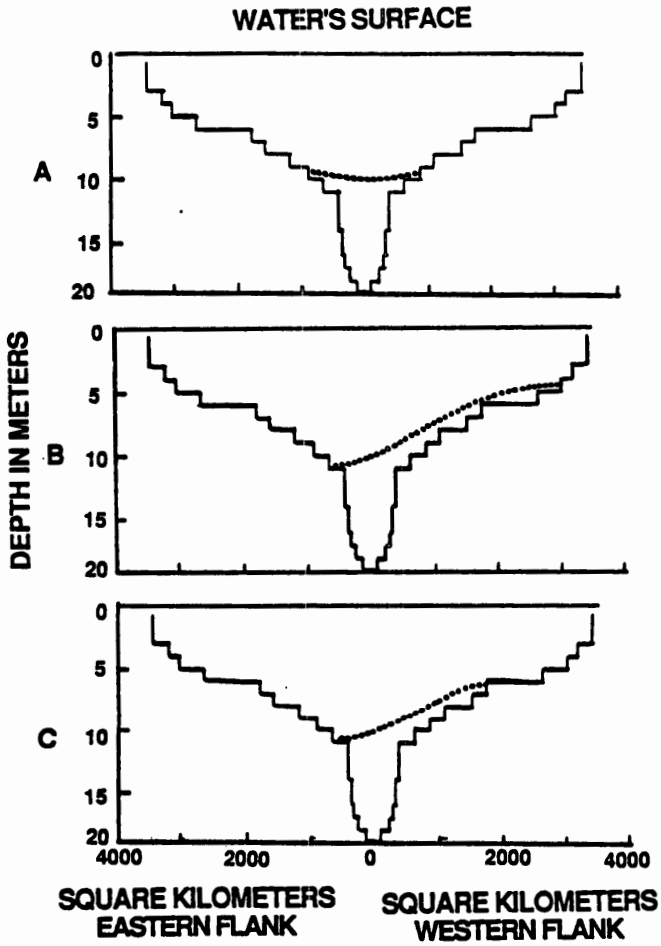
A second major environmental process, one not modeled by our current tools, is the phenomenon of pycnocline tilting. This process is shown schematically in Figure 5, version (A) representing tilt to some fixed angle with unfavorable conditions existing higher in the water column and version (B) with 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

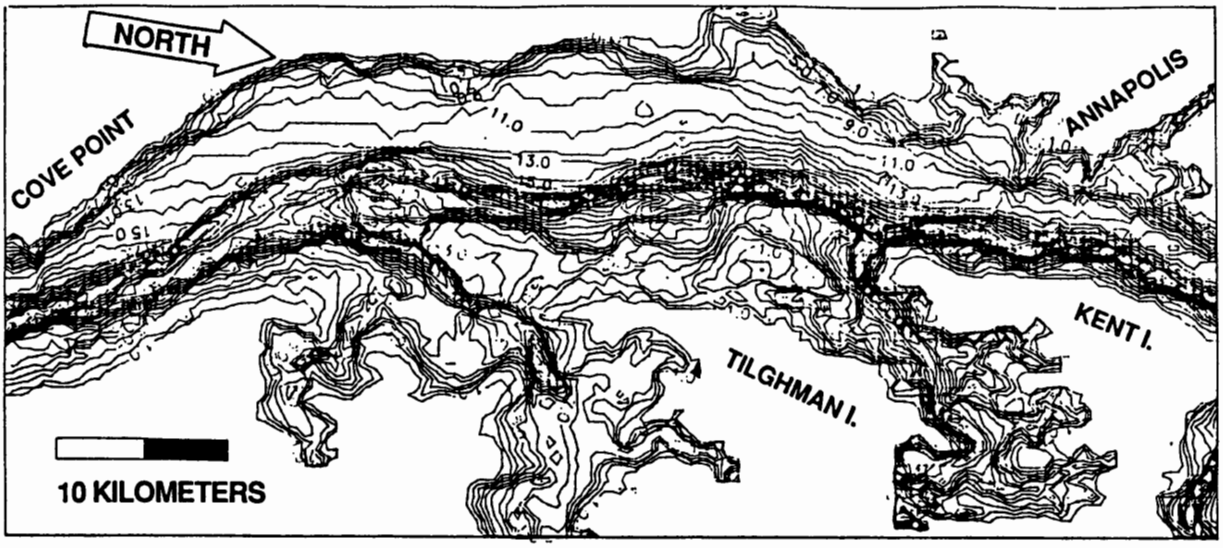


Figure 7: Middle (Maryland) reach of Chesapeake Bay with 1 m contours shown (Data Source: Goldsmith and Hutton 1977).

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 (≤ 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.

As noted above, disease has severely impacted shellfish communities in the Bay and it is possible that improved DO levels in subpycnocline habitats may not be sufficient to overcome the positive association of disease organisms with higher salinity.

There is no a priori reason to assume that in the future, following implementation of control scenarios, Chesapeake Bay will behave according to the kinetics used in this model; a construct based on observations from past years. As a working assumption, however, the Bay has shown considerable homeostasis in the past as man disrupted its once accustomed operation and it is likely to do so in the future. We believe that, if anything, the synergisms which might emerge would be more progressive than retrogressive.

REFERENCES

App, C.; Fitzpatrick, J. Insight to the Chesapeake Bay's Eutrophication Process through Modeling. (in) Understanding the Estuary: Advances in Chesapeake Bay Research. Ches. Research Consortium, Gloucester Pt., VA. 23062; 1988.

Barnes, H. Apparatus and Methods of Oceanography, Part 1: Chemical. George Allen and Unwin, Ltd., London; 1959: 341 p.

Beck, A.; Hanson, D.; Personal communication, USEPA Narragansett, RI 02882; 1987.

Breitburg, D.; Personal communication. Academy of Natural Sciences of Philadelphia, Benedict, MD 20612; 1987.

Cronin, W. Volumetric, Areal, and Tidal Statistics of the Chesapeake Estuary and its Tributaries. 1971; Special Report 20: p. 135. Chesapeake Bay Institute, Johns Hopkins University.

Coutant, C., Personal communication and seminar at Chesapeake Bay Program; Annapolis, MD; Dissolved oxygen and *Morone saxatalis* habitat; 1987

Fair, G.M.; Geyer, J.C. Water Supply and Waste-water Disposal, NY: John Wiley & Sons, Inc.; 1963: p. 973

Bathymetry of the Chesapeake Bay; Bathymetric Chart Series No. 2.; Goldsmith, V.; Hutton, C.H.; 1977. Virginia Inst. Mar. Sci., Gloucester Pt., VA 23062.

Holland, A.F.; A. Shaughnessy. Personal communication. Versar/ESM Inc., Columbia, MD. 21045; 1987

HydroQual, Inc. A Steady-State Coupled Hydrodynamic/Water Quality Model of the Eutrophication and Anoxia Process in Chesapeake Bay. Volume I and Appendix. Batelle Ocean Sciences, Duxbury, MA.: 1987

Magnien, R.; Personal communication. Maryland Department of the Environment, Baltimore, MD. 21203; 1988

Mann, R.; Meehan, B.; Ranier, J.S.; Kennedy, V.S.; Newell, R.E.I.; Van Heukelem, W.F. Influence of Low Oxygen Tensions on Larvae and Post Settlement Stages of the Oyster *Crassostrea virginica.* in G. Mackiernan, Ed. Dissolved Oxygen in the Chesapeake Bay: Processes and Effects. Univ. MD. Sea Grant, College Park, MD. 177 p.; 1987

Mountford, K.; Mackiernan, G. A Multi-decade Trend Monitoring Program in Chesapeake Bay, a Temperate East Coast Estuary. (in) ESA/ASLO Symposium on Long Term Monitoring of Aquatic Systems. T.P. Boyle (Ed.) Amer. Soc. Testing and Mater., Phila., PA. Spec. Pub. 941: 91-106; 1986

Mountford, N.K.; Holland, A.F.; Mihursky, J.A. Identification and description of macrobenthic communities in the Calvert Cliffs region of the Chesapeake Bay. Ches. Sci. 18-4:360-369; 1977

Mutman, M.; Masse, A.N. Investigation of Nutrient Sources. Potomac River. U.S. Envir. Prot. Agency, Nat'l Enforcement Investig. Center, Denver, CO. EPA-330:2-85-001; 1985

Newell, R.E.I. in G. Mackiernan, Ed. Dissolved Oxygen in the Chesapeake Bay: Processes and Effects. Univ. MD. Sea Grant, College Park, MD. 177 p.; 1987

Sale, J.W.; Skinner, W.W. The vertical distribution of dissolved oxygen and the precipitation by salt water in certain tidal areas. J. Franklin Inst. 148 (1104-58): 837-848p.; 1917

Sanford, L.; Sellner, K.; Bundy, M. Moored measurements of Dissolved Oxygen in the Chesapeake Bay during the Summer of 1987. AGU Ocean Sciences Meeting. New Orleans, LA. 1987

Seliger, H.; Boggs, J.; Biggley, W. Catastrophic Anoxia in the Chesapeake Bay in 1984. Science. 228:70-73p.; 1985

Seitzinger, S. P. The effect of pH on the Release of Phosphorus from Potomac River Sediments. Report 86-8F Acad. Nat. Sci. Phila. Publ. by U.S. Envir. Prot. Agency. Chesapeake Bay Prog., Annapolis, MD. 21403; 50 p.; 1985

USEPA. Chesapeake Bay: A Profile of Environmental Change. Appendices Sec. 5 Trends in Dissolved Oxygen. U.S. Environ. Prot. Agency. Region III. Phila., PA. 19106; 1983a

USEPA. Chesapeake Bay: A Profile of Environmental Change. U.S. Environ. Prot. Agency. Region III. Philadelphia, PA. 19106; 200 p.; 1983b

Understanding the Estuary: Advances in Chesapeake Bay Research. Proceedings of a Conference. 29-31 March 1988. Baltimore, Maryland. Chesapeake Research Consortium Publication 129. CBP/TRS 24/88.

Abstract only

Variability of Dissolved Oxygen in the Mesohaline Chesapeake Bay

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Time series measurements of dissolved oxygen (DO) concentration were made at 5 locations (4 stations) across the mid-Chesapeake Bay and up the axis of the lower Choptank River during the summer of 1987. The data were collected with ENDECO Pulsed DO Sensors, sampled at 5-15 minute intervals. Near continuous records were obtained from 17 July to 9 September. In the best case, an estimated accuracy of about 0.1 mg/l was achieved. Simultaneous measurements of temperature, salinity, and current speed and direction also were made at most locations.

The data reveal a remarkable degree of short term variability in DO concentrations; fluctuations of 1-5 mg/l over intervals of minutes to days are not uncommon. The standard deviation of near-bottom DO concentration is 1-2 mg/l on the eastern and western flanks of the Bay, and less than 1 mg/l on the eastern side of the deep trough. Records from different locations are both qualitatively and quantitatively different. Comparison to salinity and current data indicates that much, but not all, of the observed high frequency variability is advective. Physical processes that may lead to advective changes in DO are high salt/low DO intrusion, horizontal advection of spatial patchiness, internal waves, and tidal mixing or resuspension.

Understanding the Estuary: Advances in Chesapeake Bay Research. Proceedings of a Conference. 29-31 March 1988. Baltimore, Maryland. Chesapeake Research Consortium Publication 129. CBP/TRS 24/88.

Metabolic and Respiratory Compensation During Long Term Hypoxia in Blue Crabs, *Callinectes sapidus*

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ABSTRACT

The present work describes physiological mechanisms that permit blue crabs, Callinectes sapidus Rathbun, to survive hypoxia for as long as 25 da. Blue crabs were exposed to hypoxia (45 to 55 mmHg) at 21-23 °C, 16 °/oo for as long as 25 days. The initial response to hypoxia was hyperventilation and an increase in heart rate. Both increases persisted for several days in hypoxia, but by 5 days, there was evidence of abatement of the responses. Despite the increase in blood flow and in water flow over the gills, blood oxygen tension fell from ca. 98 to 18 mmHg in the first 24 hours. Simultaneously, total carbon dioxide increased twofold and pH remained stable, both responses continuing unchanged for the entire exposure. The internal hypoxia was not so severe as to result in a significant increase in blood lactate, suggesting that metabolic demands were met aerobically. Mitochondrial density was greatly reduced in muscle tissue of crabs exposed to hypoxia for 20 da., indicating that some of the long term changes may not be readily reversible upon return to normoxia. The compensatory response of blue crabs to hypoxia occurs during the first 24 h and persists for several days while other, metabolic mechanisms are initiated. The metabolic adjustments include alteration of the subunit composition of the respiratory pigment, hemocyanin, changing the equilibruim point of the hemolymph acid-base system, and resorption of most of the mitochondria present in skeletal muscle.

INTRODUCTION

The effect of ambient oxygen levels on respiratory function of aquatic decapod crustaceans has been the subject of investigation for more than five decades (Fox and Johnson, 1934). Few investigators have examined the response of decapods to hypoxia of more than a few hours, and previous such work is on macruran decapods (McMahon et al., 1974; McMahon et al., 1978; Butler et al., 1978; Wilkes and McMahon, 1982a, 1982b). Generally, the two types of responses which animals exhibit during hypoxia are classified as respiratory or metabolic. Respiratory responses are those which increase the uptake and/ or transport of oxygen via changes in ventilation or perfusion of the respiratory surface, or by reducing the diffusion barriers at some point in the oxygen transport circuit. Metabolic responses are those involving a cellular process affecting oxygen demand, carbon dioxide production or the level of an ion or molecule that modifies respiratory function (i.e. hydrogen ion concentration).

Both lobster (McMahon et al., 1978; Butler et al., 1978 and crayfish (McMahon et al, 1974; Wilkes and McMahon, 1982a, 1982b) increase the ventilatory flow of water through the branchial chamber and increase heart rate during hypoxia. These responses enhance the convective transport of oxygen during conditions of limited oxygen supply. Lobster and crayfish both utilize compensatory respiratory responses during the short term, but make metabolic adjustments over longer periods (>3 da.). These metabolic responses modify the oxygen affinity of the respiratory pigment, hemocyanin, but via different routes. The lobster exhibits an increase of pH for at least part of the hypoxia, and crayfish elevate oxygen affinity via an undetermined mechanism. At least one other species, the brown shrimp Crangon crangon, is capable of actually incrasing hemocyanin concentration when exposed to moderate hypoxia for 20 da. or longer (Hagerman, 1986).

The present research extends the observations that have been made on macruran species to the euryhaline blue crab, Callinectes sapidus Rathbun. Blue crabs may be exposed to conditions of low oxygen annually, when part of the central Chesapeake Bay becomes anoxic during the summer months. Areas adjacent to or overlying the anoxic zone may have severely reduced oxygen levels, and animals such as blue crabs, migrating through these areas will be exposed to hypoxia for at least hours, if not longer. Even 2 h exposure to 10% air saturated water results in a substantial increase in hemolymph lactate (Lowrey and Tate, 1986; Lallier et al., 1987). Considering that lactic acid is the end product of anaerobic metabolism in crustaceans (Phillips et al., 1977), it is clear that hypoxic exposure limits the metabolic scope of these animals. The extent and nature of the limitation is one of the topics presently under investigation.

METHODS

Intermolt male blue crabs (110-280 g.) were collected from local waters near the mouth of the Rhode and Patuxent Rivers, MD using a beach seine, crab pot or oyster dredge. Crabs were collected by hand when possible, however, during the winter months, it was necessary to obtain the animals from local fisherman. Crabs were returned to George Mason University and maintained in recirculating, aerated seawater (P_IO_2 =130-140 torr; 16°/oo) at 20-21°C. Upon return to the laboratory, the animals were acclimated for approximately 1 week and were fed a diet of frozen fish prior and during experiments, except those animals in which ventilation was measured.

Blue crabs were exposed to hypoxia (33% air saturation or less, 45-55 torr) for 5 to 25 da. using nitrogen gas to displace oxygen from the water. Hypoxia was maintained using a control system which used the output of an oxygen electrode and meter (Instrumentation Lab.) as the input signal upon which was based the control of an electric gas valve. The control system was set to maintain ambient oxygen at approximately 50 torr. The controller balanced ambient oxygen by allowing nitrogen gas to flow through a regulator valve only when P_IO_2 rose above 50 mmHg.

Two aspects of the responses to low oxygen were examined. First, the ventilatory and circulatory responses were determined by measuring ventilation and heart rate. Measurements were made only during the shorter term experiments in which crabs were exposed to hypoxia for 5 da. In 7 da. exposure and in the longer experiments, hemolymph (blood) variables were measured before, during and, in some cases, following hypoxia. Hemolymph oxygen tension (PO_2), total CO_2, pH, lactate, protein and inorganic ions were measured from postbranchial hemolymph samples.

Heart rate was measured using an impedance technique (Ansell, 1973) in which two fine (0.005") teflon coated stainless steel wires were implanted over the heart. The signal resulted from the change in impedance due to heart beating, detected using an impedance converter. The signal from the converter (UFI model 2991) was amplified and displayed on a pen recorder (Narco model Mk-IV).

Water flow over the gills was measured directly using the technique described by deFur and McMahon (1984) in which the water flow was funnelled through a polyethylene "mask" and flow detected by an electromagnetic flow probe and flowmeter (BLI model 610). The flowmeter and probe system was calibrated using a manometer (Gilson). The output from the flowmeter was amplified and displayed on the same pen recorder as heart rate. Recordings of ventilation were analyzed in entirety by measuring the area under the flow recording, using a digitizing scanner (Hewlett-Packard 9825A and 9874A).

Postbranchial (arterial) hemolymph was sampled via holes drilled in the carapace dorsolateral to the heart and covered with latex rubber affixed with cyanoacrylate. Animals were permitted at least three days recovery from preparations before being used in an experiment. Hemolymph samples were withdrawn into iced 1 ml glass syringes which were immediately replaced on ice to slow the clotting. The following hemolymph variables were measured, though not all variables were measured on all samples: pH, oxygen tension (PO_2), total carbon dioxide (Cco_2), lactate concentration, protein, osmolality and chloride. Procedures previously published were used to measure all variables and the reader is referred to papers by deFur *et al.* (1988) and Mangum *et al.* (1985) for details. *In vivo* pH, Po_2 and Cco_2 were measured immediately, and other variables were measured later on prepared samples. Protein was measured using the Biuret method and lactate was determined with the modifications suggested by Graham *et al.* (1981).

The crabs used for electron microscopy were obtained in August 1987 from the Patuxent River, MD. Tissues were fixed using a combination of vacular perfusion and immersion. Crabs were removed from the aquarium, placed into a finger bowl of seawater with 2% glutaraldehyde, bufferred to pH 8.0. Fixative was injected into the pericardial sinus and into the second joint of the swimming appendage. The exoskeleton was then removed from the second segment of the fifth leg, the tissue removed and cut into 0.5mm cubes, and placed into vials of fixative. The vials were placed on a slow shaker and gently agitated during fixation at 20°C for 9 h. The tissue was then washed three times in bufferred seawater, post-fixed in 4% osmium tetroxide and incubated for an additional 5 h. The tissue was washed three times in bufferred seawater, dehydrated in ETOH, placed in propylene oxide and embedded in "Spurs" embedding plastic. Sections were stained with 2% uranyl acetate and Reynold's lead citrate for 10 min each and examined on a JEOL 100c electron microscope.

RESULTS

The respiratory responses (ventilation and circulation) during five days hypoxia were initiated largely during the first 24 h of hypoxia (Figs. 1 and 2). Mean ventilation increased approximately 100% (Fig. 1) as heart rate was elevated by 20% (Fig. 2). The changes in mean ventilation between days 1 and 4 of hypoxia were not significant, but mean ventilation on day 5 was significantly lower than the highest value, recorded on day 2 of hypoxia. Heart rate was also significantly elevated during the first three da. of hypoxia, but declined thereafter and was not significantly different from mean heart rate before hypoxia. By 24 h after returning to normoxia (150 mmHg), mean ventilation and heart rate had declined to values which were similar to and not significantly different from initial values.

Ventilation also was measured in three animals during seven days of hypoxic exposure and one day of recovery in normoxia . The individual responses of these crabs were consistent with those observed in the crabs exposed to hypoxia for 5 da. Unlike the results from the 5 da. exposure there was considerable variability in the responses during days 4 to 7. Upon return to normoxia, there was a consistent decrease in ventilation, as in the other experiment.

Postbranchial (arterial) oxygen tension (Po_2) decreased dramatically from a mean of nearly 100 mmHg to only 18.1 mmHg by 24 h of hypoxia (Fig 2). This internal hypoxia was accompanied by an increase in total carbon dioxide (Cco_2), approximately 95% of which exists in the hemolymph as HCO_3^- (Truchot, 1976). Hemolymph pH did not change significantly throughout the hypoxia or after return to normoxia (Fig.2) despite the increased ventilation and elevated total CO_2.

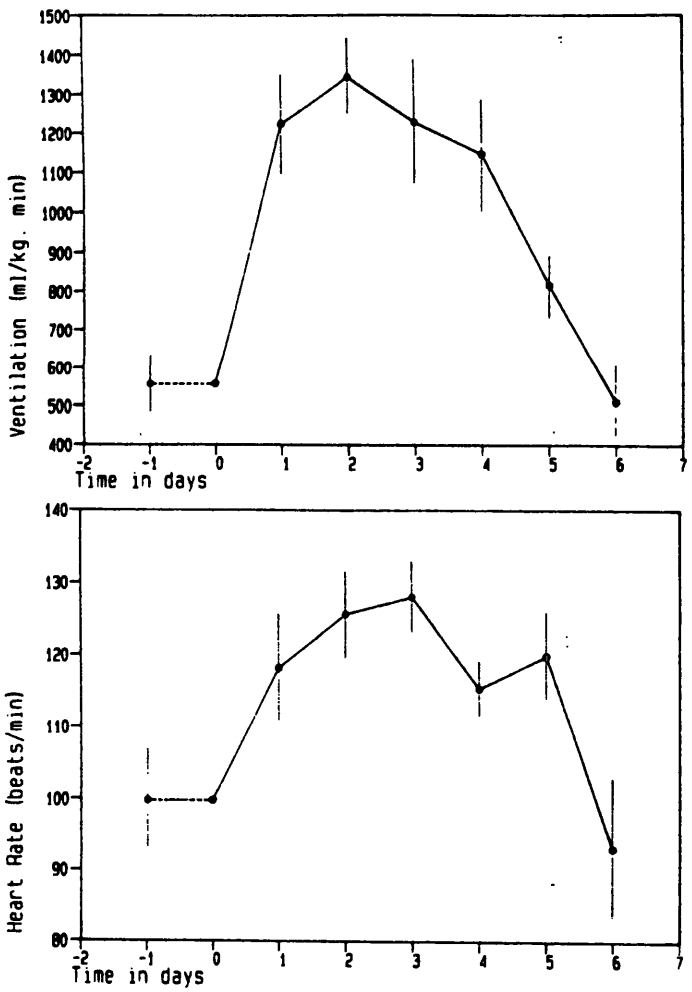

Fig.1. Ventilation (upper) and heart rate (lower) of blue crabs in hypoxia (50 mmHg),22 C, 16°/oo. Data are mean values for 7 crabs,+S.E.. Control value (day-1 is for 1 da normoxia, carried to beginning of hypoxic exposure. Day 6 is recovery.

During exposure to hypoxia for as long as 25 da., there were no changes in hemolymph osmolality, chloride ion concentration, or total protein concentration. Likewise, we did not observe any significant increase in the hemolymph concentration of lactic acid, the end product of anaerobic metabolism in decapod crustaceans (Phillips, 1977).Hemolymph pH and PO_2 also were unaffected by the longer hypoxia exposure period.

There were profound changes in the ultrastructure of the striated muscle taken from the swimming appendage of crabs exposed to hypoxia for 20 da. Mitochondria were so reduced in density that it was necessary to scan numerous tissue grids

to locate any. Mitochondria present were substantially smaller than those in muscle from normoxic crabs. Although not quantified, there seemed to be an increase in the total cross sectional area occuppied by sarcoplasmic reticulum.

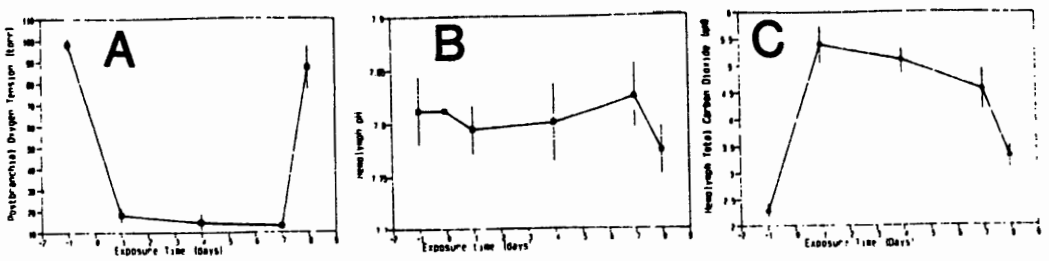

Fig. 2. Postbranchial oxygen tension (A), pH (B), and total CO_2 (C) of blue crabs in hypoxia (50 mmHg) for 7 da, 22 C, 16 °/oo. Data are X ± S.E.,n=8. Day data 8 are recovery.

DISCUSSION

The response to hypoxia in blue crabs is an increase in water flow over the gills and blood flow through the gills, this latter assumes that cardiac stroke volume does not decrease. These responses are compensatory in elevating the rate of delivery of oxygen to the site of oxygen uptake and to the site of utilization. The response is obviously effective over at least 5 da. and perhaps somewhat longer. There was some evidence for an increase in the effectiveness of oxygen transport across the respiratory surface during hypoxia. The diffusion gradient decreased from approximately 40 mmHg in normoxia to ca. 30 mmHg in hypoxia, a decrease of 25%. Nevertheless, the oxygen supply to the tissues was sufficient to meet the metabolic demands aerobically, as evidenced by the absence of lactic acid build up in the hemolymph.

The decrease in ambient oxygen tension undoubtedly limits the oxygen transport capacity of the respiratory system. At the low arterial Po_2 measured here, the respiratory pigment, hemocyanin, may not be fully saturated with oxygen at the gills. The oxygen affinity of blue crab hemocyanin is approximately 12 mmHg under the present experimental conditions (deFur et al., 1988). Thus, at an arterial Po_2 of 18 mmHg, the hemocyanin will be nearly fully saturated in transit through the gills. As arterial Po_2 decreases further, however, the loading of oxygen at the respiratory surface will be compromised unless oxygen affinity is increased. There is a decrease in P_{50} of the hemocyanin (deFur et al., 1988), but the timing is not exactly coordinated with the further decrease in arterial Po_2 from 18 to 12 mmHg (Fig. 2) and the decrease in ventilation (Fig. 1). The increase in affinity occurs between 7 and 20 days (deFur et al., 1988) permitting an increase in oxygen saturation of hemocyanin at the gill.

Other aquatic decapods, including the related species, Carcinus maenas, (Lallier et al, 1987) exhibit an alkalosis during declining oxygen. McMahon et al. (1978) concluded that the initial alkalosis in lobster resulted from the hyperventilation which increased the loss of dissolved carbon dioxide at the gills. Burnett and Johansen (1981), however, addressed that point by artificially ventilating crabs at a range of flow rates to determine the relationship between flow and hemolymph pH. They concluded that ventilation alone did not determine pH under the experimental conditions used. The present results are consistent with that conclusion and suggest that metabolic mechanisms act in concert with respiratory ones. In the present work, such metabolic mechanisms may be changes in levels of Ca++, Na+ or in the buffering properties of hemocyanin with a different subunit composition.

LITERATURE CITED

Ansell, A.D. 1973. Changes in oxygen consumption, heart rate and ventilation accompanying starvation in the decapod crustacean Cancer paguris. Neth. J. Sea Res. 7: 455-475.

Burnett, L.E. and K. Johansen. 1981. The role of branchial ventilation in hemolymph acid-base changes in the shore crab Carcinus maenas during hypoxia. J. Comp. Physiol. 141: 489-494.

Butler, P.J., E.W. Taylor and B.R. McMahon. 1978. Respiratory and circulatory changes in the lobster (Homarus vulgaris) during long term exposure to moderate hypoxia.J. Exp.Biol.73:131-146.

deFur, P.L., C. P. Mangum and C.L. Reiber. 1988. Metabolic compensation in blue crabs, Callinectes sapidus, during long term hypoxia. Biol. Bull. (submitted).

deFur, P.L., D. Nusbaumer and R.J. Lewis. 1988. Physiologicl aspects of molting in blue crabs from the tidal fresh-water Potomac River, Virginia. J. Crust. Biol. 8: 12-19.

deFur, P.L. and B.R. McMahon. 1984. Physiological compensation to short term air exposure in Red Rock crabs, Cancer productus Randall, from littoral and sublittoral habitats.I. Oxygen uptake and transport. Physiol. Zool. 57: 137-150.

Fox, H.M. and M.L. Johnson. 1934. The control of respiratory movements in Crustacea by oxygen and carbon dioxide. J. Exp. Biol. 11:

Graham, R.G., N.B. Terwilliger and C.P. Mangum. 1981. The lactate sensitivity of hemocyanin from the Dungeness crab, Cancer magister. Comp. Biochem. Physiol. 74A 45-50.

Hagerman, L. 1986. Haemocyanin concentration in the shrimp

Crangon crangon (L.) after exposure to moderate hypoxia. Comp. Biochem. Physiol. 85A: 721-724.

Lallier, F., F. Boitel and J.P. Truchot. 1987. The effect of ambient oxygen and temperature on haemolymph L-lactate and urate concentrations in the shore crab Carcinus maenas. Comp. Biochem. Physiol. 86A:255-260.

Lowery, T.A. and L.G. Tate. 1986. Effect of hypoxia on hemolymph lactate and behavior of the blue crab Callinectes sapidus Rathbun in the laboratory and field. Comp. Biochem. Physiol. 85A:689-692.

Mangum, C. P., B.R. McMahon, P.L. deFur and M.J. Wheatly. 1985. Gas exchange, acid-base balance, and the oxygen supply to the tissues during a molt of the blue crab, Callinectes sapidus. J. Crust. Biol. 5: 188-206.

McMahon, B.R., P.J. Butler and E.W. Taylor. 1978. Acid-base changes during recovery from disturbance and during long term hypoxic exposure in the lobster Homarus vulgaris. J. Exp. Zool. 205:361-370.

McMahon, B.R., W.W. Burggren and J.L. Wilkens. 1974. Respiratory responses to long-term hypoxia stress in the crayfish Orconectes virilis. J. Exp. Biol. 60:195-206.

Pease, A.L., P.L. deFur and C. Chase. 1986. Physiological compensation to long term hypoxia in the blue crab, Callinectes sapidus. Am. Zool. 26:122A.

Phillips, J.W., R.J.W. McKinney, F.J.R. Hird and D.L. MacMillan. 1977. Lactic acid formation in crustaceans and the liver function of the mid-gut gland questioned. Comp. Biochem. Physiol. 56B: 427- 433.

Truchot, J.P. 1976. Carbon dioxide combining properties of the blood of the shore crab, Carcinus maenas (L.): carbon dioxide solubility coefficients and carbonic acid dissociation constants. J. Exp. Biol. 64: 45-57.

Wilkes, P.R.H. and B.R.McMahon. 1982. Effect of maintained hypoxic exposure on the crayfish Orconectes rusticus. 1. Ventilatory, acid-base and cardiovascular adjustments. J. Exp. Biol. 98:119-137.

Wilkes, P.R.H. and B.R. McMahon. 1982. Effect of maintained hypoxic exposure on the crayfish O. rusticus: 2. Modulation of haemocyanin oxygen affinity.J.Exp.Biol. 98:139-149.

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CONCURRENT SESSION:

SUBMERGED AQUATIC VEGETATION

Chairs:

Robert J. Orth and Ken Moore
Virginia Institute of Marine Science
College of William and Mary

Contributions by:

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Linda Hurley
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Richard Batiuk
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Nancy Rybicki
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Understanding the Estuary: Advances in Chesapeake Bay Research. Proceedings of a Conference. 29-31 March 1988. Baltimore, Maryland. Chesapeake Research Consortium Publication 129. CBP/TRS 24/88.

Submerged Aquatic Vegetation in the Chesapeake Bay: A Barometer of Bay Health

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INTRODUCTION

In 1978, a program was initiated in the Chesapeake Bay region to investigate the decline of submerged aquatic vegetation (SAV), potential factors that may have led to its decline, its distribution and abundance, and its role and value. The program began with little available background data, but some very basic questions about SAV in the Bay were answered in the approximately three years of research that were funded. For example, it was determined that the decline of SAV was Bay-wide. All SAV species were affected and the decline was unprecedented in the recent history of the Bay. A second important finding was that the decline of SAV was most probably not related to any specific contaminant per se (e.g., herbicide contamination) but appeared to be related to deteriorating water quality in the Bay. Research has demonstrated that SAV species are very sensitive to environmental perturbations, especially those that affect the quantity of light reaching the plant surface.

Managers and citizens have become increasingly aware of the importance of SAV in ensuing years, and citizens have become actively involved in several programs such as the SAV Hunt program, which has provided ground-truth information to the Bay-wide aerial monitoring program. Both Maryland and Virginia have also initiated efforts to restore SAV in currently denuded areas and to develop an understanding of the relationships between SAV survival and environmental quality. These and other projects have yielded significant results that have assisted in Bay management. More research certainly needs to be done. More important, scientists and managers must work together to develop sound strategies for SAV, in concert with an overall Chesapeake Bay policy.

Our attempt in the SAV session at the Baltimore Chesapeake Bay Research Conference was to bring the scientist, manager, and citizen together to discuss recent management needs and research results in four major areas: distribution and abundance, water quality, natural resource value, and restoration. We hope the results of this blend will yield a new perspective on Bay SAV and identify what we must do to manage this resource effectively.

DISTRIBUTION AND ABUNDANCE: A DECADE OF CHANGE

An important component of the early program was an integrated aerial mapping survey of Bay-wide SAV distribution in 1978. This first synoptic aerial view of the Bay has served as a baseline for more recent work.

In attempting to examine historical trends of SAV distribution, it became clear how important comprehensive distributional data are in relating the SAV resource to water quality, climatic factors, or biological changes. Although regular monitoring of SAV distribution was strongly recommended on the basis of the 1978 study, it was 1984 before the next Bay-wide survey was conducted. (SAV was mapped in Virginia in 1980 and 1981; in Maryland in 1979. Maryland has conducted an annual ground survey of SAV since 1972, and the U. S. Geological Survey [USGS] has been monitoring SAV in the Potomac River since 1978.) Subsequent Bay-wide surveys were made in 1985, 1986, and 1987. Studies were also conducted in the Potomac River in 1981 coinciding with the introduction of *Hydrilla verticillata* to the Dyke Marsh area in the tidal freshwater reach of the river. Local citizens have assisted in ground-truthing much of the aerial photography.

The focus of this section of the session was to

Contribution No. 1470 from the Virginia Institute of Marine Science, School of Marine Science, College of William and Mary.

address questions regarding the recent changes in SAV distribution:

- What has happened with SAV in the last decade?
- Has the current SAV distribution information been useful for the manager and scientist?
- What is the best monitoring strategy given the current levels of financial commitment from the federal and state agencies?
- What is the future of *H. verticillata* in the Potomac?

The Bay-wide Status of SAV

The 1978 aerial survey revealed a total of approximately 17,000 hectares of SAV (Figure 1), of which 56% was found in the lower Bay zone (an area from Smith Island to the mouth of the Bay), 27% in the middle Bay zone (Smith Island to the Chesapeake Bay Bridge), and 17% in the upper Bay zone (Bay Bridge to the Susquehanna River). Major areas of SAV abundance documented in this first survey were: Tangier-Smith Island area, Mobjack Bay in Virginia, lower Eastern Shore from Cape Charles to Pocomoke Sound, Eastern Bay area, Choptank River, and Chester River. By 1986, approximately 19,000 hectares of SAV were present in the Bay with 64%, 21%, and 14% found in the lower, middle and upper Bay zones, respectively. Major areas of SAV abundance included not only the same areas as in 1978, but also the tidal freshwater area of the Potomac River and the middle Eastern Shore area, especially around the Barren Island and Honga River in Maryland. Additional increases have been observed in many other sections of the Bay, especially near existing beds of SAV. Spread of SAV has occurred from seed dispersal, which may be one important mechanism not only for bed maintenance but also for revegetation of denuded areas. Reasons for the recent increase of SAV in the mid-sections of the Bay are presently not known. Caution is urged, however, in attempting to relate this modest increase to the recent Bay cleanup efforts. Climatic factors, such as reduced rainfall in the Bay region in recent years, may be one of several important but unknown controlling factors.

The Bay-wide monitoring of SAV has provided valuable information for resource managers including the most up-to-date data on the distribution and abundance of SAV. Products of the annual SAV surveys include (1) photographic imagery, which in addition to documenting SAV occurrence is useful for other activities (land use studies, shoreline erosion studies, etc.); (2) maps based on USGS topographic quadrangles (scale of 1:24,000) delineating all beds of SAV including species information as available from field surveys and ground-truthing; and (3) digitized bed outlines and other accompanying data, which are now stored on computer and can easily be networked into regional or Bay-wide information systems.

For the scientists, the annual survey has provided a synoptic view of the distribution of SAV for the entire Bay in one year. These data serve as an important baseline that will allow the accurate assessment of SAV changes from region to region. Because SAV systems respond to some water quality changes, SAV may be a good indicator to assess the progress of the Bay cleanup. Defining relationships between the water quality conditions and SAV abundance will be very important to Bay managers and regulators who have the ultimate responsibility of insuring the long-term viability of the Bay and its living resources.

The relevance of the monitoring data to the Bay management efforts suggests strongly that this monitoring program should be continued. Scientists are currently conducting the aerial survey annually. This activity represents a modest commitment of funds, which to date have been provided through a cooperative effort of state and federal agencies. Acquisition of adequate funds for an integrated annual survey has been difficult, and when funding is incomplete, significant modifications to the products must be made. Since significant changes can occur rapidly and the natural variability in the system is only beginning to be quantified, ideally the survey should be continued annually, with both aerial photography and digitized computer mapping. A second option would be acquisition of the aerial photographs each year with mapping of all beds and ground-truthing conducted only every second year. SAV abundance might be determined in alternate years by subsampling. If significant changes occurred during the two-year interval, reference could be made to the aerial photographs from the intervening year to determine the timing of the changes. Such an approach may, however, threaten the continuity of the program with repeated mobilization and demobilization of personnel and equipment.

If SAV is to be used as an indicator or "barometer" of Bay health, a commitment must be made at both the state and federal levels to insure that this program continues and is adequately funded. A valuable data base has been developed that has been useful in the development of the Bay cleanup efforts. Every effort should be made to continue the program.

Potomac River: Boom or Bust

The Potomac River provides a case example of a system that has undergone large changes in SAV in the last decade. This region has been known for periods of either abundant SAV, mostly with exotic species, or no SAV at all. Abundant native SAV species were noted in the early 1900's. *Trapa natans*, an exotic, reached

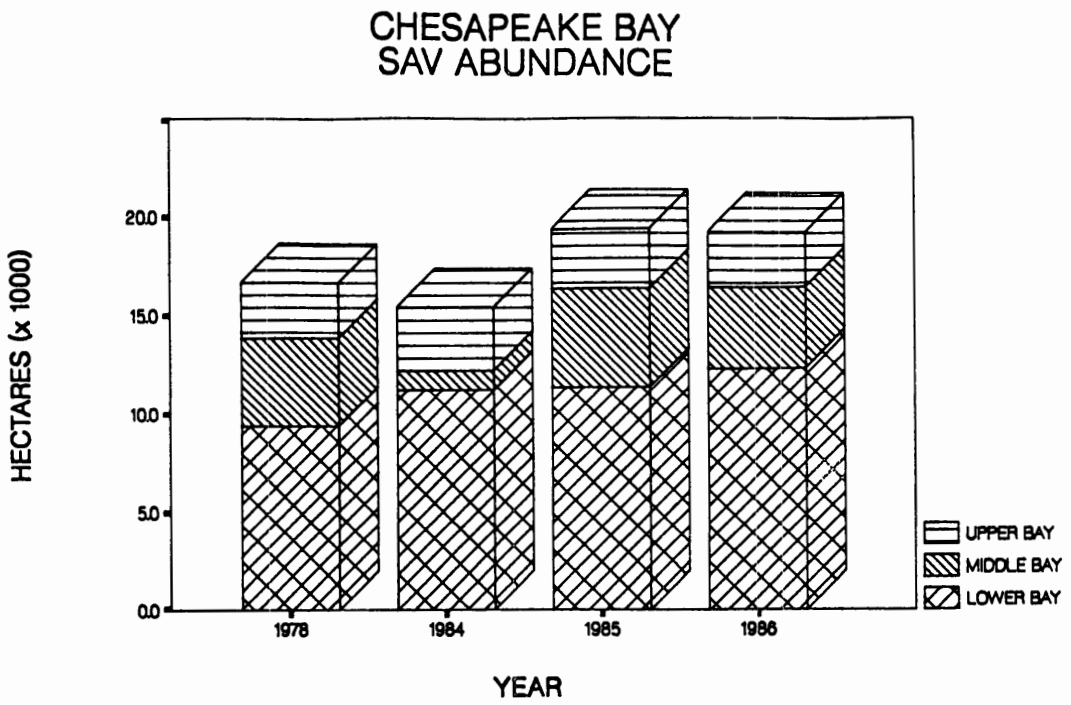

Figure. 1. Abundance of SAV in the three major zones of the Chesapeake Bay for 1978, 1984, 1985, and 1986.

nuisance levels in the tidal portion of the Potomac in the 1920's–1930's and 1950's, and *Myriophyllum spicatum* (another exotic) was very abundant in the transition zone in the late 1950's and early 1960's. Native vegetation in the tidal freshwater and transition zone has been almost absent since the 1920's. *Zostera marina*, a native species that was present in the higher-salinity sections of the river, has been absent since the 1960's. A 1978-1981 survey of the tidal Potomac River and estuary found that SAV was virtually absent from the tidal river and was maintaining only low population levels in the transition zone of the estuary.

Twelve species of SAV were found in the tidal sections of the Potomac in 1983, including *Hydrilla verticillata*, an exotic from southeastern Asia. In 1983 and 1984, there was patchy distribution of all SAV species with *H. verticillata* concentrated mostly on the Virginia side, near Dyke Marsh where it had mistakenly been planted. Between 1984 and 1987, SAV increased from 243 to 1580 hectares and spread downriver. *Hydrilla* has increased from a small percentage of the species in a diverse population of plants, to near domination of the upper tidal river. It became firmly established in Mallows Bay at the upper end of the transition zone. *Hydrilla* could eventually cover all shallow (<2.5 m at mean low water) areas in this section of the Potomac. It also has the potential to become established in similar zones in all the tidal rivers feeding the Chesapeake Bay, although its ultimate distribution will likely be limited by the salinity of the water (approximately 5 o/oo).

The Role of the Citizen

The Bay-wide SAV survey using aerial photography requires considerable ground-truthing to substantiate the presence and species composition of SAV on the photographs. Because ground-truthing of all the SAV beds by trained scientists is impossible, a plan was devised to organize Bay citizens to assist in an "SAV Hunt". This was a cooperative venture between the respective staffs of the U. S. Fish and Wildlife Service, the Chesapeake Bay Foundation, the Alliance for the Chesapeake Bay (formerly the Citizens Program for the Chesapeake Bay), the Maryland Department of Natural Resources, and the Virginia Institute of Marine Science. In addition, members of the Maryland Charterboat Association, funded by Maryland's DNR, also participated in the ground-truthing program. Using citizens to help in the ground-truthing serves three purposes: it provides additional information for the annual aerial survey; it is valuable in educating the citizen about the importance of SAV; and it provides concerned citizens with an opportunity to actually get involved and be a part of the whole "Save the Bay" effort.

Citizens and charterboat captains were asked to go to beds delineated on maps and determine the bed's presence or absence for the current year. Any new beds found were also to be reported. Species information

was collected if species identification could be reliably determined. For the last three years, approximately 150 citizens and 15 charterboat captains have participated annually. Results from the 1986 survey showed that 673 SAV beds were field-checked in Maryland and Virginia. These data have provided valuable information as to the presence of SAV in many areas of the Bay not previously examined. The experience gained from this program has been invaluable for both citizens and scientists, and the program certainly should be continued.

WATER QUALITY: HOW CLEAN MUST THE BAY BE FOR SAV?

There is widespread agreement among scientists, citizens, and managers that improving water quality in the Bay is the number one issue today. As with other aquatic living resources, poor water quality is a major factor affecting SAV growth and production. If it is to remain a viable natural resource, attempts to set nutrient and chlorophyll standards for the Bay must therefore take into consideration the nutrient and light requirements of SAV. This will be no simple task. There are many different SAV species in the Bay with different life history patterns and potentially different growth requirements. SAV grow in rivers whose watershed characteristics are different and where strategies for nutrient control may require different sets of rules.

Important questions that should be addressed related to the SAV living resource are:

- What parameters shall be used in setting goals for water quality criteria or standards?
- Is SAV being considered an important component in the development of the overall water quality criteria?
- Should water quality criteria for SAV be developed for the entire Bay or should there be basin-by-basin criteria?
- What data are needed for setting criteria and how are they being obtained?
- How realistic are these criteria?

Development of Water Quality Criteria: How Critical Is It For The Bay Cleanup?

Since 1983, most of the research and planning efforts for restoring and protecting the Chesapeake Bay have focused on documenting the present water quality of the bay and refining strategies for reducing or stabilizing nutrient and contaminant loads. Strategies based only upon traditional water quality standards, however, cannot necessarily ensure the restoration and protection of living resources.

There is a growing recognition that the Chesapeake Bay must be managed from an ecosystem perspective, requiring innovative approaches to resource and habitat management. The 1987 Chesapeake Bay Agreement states the Program's primary goal is to "provide for the restoration and protection of the living resources, their habitats, and ecological relationships."

Recognition of restoration of living resources as the ultimate goal of the Bay Program has caused a re-examination of how to effectively focus regulatory and management actions to protect or improve habitat quality. Since the early 1970's water quality management has focused on meeting the fishable/swimmable/drinkable goals of the Clean Water Act through the application and enforcement of water quality standards. EPA criteria and state standards are still limited to conventional water pollution parameters (e.g., dissolved oxygen, temperature), and to some toxic metals and organic compounds listed as EPA priority pollutants. The underlying assumptions have been that reducing pollutant loads to meet water quality standards would result in meeting the designated use classifications for certain stream segments. Existing water quality criteria and standards do not well serve the needs of some living aquatic resources and should be reviewed in light of the Bay's overall restoration.

One of the critical limitations of existing state standards is in geographical application. In Maryland and Virginia, use designations within the tidal Chesapeake Bay are geographically defined by the boundary where tidal fresh waters meet oligohaline waters in the tributaries with the mainstem portion of the tributaries. Jurisdictional boundaries between the states at the mouth of the Potomac constitute another artificial barrier to the Bay-wide application of water quality standards.

The Living Resources Task Force, in its September 1987 report on Living Resources Habitat Requirements, suggested that Chesapeake Bay living resources be managed on a regional basis. Regional habitat objectives, based on protecting the combined most sensitive life stages of the representative resources living within that habitat, should be applied on the basis of geographical distribution of living resource habitats. Ideally, only habitat, not political boundaries, would be the determining factor for their application.

Submerged aquatic vegetation has come to play a significant role in the development of regional habitat objectives. For example, it provides the means to bridge the gap between the stated management goal to reduce total nitrogen and phosphorus inputs to the bay by 40% and specific numerical targets for Bay nutrient levels and overall habitat quality. Since eutrophication has been related to SAV decline, SAV can be an important indicator of regional water quality. Lacking in all existing EPA criteria documents and state standards so far are the nutrients, as well as specific

indicators of light transparency, both of which are fundamental to the management of the Bay as an ecosystem. As its restoration to historical abundance has been a key objective of the Chesapeake Bay Program, SAV's utility as an indicator organism is thereby strengthened.

Turbidity, total suspended solids, secchi depth, light intensity, light attenuation, chlorophyll α, dissolved inorganic nitrogen, dissolved inorganic phosphorus, herbicides, sediment type, salinity, pH, temperature, as well as the physical environment (e.g., fetch, waves, etc.) are the types of SAV habitat requirements which could be used in drafting of regional habitat objectives. Laboratory experimentation with field validation would then be necessary to confirm SAV habitat requirements for the above listed parameters. Experimentation should focus on different salinity regimes, representing different species groups' habitats.

Implementation of regional SAV habitat objectives could be the management tool to bridge the existing gap between use and application of existing water quality criteria and standards. In the years to come, linkage of water quality and habitat conditions to changes in living resources would become confirmed through scientific study and monitoring.

Water Quality Criteria

The desire to establish water quality standards based on living resource requirements has focused attention of managers and scientists alike on the necessity for relating potentially important environmental factors to SAV growth and survival. Although research has suggested that various environmental factors can influence production and consequently survival, the actual levels necessary to support growth and survival have only recently been investigated. In the late 1970's researchers at the University of Maryland studied the relationships between SAV survival and eutrophication using 1/8-acre ponds vegetated with native macrophyte species and enriched with fertilizer. Results suggested a direct relationship between nutrient loadings and SAV survival. At the Virginia Institute of Marine Science, investigations with more marine SAV species suggested that SAV in this region may be living close to their levels of environmental tolerance and that, within the physiological constraints of temperature and salinity for the area, reductions in light may be the principal factor controlling SAV growth and survival.

In 1984 a research group at the Virginia Institute of Marine Science began to investigate the relationships between environmental quality and SAV growth and survival in a series of field studies. Objectives were threefold: first, to monitor the environmental quality along an upriver gradient of sites that both currently and formerly supported vegetation; second, to determine the potential for plant production and survival at these sites; and third, on the basis of these two sets of information to determine the levels of environmental variables that characterize the SAV communities in the region. Biweekly sampling of a series of sites in shoal areas along the York River was undertaken. Upriver stations were characterized by complete decline of SAV, while in downriver stations the loss of vegetation decreased with distance to the mainstem of the bay. In addition to the environmental monitoring, eelgrass, the dominant species of macrophyte in the region, was transplanted each fall to determine the potential for SAV growth, production, and survival at the sites.

In 1985 researchers at the University of Maryland, Horn Point Environmental Laboratory, initiated a similar monitoring program in the Choptank River along the upper Bay's Eastern Shore. As in the York River, SAV survived only along the lower section of the river. At nine sites along the tributary, plugs of native species including widgeon grass, redhead grass and sago pondweed were transplanted. The transplants were successful only in the most downriver sites. Since SAV species in this region exibit shoot growth for approximately a six-month period, monthly measures of water quality obtained along a gradient of sites in the Choptank were averaged over this time period to compare stations in the upriver and downriver areas.

Results of both of these studies suggest that there may be similar thresholds for SAV growth in widely divergent areas of the bay. They also suggest that differences between sites that support or do not support growth are quite small and that very small changes in environmental quality can have a significant affect on the vegetation. While these studies are an important step in defining water quality standards many questions remain unanswered. How, for example, do these water quality models fit other river systems in the Bay? What are the interractive relationships between the various factors? What are the seasonal aspects of susceptibility to limiting factors? What role might sediments play in regulating SAV? Other topics important to managers include the impacts of marinas and boating activity on SAV beds. These and other questions need to be investigated, allowing Bay managers to develop effective strategies for restoring living resources in the Bay system.

NATURAL RESOURCE VALUE: A DIFFERENT LOOK

One of the most often-repeated comments made in the last decade about SAV has been that these areas are an important habitat, particularly as a nursery and feeding

ground for many species of invertebrates and vertebrates. SAV beds support much greater densities of macroinvertebrates than adjacent unvegetated areas. Rates of secondary production are extremely high in SAV. The beneficial aspects of SAV have been recently illustrated in the Potomac River. Water clarity has increased substantially in the vicinity of the SAV beds. Positive relationship has been observed between the spread of *Hydrilla* and increased waterfowl utilization as well as increased catch of finfish near these beds.

As pressures continue to grow due to development of the shoreline and watershed of the Bay region, a number of important questions remain:

- Do managers need to know more about SAV functioning to conserve SAV?
- Are all SAV beds considered of equal importance?
- What are the relationships between the role and value of SAV and the size of a SAV bed, the abundance of SAV in an area, or the location of the bed in the estuary?
- Do SAV beds enhance local or Bay-wide productivity?
- Are all SAV beds the same in terms of resource value for individually important species such as the blue crab?

Resource Value—What More Do We Need to Know?

A considerable body of published material describes the resource values of SAV and justifies its conservation. Additional information on biological values is needed, however, particularly concerning which fauna are most dependent on SAV and which SAV species form the most important useful habitat. For example, what are the relationships between SAV in the Bay and waterfowl usage?

One poorly understood relationship is that of SAV bed size and bed function. Are the values of sparsely vegetated beds the same as for a large, densely vegetated area? Are they heavily used by fish and invertebrates, and are they important in habitat expansion? What is the role and value of areas that previously supported SAV but are now unvegetated? Should they be replanted as part of an overall management plan?

SAV beds are utilized by diverse groups of animals. Although their abundances are usually much greater in SAV than in adjacent unvegetated areas, few are exclusively found associated with SAV. For example, some waterfowl species such as canvasbacks and Canada geese, which relied heavily on SAV in the past for food, were able to shift their diets to other sources (e.g., field corn or clams), when SAV declined. Other species, such as redhead ducks, have not shown this flexibility, and their numbers are much reduced in the Bay.

Our understanding of how the large secondary production component fuels other systems, especially species (such as most finfish) that are migratory and not directly dependent on SAV, is very poor. We do not know what proportion of this production remains within the bed and how much may be exported. Because in the past SAV beds occupied a much greater proportion of the Bay bottom, their relative influence compared to today must have been much greater.

The high abundances of fauna in SAV beds have often been related to the refuge from predation they offer. High abundances may also result from enhanced settlement into these habitats. SAV baffles currents and wave action, resulting in deposition of fine sedimentary material; and larvae of invertebrates may act like sediment particles and be selectively deposited in the SAV beds. We have very little information on larval behavior with respect to habitat selection and the settlement process. The high abundances of animals in SAV may first be set by larval supply rates and processes acting on supply rates into a grassbed. Once in a grassbed, larval behavior, vegetation type and density, current speeds, and volume flux all contribute to settlement abundances. Once larvae are established, post-settlement factors affecting survivorship such as predation become very important.

The importance of SAV in blue crab populations of the Chesapeake Bay has been a topic of debate since the large decline of SAV in the 1970's. Blue crab populations have not declined as dramatically as SAV. Blue crab populations are not completely dependent on SAV; states such as Georgia and South Carolina have large blue crab stocks but do not have seagrasses. The Bay region, however, has by far the highest catch of blue crab throughout its entire range, perhaps due to the presence of SAV for several critical life stages.

Juvenile blue crabs are significantly more abundant in SAV beds than in adjacent marsh creeks or bare sand areas. Blue crabs recruit into the Bay as planktonic megalopae (the last stage before the crab assumes primarily a benthic mode), and studies suggest settlement may be much higher in the lower Bay than other sections. Since SAV beds in the lower Bay have declined the least and the lower Bay contains over one-half of all SAV in the Bay, the impacts of SAV loss on the blue crab may not be as large as once thought. Marsh creeks, although of lesser value, may be important nursery sites in areas without SAV. Studies of the relative role of vegetated areas (marshes vs. seagrasses) and the proximity of these areas to larval supply will yield important information on the value on these areas to commercial stocks, not only for the blue crab but also for many other species.

RESTORATION

The loss of SAV in many sections of the Bay prompted scientists and managers to ask whether SAV beds could ever recover naturally, given that whole rivers were completely denuded and were distant from sources of naturally occurring stock. This concern led both Maryland and Virginia to develop restoration programs. Questions central to restoration programs are:

- Was the lack of revegetation due to chronic poor site habitat, poor water quality, or simply a lack of propagules?
- What are the best species to use?
- Are single-species or mixed-species plantings superior?
- What is the best spacing of plants to insure the most rapid recovery of an area?
- How important are patch size and location in improving transplanting success?

Restoration with SAV, although similar to marsh planting, presents a unique problem to the manager and researcher in that all work must be done underwater. Choosing sites for replanting is critical, and success may indeed be related to getting as rapid a spread as possible. In 10 years of pilot transplanting, there have been both successes and failures. During this period we also observed rapid natural recovery of SAV in several areas that provided crucial insights regarding SAV colonization of new areas. These studies suggest that there may be a distinct successional component in these events. Restoration is an important management objective today because of population pressure with its potential for disturbance of SAV habitats. As scientists and managers we must ask what the future of SAV is in the 21st century, given the tremendous projected population growth in the Bay watershed.

Population Growth and SAV—What Can Be Done?

The population in the 64,000-square-mile Chesapeake Bay Basin was estimated to be 13 million in 1980 and is predicted to grow to 16 million people by the year 2,000. Although the current growth rate (1% annually) is anticipated to slow as we enter the 21st century, Bay managers will still be confronted with increasing pressures on dwindling natural resources such as SAV.

SAV is provided indirect protection by point and non-point water pollution control programs, which include sediment and erosion control programs, agricultural best management practices, shore erosion control, and sewerage treatment programs. We need to encourage the expansion and upgrading of these activities and assure that regulatory policies include the conservation of SAV resources. Since regulatory programs provide SAV with direct protection from specific development activities, SAV management has developed into a multiagency responsibility that must be as well coordinated as possible. Regulatory agencies include the U. S. Army Corps of Engineers, the Maryland Department of the Environment, the Maryland Department of Natural Resources, the Virginia State Water Control Board, and the Virginia Marine Resources Commission. In addition, the U.S. Fish and Wildlife Service, Environmental Protection Agency, National Marine Fisheries Service, Virginia Institute of Marine Science, as well as other organizations, provide environmental review on development activities.

In order for these regulatory agencies to adequately protect SAV, they need guidance on methods to minimize the impacts of development activities. Recommendations for minimizing these impacts could include the following:

- No dredging should be permitted between April 15 and October 15 on project sites that currently support or have historically supported SAV;
- Due to differing abundances of SAV between watersheds, one site visit during the growing season (April 15–October 15) should be required on proposed project sites that currently support or have historically supported SAV;
- Watershed-by-watershed protection plans should be developed for the protection of SAV.

In cases where development impacts cannot be avoided, compensation should be given careful consideration. Any type of compensation policy might be based on the premise of no net loss of SAV Bay-wide due to development and associated projects. Strict guidelines should be established for compensation projects, and compensation should be viewed as the last alternative after avoidance and minimization. Compensation/mitigation projects should attempt to be acre for acre, species for species, and habitat value for habitat value. It may also be useful to expand mitigation projects to include those which increase species diversity in already existing beds. For small project impacts, the use of compensation fees should be given consideration. These compensation fees could be used for larger transplant, research, or monitoring projects. Whatever the final management plan, achieving a good consensus will require considerable interaction between the various regulatory and advisory groups.

The use of compensation for ameliorating the adverse impacts of development is based on the premise that transplanting is a viable technology. However, transplanting efforts have met with limited success nationwide and the cost for these projects has been extremely high. Currently, scientists view transplanting as most useful in small-scale projects designed to increase the knowledge of SAV life cycles, transplant-

ing techniques, and water quality parameters and sediment characteristics necessary for healthy SAV growth. Ultimately, small-scale transplanting projects could be used as a gauge for measuring the local effectiveness of Bay clean-up efforts.

Transplanting Programs in the Chesapeake Bay — Progress

The two main goals of SAV restoration programs initiated by Virginia and Maryland over the past few years have been to understand those factors controlling SAV distribution and abundance, and to develop improved methodologies for transplanting in this estuary.

Transplanting SAV can be a difficult undertaking. SAV planting is similar to marsh planting in that (1) whole plants are used in many cases; (2) seasonal timing is important; (3) substrate elevation, sediment type, and salinity are important environmental factors; (4) growth and survival are improved with fertilizer applications to sediments; and (5) plantings are subject to disturbances from physical and biological factors. Major planting differences also exist: (1) SAV are more difficult to harvest, store, transplant, or plant; (2) SAV are subject to additional stress of water quality conditions; and (3) SAV transplants are more difficult to monitor for success and failure.

Transplanting efforts in the Bay have been focused primarily on areas that formerly supported SAV but currently have little or no vegetation. At the Virginia Institute of Marine Science, transplanting has been conducted principally with the seagrass, *Zostera marina*, in the western tributaries of the lower Bay (York, Piankatank, Rappahannock, and Potomac Rivers). At the University of Maryland, Horn Point Environmental Laboratory, transplanting has been undertaken with *Ruppia maritima* and other low-salinity species in the Choptank River. In the Susquehanna Flats and Sassafras River regions, investigators at Harford Community College have utilized *Vallisneria americana* and other freshwater species in their transplanting attempts. Finally, scientists at the USGS have focused their efforts in the upper Potomac River on *V. americana*.

Most transplanting has been done with whole plants, both with and without sediment, because of the availability and ease of collection. The use of tubers and seeds is currently being investigated.

Plantings have varied from small test plots of 1–25 m^2, to larger plots of 900–7200 m^2. Various plant densities and patch sizes within the plots have been tested for their effect on survival. Transplants have been most successful in areas that currently support low abundances of SAV, although regrowth has been generally slow. Regions distant from existing SAV have usually had the poorest success. In some well-monitored experiments differences in success can be associated with differences in water quality. In a number of sites success can be directly related to site exposure. Timing of planting is critical. For example, *Z. marina* planted in the fall is more successful than at other times of the year; *V. americana*, in contrast, does better when planted in the spring. Native stock (for both plants and tubers) have generally yielded greater success than non-native stock.

CONCLUSIONS

Since interest was first focused on SAV in the late 1970's, SAV has come to be recognized not only as a habitat important in its own right, but also to some degree as a model for the entire Bay environment. Requirements for SAV growth, including water that is low in suspended sediments, dissolved nutrients, and phytoplankton, represent what many consider good overall Bay water quality. Because observations worldwide indicate that the health of submerged grass communities can be used as an early indicator of eutrophication, SAV abundance and diversity have been judged to be barometers or indicators of Bay health, and SAV community requirements will be important in the development of regional habitat objectives.

Management of SAV in the Bay may also serve as a model for management of other important Bay resources. It is a management approach that recognizes the importance of setting goals, objectives and plans based on good scientific knowledge, and where the knowledge is lacking, having at hand the mechanism for asking the appropriate questions so that the gaps may be filled. To accomplish this a good relationship has been developed between Bay SAV scientists and Bay managers. This relationship has been fostered by broad public support and to some degree by active participation in SAV programs.

It was the objective of this session of the conference to bring the manager and scientist together to provide not only an update on current research findings, but also a forum for an exchange of understandings. Review of SAV monitoring programs illustrated that refinement is continuing in a program that has had widespread usefulness and is well prepared to participate in the geographic information systems being developed for the Bay region. Yearly monitoring has demonstrated some recent regrowth of SAV in several regions of the Bay and has shown overall levels significantly higher than in 1978, although nowhere near pre-decline levels. Participation by citizens in the monitoring program has been positive and provides important ground-truth in-

formation that in most cases is quite reliable. The development of water quality criteria has been continued, with initial threshold levels established for some species in some areas. The goal of these studies is to assist the managers in setting regional water quality standards based on the requirements of living resources rather than only on traditional water quality management criteria. Investigations further defining the role and value of SAV habitats have been undertaken, along with transplanting studies that assist scientists and managers in understanding the factors limiting natural revegetation. These studies have also provided potential mitigation and compensation tools, the usefulness of which must be further identified and discussed. Regulatory offices must be assisted in setting policies that conserve SAV resources and in developing plans that implement these policies. In total, then, this session recognized that the approach to understanding and managing this important resource is multifaceted and that improvements in the quality of the Bay environment can be obtained in part by managers asking the right questions and scientists providing the correct answers.

REFERENCES

Below are listed many of the major contributions made by Bay scientists toward the understanding of SAV in the Bay. This is not meant to serve as an exhaustive list, but rather to indicate appropriate material from the last decade that may be readily accessible in most libraries. Much of the material has been referred to in this paper. The reader is referred to Stevenson and Confer [1978] for an exhaustive listing of material published before 1978.

Boynton, W. R. Ecological role and value of submerged macrophyte communities: a scientific summary. In: Macalaster, E.; Barker, D.; Kasper, M., eds. Chesapeake Bay Program Technical Studies: A Synthesis. U.S. EPA, NTIS, Springfield, Virginia.; 1982: p. 428-502.

Boynton, W. R.; Kemp, W. M.; Hermann, A. J.; Kahn, J. R.; Schueler, T. R.; Bollinger, S.; Lonergan, S. C.; Stevenson, J. C.; Twilley, R.; Staver, K.; Zucchetto, J. J. An analysis of energetic and economic values associated with the decline of submerged macrophytic communities in Chesapeake Bay. In: Mitsch, W. J.; Bosserman, R. W.; Klopatek, J. M., eds. Energy and ecological modelling. Developments in Environmental Modeling. Elsevier Scientific Pub. Co., Amsterdam.; 1981: p. 441-454.

Boynton, W. R.; Hall, C. A.; Falkowski, P. G.; Keefe, C. W.; Kemp, W. M. Phytoplankton productivity in aquatic plants. In: Lange, O. L.; Nobel, P. S.; Osmond, C. B.; Ziegler, H., eds. Physiological Plant Ecology, IV. Springer. Berlin; 1983: p. 305-327.

Brush, G. S.; Davis, F. W.; Stenger, C. A. Biostratigraphy of Chesapeake Bay and its tributaries: a feasibility study. U.S. EPA Final Report, Grant No. R806680, 68 p.; 1981.

Callender, E.; Carter, V.; Hahl, D. C.; Hitt, E., eds. A water-quality study of the tidal Potomac River and estuary - an overview: U. S. Geological Survey Water-Supply Paper 2233:46 p.; 1984.

Carter, V.; Gammon, P. T.; Bartow, N. Submersed aquatic plants of the tidal Potomac River. U. S. Geol. Sur. Bull. 1543, 58 p.; 1983.

Carter, V.; Bartow, N. The effects of grazers and light penetration on the survival of transplants of *Vallisneria americana* Michx in the tidal Potomac River, Maryland. Aq. Bot. 23:197-213; 1985.

Carter, V.; Rybicki, N. B. Resurgence of submersed aquatic macrophytes in the tidal Potomac River, Maryland, Virginia and the District of Columbia. Est. 9:368-375; 1986.

Carter, V.; Rybicki, N. B. Resurgence of submersed aquatic macrophytes in the tidal Potomac River, Maryland, Virginia, and the District of Columbia. Est. 9:368-373; 1987.

Correll, D. L.; Wu, T. L. Atrazine toxicity to submersed vascular plants in simulated estuarine microcosms. Aq. Bot. 14:151-158; 1982.

Cunningham, J. J.; Kemp, W. M.; Lewis, M. R.; Stevenson, J. C. Temporal responses of the macrophyte, *Potamogeton perfoliatus* L., and its associated autotrophic community to atrazine exposure in estuarine microcosms. Est. 7:519-530; 1984.

Davis, F. W. Historical changes in submerged macrophyte communities of upper Chesapeake Bay. Ecol. 66:981-993; 1985.

Delistraty, D.; Hershner, C. Determination of adenine nucleotide levels in *Zostera marina* (eelgrass). J. Appl. Biochem. 5:404-419; 1984.

Delistraty, D.; Hershner, C. Effects of the herbicide atrazine on adenine nucleotide levels in *Zostera marina* L. (eelgrass). Aq. Bot. 18:353-369; 1984.

Evans, A. S.; Webb, K. L.; Penhale, P. A. Photosynthetic temperature acclimation in two coexisting seagrasses, *Zostera marina* L. and *Ruppia maritima* L. Aq. Bot. 24:185-197; 1986.

Flemer, D. A.; Boynton, W. R.; D'Elia, C. F.; Kemp, W. M.; Nichols, M.; Orth, R. J.; Smollen, J. T.; Taft, J.; Wetzel, R. L. The Chesapeake Bay Program, a summary of scientific research to address management needs for Chesapeake Bay. In: Chao, N. L.; Kirby-Smith, W. eds., Proc. Internatl. Sym. Utilization of Coastal ecosystems: Planning, Pollution and Productivity, Editora da Furg, Rio Grande, Brazil. 1985; p.399-438.

Fredette, T. J.; Diaz, R. J. Life history of *Gammarus mucronatus* say (Amphipoda:Gammaridae) in warm temperate estuarine habitats, York River, Va. J. Crust. Biol. 6:57-78; 1986.

Fredette, T. J.; Diaz, R. J. Secondary production of *Gammarus mucronatus* say (Amphipoda:Gammaridae) in warm temperate estuarine habitats, York River, Va. J. Crust. Biol. 6:729-741; 1986

Goldsborough, W. J.; Kemp, W. M. Light response and acclimation for the submersed macrophyte, *Potamogeton perfoliatus* L.: implications for survival in turbid waters. Ecol. (in press).

Haramis, G. M.; Carter, V. Distribution of submersed aquatic macrophytes in the tidal Potomac River. Aq. Bot. 15:65-79; 1983.

Heck, K. L., Jr.; Orth, R. J. Seagrass habitats: the roles of habitat complexity, competition and predation in structuring associated fish and motile macroinvertebrate assemblages. In Kennedy, V., ed. Estuarine Perspectives, Academic Press, New York. 1980; p. 449-464.

Heck, K. L., Jr.; Orth R. J. Structural components of eelgrass (*Zostera marina*) in the lower Chesapeake Bay - Decapod Crustacea. Est. 3:289-295; 1980.

Heck, K. L., Jr.; Thoman, T. A. The nursery role of seagrass meadows in the upper and lower Chesapeake Bay. Est. 7:70-92; 1984.

Hershner, C. H.; Wetzel, R. L. Submerged and emergent aquatic vegetation of the Chesapeake Bay. In: Majumdar, S.; Hall, L.; Austin, H., eds. Contaminant problems and management of living Chesapeake Bay resources. Pennsylvania Acad. Sci. Typehouse of Easton, Phillipsburg, NJ. 1987; p. 116-133.

Jones, T. W.; Kemp, W. M.; Stevenson, J. C.; Means, J. C. Degradation of atrazine in estuarine water/sediment systems and selected soils. J. Env. Quality 11:632-638; 1982.

Jones, T. W.; Winchell, L. Uptake and photosynthetic inhibition by atrazine and its degradation products on four species of submerged aquatic vegetation. J. Env. Qual. 13:243-247; 1983.

Jones, T. W.; Estes, P. S. Uptake and phytotoxicity of soil-sorbed atrazine for the submerged aquatic plant, *Potamogeton perfoliatus* L. Arch. Environ. Contam. and Tox., 13:237-241; 1983.

Kahn, J. R.; Kemp, W. M. Economic losses associated with the degradation of an ecosystem: the case of submerged aquatic vegetation in Chesapeake Bay. J. Env. Econ. Manag. 12:246-263; 1984.

Kemp, W. M.; Means, J. C.; Jones T. W.; Stevenson, J. C. Herbicides in Chesapeake Bay and their effects on submersed aquatic vegetation. In: Chesapeake Bay Program technical studies: a synthesis. 1981; p. 502-558.

Kemp, W. M.; Boynton, W. R.; Twilley, R. R.; Stevenson, J. C.; Means, J. C. The decline of submerged vascular plants in upper Chesapeake Bay: summary of results concerning possible causes. Mar. Tech. Soc. J. 17:78-89; 1983.

Kemp, W. M.; Boynton, W. R.; Hermann, A. J. A simulation modeling framework for ecological research in complex systems: the case of submerged vegetation in upper Chesapeake Bay. NOAA publication, Marine Ecosystem Modeling, Proceedings from Workshop, p. 131-158; 1983.

Kemp W. M. Seagrass ecosystems as a coastal resource. Mar. Tech. Soc. J. 17:3-5; 1984.

Kemp, W. M.; Boynton, W. R.; Twilley, R. R.; Stevenson, J. C.; Ward, L. G. Influences of submersed vascular plants on ecological processes in upper Chesapeake Bay. In Kennedy, V., ed. The estuary as a filter. Academic Press, N.Y. 1984; p. 367-394.

Kemp, W. M.; Boynton, W.; Cunningham, J.; Stevenson, J. C.; Jones, T.; Means, J. Effects of atrazine and linuron on photosynethesis and growthof the macrophytes, *Potamogeton perfoliatus* L. and *Myriophyllum spicatum* L., in an estuarine environment. Mar. Env. Res. 16:255-280; 1984.

Kemp, W. M.; Lewis, M. R.; Jones, T. W. Comparison of methods for measuring production by the submersed macrophyte, *Potamogeton perfoliatus* L. Limnol. Oceanogr. 31:1322-1334; 1986.

Kemp, W. M.; Murray, L. Oxygen release from roots of the submersed macrophyte, *Potamogeton perfoliatus* L: regulating factors and ecological implications. Aq. Bot. 26:271-283; 1986.

Kollar, S. A. SAV reestablishment results. Upper Chesapeake Bay. In: Coastal zone '85. Proceedings of the Fourth Symposium on Coastal and Ocean Management. p. 759-777; 1985.

Macalister, E. G.; Orth, R. J. Chesapeake Bay's underwater forests. Sea Frontiers 20:115-121; 1984.

Murray, L.; Wetzel, R. L. Oxygen metabolism of the principal autotrophic components of a temperate seagrass community: plant-epiphyte, phytoplankton and benthic algae. Mar. Ecol. Prog. Ser. 38:231-239; 1987.

Olney, J. E.; Boehlert, G. W. Nearshore ichthyoplankton associated with seagrass beds in the lower Chesapeake Bay. Mar. Ecol. Prog Ser. (in press).

Orth, R. J. Submerged aquatic vegetation in the Chesapeake Bay: value, trends and management. In: Groman, H.; Meyers, E.; Burke, D.; Kusler, E., eds. Wetlands Conference on the Chesapeake Bay. Environmental Law Institute. Washington, D.C. 1985; p. 84-95.

Orth, R. J. Grasses beneath the bay. Virginia Wildlife 48:28-31; 1987.

Orth, R. J.; Heck, K. L., Jr. Structural components of eelgrass (*Zostera marina*) meadows in the lower Chesapeake Bay - fishes. Est. 3:278-288; 1980.

Orth, R. J.; Heck, K. L., Jr.; Diaz, R. J. Littoral and intertidal systems in the mid-Atlantic coast of the United States. In: Nienhuis, P.; Mathieson, A., eds. Ecosystems of the World - Intertidal and Littoral Systems of the World. Elsevier Sci. Publ. (in press).

Orth, R. J.; Heck, K. L., Jr.; van Montfrans, J. Faunal communities in seagrass beds: a review of the influence of plant structure and prey characteristics on predator-prey relationships. Est. 7:339-350; 1984.

Orth, R. J.; Heck, K. L., Jr.; Weinstein, M. P. Faunal relationships in seagrass and marsh ecosystems. Est. 7:273-470; 1984.

Orth, R. J.; Moore, K. A. Submerged aquatic vegetation in the Chesapeake Bay: past, present and future. In Proc. 46th North American Wildlife and Natural Resources Conference. Wildlife Management Institute, Washington, D.C. 1981; p. 271-283.

Orth, R. J.; Moore, K. A. The effect of fertilizers on transplanted eelgrass, *Zostera marina* L., in the Chesapeake Bay. In: Webb, F., ed. Proc. Ninth Annual Conference on Wetlands Restoration and Creation. Hillsborough

Community College, Tampa, FL, May 20-21, 1982; p. 104-131.

Orth, R. J.; Moore, K. A. Seed germination and seedling growth of *Zostera marina* L. (eelgrass) in the Chesapeake Bay. Aq. Bot. 15:117-131; 1983.

Orth, R. J.; Moore, K. A. Chesapeake Bay: an unprecedented decline in submerged aquatic vegetation. Sci. 222:51-53; 1983.

Orth, R. J.; Moore, K. A. Submersed vascular plants: techniques for analyzing their distribution and abundance. Mar. Tech. Soc. J. 17:38-52; 1983.

Orth, R. J.; Moore, K. A. Distribution and abundance of submerged aquatic vegetation in Chesapeake Bay: an historical perspective. Est. 7:531-540; 1984.

Orth, R. J.; Moore, K. A. Seasonal and year-to-year fluctuations in the growth of eelgrass (*Zostera marina* L.) in the lower Chesapeake Bay, Virginia, USA. Aq. Bot. 24:335-341; 1986.

Orth, R. J.; Moore, K. A. Distribution of *Zostera marina* L. and *Ruppia maritima* L. s.l. along depth gradients in the lower Chesapeake Bay, USA. Aq. Bot. (in press).

Orth, R. J.; Simons, J.; Allaire, R.; Carter, V.; Hindman, L; Moore, K.; Rybicki, N. Distribution of submerged aquatic vegetation in the Chesapeake Bay and tributaries - 1984. Final report. U.S.E.P.A. 155 p.; 1985.

Orth, R. J.; Simon, J.; Capelli, J.; Carter, V.; Hindman, L.; Hodges, S.; Moore, K.; Rybicki, N. Distribution of submerged aquatic vegetation in the Chesapeake Bay and tributaries - 1985. Final Report. U.S.E.P.A. 305 p.; 1986.

Orth, R.; Simons, J.; Capelli, J.; Carter, V.; Hindman, L.; Hodges, S.; Moore, K.; Rybicki, N. Distribution of submerged aquatic vegetation in the Chesapeake Bay and tributaries and Chincoteague Bay - 1986. Final Report. U.S.E.P.A. 191 p.; 1987.

Orth, R. J.; van Montfrans, J. The role of micrograzing on seagrass periphyton: a review. Aq. Bot. 18:43-69; 1984.

Orth, R. J.; van Montfrans, J. Utilization of a seagrass meadow and tidal marsh creek by blue crabs, *Callinectes sapidus* Rathbun. I. Seasonal and annual variations in abundance with an emphasis on post-settlement juveniles. Mar. Ecol. Prog. Ser. 41:283-294; 1987.

Rizzo, W. M.; Wetzel, R. L. Intertidal and shoal benthic community metabolism in a temperate estuary: studies of spatial and temporal scales of variability. Est. 8:342-351; 1985.

Roberts, M. H., Jr.; Orth, R. J.; Moore, K. A. Growth of *Zostera marina* L. seedlings under laboratory conditions of nutrient enrichment. Aq. Bot. 20:321-328; 1984.

Rozas, L. P.; Odum, W. E. The role of submerged aquatic vegetation in influencing the abundance of nekton on contiguous tidal fresh-water marshes. J. Exp. Mar. Biol. Ecol. 114:289-300; 1987.

Rybicki, N. B.; Carter, V. Effects of sediment depth and sediment type on the survival of *Vallisneria americana* Michx grown from tubers. Aq. Bot. 24:233-240; 1986.

Ryer, C. H. Temporal patterns of blue crab feeding in a lower Chesapeake Bay tidal marsh creek and adjacent seagrass meadow. Est. 10:136-140; 1987.

Ryer, C. H.; Boehlert, G. W. Feeding chronology, daily ration, and the effects of temperature upon gastric evaluation in the pipefish, *Syngnathus fuscus*. Env. Biol. Fish. 9:301-306; 1983.

Ryer, C. H.; Orth, R. J. Feeding ecology of the northern pipefish, *Syngnathus fuscus*, in a lower Chesapeake Bay seagrass community. Est. 10:330-336; 1987.

Shabman, L. A. Benefit taxation for environmental improvement: a case example of Virginia's soft crab industry. Land Econom. 61:398-408; 1986.

Silberhorn, G.; Orth, R. J.; Moore, K. A. Anthesis and seed production in *Zostera marina* L. (eelgrass) from the Chesapeake Bay. Aq. Bot. 15:133-144; 1983.

Stevenson, J. C.; Confer, N. M. Summary of available information on Chesapeake Bay submerged aquatic vegetation. United States Fish and Wildlife Service Biological Services Program FWS/OBS - 78/66, Washington, D. C. U.S.A.; 1978.

Steward, K. K.; Van, T. K; Carter, V.; Pieterse, A. H. *Hydrilla* invades Washington, D. C. and the Potomac. Am. J. Bot. 71:162-163; 1984.

Twilley, R. R.; Kemp, W. M.; Staver, K. W.; Stevenson, J. C.; Boynton, W. R. Nutrient enrichment of estuarine submersed vascular plant communities. I. Algal growth and effects on production of plants and associated communities. Mar. Ecol. Prog. Ser. 23:179-191; 1984.

Twilley, R. R.; Ejdung, G.; Romase, P.; Kemp, W. M. A comparative study of decomposition, oxygen consumption and nutrient release for selected aquatic plants occurring in an estuarine environment. Oikos 47:190-198, 1984.

van Montfrans, J.; Orth, R. J.; Vay, S. Preliminary studies of *Bittium varium* grazing on eelgrass periphyton. Aq. Bot. 14:75-90; 1982.

van Montfrans, J.; Wetzel, R. L.; Orth, R. J. Epiphyte-grazer relationships in seagrass meadows: consequences for seagrass growth and production. Est. 7:289-309; 1984.

Ward, L. G.; Kemp, W.M.; Boynton, W. R. The influence of waves and seagrass communities on suspended sediment dynamics in an estuarine embayment. Mar. Geol. 59:85-103; 1984.

Weinstein, M. P. Population dynamics of an estuarine-dependent fish, the spot, along a tidal creek-seagrass meadow coenocline. Canadian J. Fish. Aquat. Sci. 40:1633-1638; 1983.

Weinstein, M. P.; Brooks, H. A. Comparative ecology of nekton residing in a tidal creek and adjacent seagrass meadow: community composition and structure. Mar. Ecol. Prog. Ser. 12:15-25; 1983.

Wetzel, R. L.; Penhale, P. A. Production ecology of seagrass communities in the lower Chesapeake Bay. Mar. Tech. Soc. J. 17:22-31; 1983.

Wetzel, R. L.; Neckles, H. A. A model of *Zostera marina* L. photosynthesis and growth: simulated effects of selected physical-chemical variables and biological interactions. Aq. Bot. 26:307-324; 1986.

Zieman, J. C.; Orth, R.; Phillips, R.; Thayer, G.; Thorhaug, A. The effects of oil spills on seagrass ecosystems. In Cairns, J.; Buikema, A., eds. Restoration of habitats impacted by oil spills. Butterworth Publ. Mass. 1984; p. 37-64.