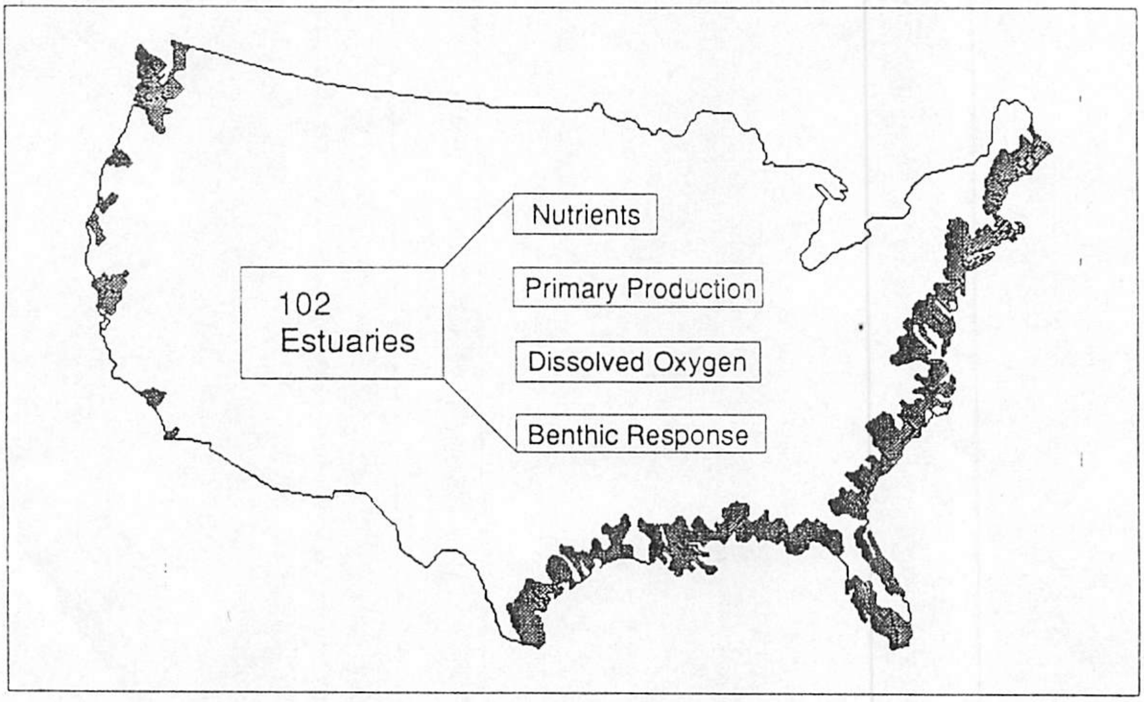
Strategic Assessment of Near Coastal Waters

The National Estuarine Eutrophication Project: Workshop Proceedings



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The National Estuarine Eutrophication Project Workshops assembled a targeted group of experts to specify the potential range of eutrophication effects and associated factors which influence estuarine environments. This report presents 21 abstracts of workshop presentations, and a data table summarizing factors affecting eutrophication which was developed during the workshops. The purpose of this paper is to share these preliminary results with scientists and analysts to elicit additional commentary.

Background

The workshops were conducted as part of NOAA's *National Estuarine Eutrophication Project* to support development of a data-collection survey to describe the state of eutrophication in the Nation's estuaries. The Project will provide for a consistent assembly of existing data for estuarine eutrophication characterization, and will be used to further our understanding of the relationships between nutrient loading and effects.

The Project is part of NOAA's National Estuarine Inventory (NEI) Program (see below) and is being conducted through a cooperative agreement between NOAA's Office of Ocean Resources Conservation and Assessment, Strategic Environmental Assessments Division, and the University of Rhode Island's Graduate School of Oceanography. The Project is supported by EPA's Office of Marine and Estuarine Protection and is being administered through the URI/NOAA Cooperative Marine Education and Research Program.

The goal of the National Estuarine Inventory is to develop a comprehensive framework for evaluating the health and status of the Nation's estuaries and to bring these areas into focus as a resource base of national siginificance. The cornerstone of the NEI is the *National Estuarine Inventory Data Atlas, Volume 1: Physical and Hydrologic Characteristics. Volume 1*, published in 1985, identified 92 of the most important estuaries in the conterminous U.S. and established an operational framework for the organization and characterization of other estuarine attributes in: *NEI Atlas Volume 2, Land Use Characteristics; Volume 3, Coastal Wetlands, New England Region;* and *Volume 4, Public Recreation Facilities in Coastal Areas.*

New estuaries of local or regional importance have been added to the NEI. Additional attributes and improved physical and hydrologic statistics are being developed. A new NOAA report, *Estuaries of the United States: Vital Statistics of a National Resource Base,* updates information in the NEI.

Workshop Objectives

Two Eutrophication Project workshops were held at the University of Rhode Island's Graduate School of Oceanography, the first from January 3-5 and the second from January 24-26, 1991. The objectives of the workshops were: 1) To examine the type and potential extent of problems in U.S. estuaries attributed to eutrophication; 2) To identify factors which may control eutrophic effects resulting from nutrient loadings; 3) To examine hypothesized relationships between nutrient additions and effects in estuaries; and 4) To examine the properties of different estuarine systems to ensure that the data-collection survey would address a broad range of estuaries.

The workshops consisted of prepared presentations by invited speakers and round-table discussions (see workshop abstracts). The intent was to conduct modest-sized workshops to facilitate exchanges related directly to the design of the data-collection survey. Workshops were organized around specific areas of effects. The first addressed effects on primary producers; the second focused on the role and response of the benthic community to nutrient inputs and changes in oxygen conditions.

The presentations were open to the public. Many members of the local and regional scientific community attended portions of the workshop sessions, and contributed valuable insight and commentary.

Results

Prior to each workshop, participants were given a draft of a table listing proposed effects attributed to eutrophication, appropriate measures of each effect, and factors that may potentially influence the expression of the effects. Each participant was asked to review this table and to suggest any additional parameters to be considered. Additions or modifications to the table were based on speaker presentations and discussions (Table 1). The table also lists the relationships to be tested between nutrient loading and effects, and the factors controlling them.

Additional workshop discussions concerned the definition and uses of the terms *eutrophic* and *eutrophication.* Two important points were made. First, it was determined that there are many meanings attributed to the terms in different contexts. It was decided that the eutrophication survey would describe effects and attempt to clarify the relationships between estuarine nutrient loadings (especially anthropogenic loadings) and the resulting effects. Second, it was noted that the study is not designed to determine what level of nutrient additions or effects is acceptable or unacceptable in a given estuary. Eventually, information gained in the study may be important in making such decisions, but it is currently designed to be an objective measure of eutrophic conditions.

Other technical discussions included the possible division of estuaries into sub-areas for problem characterization pertinent to unique conditions of nutrient loading and salinity. The use of benthic indicator species to reflect organic loading, and the use of an appropriate temporal scale for various types of data, were also discussed.

The Next Step

Based on the refined Table of Effects, a draft data-collection survey is being developed to specify the information required. The survey will be tested on a subset of estuaries to ensure its applicability to systems with different physical structures, problem types, and available data. Based on the results, the survey will be refined prior to nationwide application.

Expected Benefits

The survey will provide the first comprehensive assessment of the scale, scope, and characteristics of eutrophication-related problems in the Nation's estuarine areas. It also will help provide a basis for determining the appropriate nationwide response to this class of important environmental quality problems.

The Project will bring together a significant amount of important research in an organized manner, and will allow hypothesized relationships between nutrient loading, modifying factors, and expressed effects to be tested. Sound and effective management of the Nation's estuaries requires a better understanding of estuarine eutrophication then is currently available. This project will lead to significant improvements in our understanding of relationships between nutrient loadings and effects and will, in turn, aid in the formulation of resource management policies and decisions.

For further information on the Eutrophication Project in general, or the survey format, contact:

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Table 1. **Factors influencing the expression of eutrophication effects.**

Effect	How Measured	Factors	Hypothesized Relationships
Phytoplankton Primary Production (gross)	mg C m^{2} Day^{-1} mg C m^{-3} Day^{-1} Total system Annual average Seasonal distribution	Loading rate: N, P, & Si by species Temporal Pattern of loadings Concentration of N or P Light and light penetration Temperature Salinity Stratification Phytoplankton assembledge Bordering Wetlands	$1° = f$(N: moles $area^{-1}$ $time^{-1}$) $1° = f$(N,P: avg. measured moles kg^{-1}) $1° = f$(N,P: ave calculated moles kg^{-1}) $1° = f$(N,P: Light availability)
Net Ecosystem Production	grams C exported and deposited per year	Loading rate: N, P, &Si Water residence time	Net ecosystem C production = 105 x moles P $time^{-1}$
Phytoplankton Abundance	Chl-a m^{-3} Chl-a m^{-2} total cell counts g C m^{3} g C m^{2}	Loading rate: N, P, & Si by species Temporal Pattern of loadings Concentration of N or P Seasonal variability in loading rate Depth Stratification Grazing Turbidity Water residence time Temporal distribution of vertical mixing energy Abundance of grazers	$[C] = f$(specific growth rate, population size, grazing, export, settling)
Phytoplankton assemblage (Diatom vs flagellate) (size) (beneficial species)	Ratio of total diatoms to total flagellates Average size	N:Si ratio in sewage effluent P:Si ratio in sewage effluent Succcession Bad water (modification)	Flagellates become dominant in high nutrient loadings Shift to smaller species in high loadings Posioning of water for "good" species

Table 1 continued. Factors influencing the expression of eutrophication effects.

Effect	How Measured	Factors	Hypothesized Relationships
Appearance of nuisance (and toxic) algal blooms	Interval between recorded blooms Areal extent of blooms Duration of Blooms Record of first appearance	Horizontally distinct water mass A vertically stratified water column Warm weather conditions High photosynthetically active radiation Enhanced allochthonous organic matter loading Enhanced allochthonous inorganic nutrient loading Adequate supply of essential metals (e.g. iron, selenium) Sediment "seed beds" Algal-bacterial synergism Algal-micrograzer synergism Selective activities of macrograzers Rate of change of nutrient or other contaminant loadings Vectors for invasion Virus-phytoplankton interactions Rain events Climate changes	"A combination of the factors will most likely lead to bloom sensitive waters"
Abundance of submerged aquatic vegetation (decrease)	Areal extent of coverage Density in covered areas	Nutrient concentration Turbidity, light color Phytoplankton abundance Epiphyte abundance Bottom substrate, Sedimentation Depth, pigment type Water flow rates, mixing Mechanical disturbance Wave action Seaweed abundance Climate changes Toxins Benthic environment changes	Decrease in Abundance is a result of phytoplankton lowering nutrient and light availability $1^\circ = f(N;P,\text{Light, water flow})$

Table 1 continued. Factors influencing the expression of eutrophication effects.

Effect	How Measured	Factors	Hypothesized Relationships
Seaweeds	Areal extent of coverage Density in covered areas Diversity	Nutrient concentration Turbidity, light color Phytoplankton abundance Epiphyte abundance Bottom substrate, Sedimentation Depth, pigment type Water flow rates Mechanical disturbance Wave action Grazers	
Floating Algal Scums	Visual reports areal extent Duration Bloom interval	Quisecent waters	
Bottom water oxygen concentrations	µmoles O_2 L^{-1} Volume of low O_2 waters Area of Low O_2 coverage Length of time of low O_2 concentrations Fish Kills	Bottom water residence time Depth Stratification Rate of organic matter decay Temperature Organic loadings DIN and DIP loadings Particle settling velocity Rate and timing of primary production Statistics of isolation events Rate of oxygen decrease	$[O_2]=f$ (organic carbon input rate, organic carbon decay rate, re-aeration rate)
Secondary production (commercial harvests)	Tons of yield y^{-1} Single species Total harvest	Primary production Fraction of areas open for harvest Trophic transfer efficiency Overfishing maintaining low population stocks	yield = f (food chain length)

Table 1-continued. Factors influencing the expression of eutrophication effects.

Effect	How Measured	Factors	Hypothesized Relationships
Abundance of benthic organisms (increase)	No. macrofauna m^{-2} Gr. macrofauna m^{-2} Bioturbation	Primary production Depth Organic carbon input Grazing rates (top down control)	Abundance is equal to f (carbon supply)
Abundance of benthic organisms (decrease)	No. macrofauna m^{-2} Gr. macrofauna m^{-2} Bioturbation	Oxygen concentrations Sediment quality Grazing rates (top down control)	Abundance is equal to f (oxygen concentration)
Benthic species composition	Diversity indicies Presence of "desirable" species Bioturbation	Primary production Oxygen concentrations Food Quality Sediment quality Organic carbon input Salinity variability	Composition is equal to f (carbon supply)
Mephitic Waters	Records of foul odor	Low oxygen concentrations Algal species Algal density	
Turbidity	Light extinction coefficient mg suspen. matter l^{-1}	Inorganic sediment inputs, runoff Sediment resuspension Storm events Phytoplankton abundance DOC concentration and adsorptivity	
Bacterial activity	Glucose or amino acid metabolism Thymidine uptake	Increased populations responding to increased organic matter	
Disease organisms (for marine populations)	Reports of diseased organisms	Altered conditions promoting survival of pathogens Altered conditions reducing resistance of organisms	

Abstracts

THE STRATIGRAPHIC HISTORY OF EUTROPHICATION AND ANOXIA IN CHESAPEAKE BAY: HUMAN AND CLIMATE FACTORS

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Vertical profiles of pollen and seeds of terrestrial and aquatic plants, diatoms, metals, carbon, nitrogen, sulfur, and charcoal from sediment cores collected in tributaries of Chesapeake Bay provide a history of change in water quality and biological populations related both to climatic change and human activity. The record covers a period of 4,000 years and includes the Medieval Warm Period (1000 to 1200 A.D.), the Little Ice Age (1200 to 1600 A.D.) and European settlement (1630 to the present). Since European settlement, the upper Chesapeake Bay has been transformed from a predominantly benthic system to one that is predominantly planktonic. This change is coincident with increased sedimentation accompanying deforestation and increased nutrients associated with agriculture and sewage effluent. Profiles of sulfur, organic carbon, and organic nitrogen in the middle upper Chesapeake Bay indicate increased anoxia since European settlement. The degree of pyritization of iron, an indicator of oxygenation of bottom waters, shows decreased oxygen during the past 40 years. Similar but less pronounced changes are recorded during the Medieval Warm Period, believed (from pollen, seeds, and charcoal preserved in the sediment) to be a period of extensive fire interspersed by heavy rainfall. Profiles of submerged aquatic vegetation, based on fossil seeds, show a non-random pattern of change from one community to another prior to European settlement, and random after settlement. The inference is that pre-European populations, responding to disturbances resulting from periodic climatic events, are characterized by a certain degree of predictability. Post-European communities, responding to a non-periodic, escalating disturbances created by massive deforestation, erosion, and siltation of the upper estuary, result in unpredictable species shifts.

SPATIAL AND TEMPORAL VARIABILITY OF PHYTOPLANKTON BIOMASS AND PRODUCTIVITY IN ESTUARIES

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A long-term observational program in San Francisco Bay demonstrates the scales and magnitudes of phytoplankton biomass variability that can exist in estuaries. For example, phytoplankton biomass can vary by a factor of five to ten over length scales of 10 km, both in the lateral dimension (e.g., across the bathymetric transitions between deep channels and the adjacent subtidal shoals) and in the longitudinal dimension (along the river-ocean continuum). Spatial distributions change over time, at periods ranging from hours to years. Short-term variability is largely a function of tidal advection, whereas variability at the longer time scales is often associated with climatic or hydrologic variability (e.g., seasonal/interannual fluctuations in river flow).

Some of the spatial-temporal variability of estuarine phytoplankton is driven by processes unrelated to nutrient delivery. The following generalizations, emerging from long-term study of San Francisco Bay, may apply to other coastal-plain estuaries: (1) Estuaries are often characterized by high concentrations of suspended sediments and therefore have shallow euphotic zones, such that phytoplankton primary productivity is limited by the availability of light energy to drive photosynthesis; (2) Top-down controls can play a major role in regulating phytoplankton dynamics. For example, annual primary production in upper San Fransisco Bay varies from about 20 g C m^{-2} to over 150 g C m^{-2} and is lowest during years when benthic suspension feeders are most abundant; (3) Physical processes, such as turbulent mixing, also play important roles in regulating phytoplankton populations. For example, in South San Fransisco Bay phytoplankton blooms occur only during episodes of persistent weak tides when density stratification is strong and vertical mixing is slow.

These generalizations have implications for any program that attempts to characterize the effects of variable nutrient inputs on lower trophic levels of estuaries: (1) Long-term measurement programs should be based on an understanding of the spatial and temporal variability at all relevant scales; (2) Individual estuaries may respond uniquely to altered nutrient loadings because of differences in the degree of light limitation, the balance between primary production and consumption by grazers, and differences in the physical regime among estuaries.

THE "BROWN TIDE" AND OTHER NOVEL ALGAL BLOOMS IN NEW YORK/NEW JERSEY COASTAL WATERS

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Several coastal embayments along the northeast coast of the United States have recently experienced novel microalgal blooms for which there is no previous record. These monospecific blooms were popularly called the "brown tide" due to the resulting water color. In the early summer of 1985 the first appearance of the "brown tide" occurred over a wide geographic range along the coast in non-contiguous bodies of water: Narragansett Bay, RI and Long Island embayments in New York, as well as Barnegat Bay, NJ. (Cosper et al., 1987; Sieburth et al., 1988; Nuzzi and Waters, 1989; Olsen, 1989; Sieburth and Johnson, 1989; Smayda and Villareal, 1989). The extent of the blooms was restricted to these coastal bay systems; blooms did not appear to follow a pattern of spreading from one bay system to the next. This suggests that the environmental factors contributing to these "brown tide" blooms were not just localized to specific conditions in a bay system, but probably were more regional, (e.g., involving meteorologically induced changes). The blooms on Long Island markedly reduced the extent of eelgrass (*Zostera marina*) beds because of increased light attenuation, and decimated populations of commercially valuable bay scallops (*Argopecten irradians irradians*), since the scallops were unable to graze adequately and starved to death (Cosper et al., 1987; Bricelj and Kuenstner, 1989; Dennison et al., 1989). Similarly, in Narragansett Bay the mussels were unable to feed and populations were severely reduced (Tracey et al.,1989).

In 1986 the blooms recurred throughout the summer months in the same Long Island embayments as previously. In Long Island and in Barnegat Bay during the summers of 1987, 1988, and 1989 the "brown tide" blooms returned only in diminishing levels (Cosper et al., 1989a, 1990; Nuzzi and Waters, 1989; Olsen, 1989). During the summer of 1990 very brief outbreaks of the bloom were recorded in certain Long Island bay areas by the monitoring program of Suffolk County (Nuzzi, 1990). Since 1985 "brown tide" blooms have not returned to Narragansett Bay (Sieburth et al., 1988; Smayda and Villareal, 1989).

The "brown tide" species was dominant in terms of cell number, and contributed greater than 80 percent of total cellular phytoplankton volume throughout most of the bloom period during the summer months (Cosper et al., 1987, 1989a). During the blooms, phytoplankton biomass, as indicated by chlorophyll-a levels, was not particularly elevated for Long Island bays, in comparison to other years, since concentrations reached levels less than 30 $\mu g\ l^{-1}$, but chlorophyll-a was concentrated in the smaller (less than five µM) fraction (Bruno et al., 1983; Lively et al., 1983; Cosper et al., 1987, 1989a). Primary productivity levels were high but also were not different from pre-bloom years. The less than 10 µM fraction of the phytoplankton contributed greater than 90 percent of the total photosynthetic activity throughout the bloom period; estimates of picoplankton carbon turnover were rapid, on the order of hours (Cosper et al., 1989a). Changes in inorganic nutrient levels, such as nitrate, nitrite, phosphate and ammonium, were not different from pre-bloom years (Hardy, 1976; Bruno et al., 1983; Lively et al., 1983). Variations in inorganic macronutrients were not correlated with variations in the productivity of the "brown tide" and there is no evidence to support increased macronutrient loading as a cause of the blooms (Cosper et al., 1989a, 1989b). These findings are consistent with similar studies in Narragansett Bay, RI (Smayda and Villareal, 1989; Keller and Rice, 1989).

Since the "brown tide" alga, *Aureococcus anophagefferens* (Sieburth et al., 1988), is a new

Table 1. Number of novel mass blooms by water mass type in New York and New Jersey between1950 and 1988 (New Jersey share is in upper left corner, New York share is in upper right).

Water Mass	1950s	1960s	1970s	1980s	Total
Bay/ Estuary	7 6 13	28 9 37	17 0 17	4 8 12	56 23 79
Coastal	0 1 1	1 1 2	3 0 3	4 0 4	8 2 10
New York Bight	0	0	1	0	1
Total	7 7 14	29 10 39	21 0 21	8 8 16	65 25 90

species not previously known to cause blooms, environmental conditions contributory to the blooming could in part relate to new anthropogenic influences in these bays. Physiological studies have identified certain micronutrients as conducive to the growth of this species, such as specific chelators (which have replaced phosphates in detergents), selenium and iron, and organic phosphates and nitrogen sources (Cosper et al., 1990). Drought conditions, which elevated salinities to a level of 30 ppt. conducive to the growth of *A. anophagefferens*, along with pulses of rain which delivered specific nutrients to the bay waters and the restricted flushing of bay waters, set the scenario for the formation of the bloom (Cosper et al., 1990). Selective grazing pressures against the "brown tide" species during the early bloom phase could have further allowed for the development of large populations (Caron et al., 1989). Since this species is still present in Long Island bays, it could potentially bloom again.

The "brown tide" bloom scenario has some similarity to the "green tide" blooms of the 1950s in Great South Bay, which also affected the bivalve oyster populations (Ryther 1954, 1989). During the early '50s, a lowering of salinity selected for two estuarine species, *Nannochloris spp.* and *Stichococcus spp.*, with a salinity of 17 ppt. optimal for growth. The recurrence of the "green tides" for several summers afterwards appeared to depend on the restricted circulation of the inshore bays and the overwintering of large enough seed populations to initiate the next summer's growth. Effluents from duck farms, which flowed into Great South Bay through creeks, were found to be supplying nitrogeneous nutrients and promoting the growth of these two species of microalgae, and these effluents were subsequently restricted.

These unusual bloom events prompted an evaluation of historical accounts of novel algal blooms for the New York/New Jersey coastal waters as well as a consideration of contributory environmental factors (Olha, 1990). The record is relatively incomplete until the 1950s. Chlorophyte blooms of "small forms" dominated in the 1950s and 1960s. In New York waters, the 1970s were characterized by a lack of blooms except for the dramatic *Ceratium tripos* bloom in shelf waters in 1976. "Small form" blooms returned in the 1980s as the "brown tide," a new picoplankton species. Dinoflagellate blooms occurred in all four decades. There is no evidence of an increase in novel blooms in the last 40 years. However, the blooms were much more likely to occur in estuarine areas than in coastal or shelf waters (Table 1). Multiple environmental variables appear to contribute to any single novel bloom and it is impossible to attribute the blooms to any unique set of variables (Table 2).

Table 2. Number of times per decade that a particular environmental variable was considered partially responsible for a novel bloom occurence.

Factor	location	1950s	1960s	1970s	1980s	Total
winds	NY	5	8	1	0	21
	NJ	3	0	1	3	
salinity	NY	5	1	0	4	19
	NJ	3	2	0	4	
rainfall	NY	1	2	0	4	16
	NJ	3	2	0	4	
runoff	NY	4	2	1	4	22
	NJ	3	1	1	6	
nutrients	NY	5	0	0	6	28
	NJ	3	6	0	8	
residence time	NY	5	8	0	6	29
	NJ	3	6	0	1	
stratification	NY	0	0	1	2	8
	NJ	1	1	1	2	
grazing	NY	0	0	1	2	15
	NJ	3	6	1	2	
temperature	NY	1	1	1	2	13
	NJ	3	1	1	3	
oceanic transport	NY	1	0	1	1	7
	NJ	0	1	1	2	
Total		52	48	12	66	178

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MODELING EUTROPHICATION AND DISSOLVED OXYGEN IN CHESAPEAKE BAY AND LONG ISLAND SOUND: INSIGHTS AND PROBLEMS

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This paper will present a review of the state of the art of deterministic mass balance modeling for eutrophication and dissolved oxygen in estuaries. As an introduction to this discussion, the use of Vollenweider-type loading models, and the precision to be expected from their application, will be examined.

The model that has been developed for Long Island Sound, LIS II, will be discussed in detail. The kinetic formulation will be presented that represents phytoplankton growth, respiration, and mortality; the nutrient cycles and mineralization; and dissolved oxygen production and consumption. A more detailed presentation of the sediment flux model will be included since this is a recent innovation that significantly increases the extent to which a complete mass balance of nutrients can be implemented. The sediment flux model receives particulate organic matter flux from the overlying water, mineralizes it, and computes the fluxes of ammonia, nitrate, oxygen, phosphorus, and silica via interstitial water-overlying water mass transfer. The model will be examined using both Chesapeake Bay and Long Island Sound sediment flux data.

The calibration of LIS II will be examined in some detail. The purpose is to highlight the variables required for the construction of these models. These variables should be included in the questionnaire so that simplified forms of these models can be applied to the NOAA estuaries.

Finally, the success of these models in predicting the results of nutrient control will be discussed within the context of Lake Erie. Significant reductions of phosphorus loading to Lake Erie have occurred since the 1970s. The response of the lake and the resulting changes in nutrients, chlorophyll, and dissolved oxygen will be compared to predictions made for the 1978 Water Quality Agreement between the U.S. and Canada.

NUTRIENT CONTROL & FISH STOCKING: LAKE ONTARIO STRATEGIES IN CONFLICT?

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Eutrophication in the Great Lakes is influenced by the synergistic actions of many different factors, none of which act independently with similar results. For example, Lake Ontario is presently experiencing perturbations through nutrient fluxes as well as introductions of fish and zooplankton that potentially represent large-scale changes in function of this Great Lake ecosystem. Data covering various system components are presented here to highlight some of the long-term trends in Lake Ontario and to investigate some potential causative factors for changes observed over time.

Water quality changes have been found to be occurring in Lake Ontario at rates and in ways that are different from what would be expected. In examining the long-term trends of system variables for which we possess data, some interesting patterns emerge. Phosphorus loading has decreased substantially since the early 1970s. In contrast, surface, midlake nitrate, and nitrite have exhibited an increasing linear trend of 9.18 $\mu g l^{-1} yr^{-1}$ as determined for the period 1968 to 1986.

Biological components have also shown interesting patterns. For example, phytoplankton biomass has exhibited a decline over the early 1970s. Top predator biomass, on the other hand, has shown a major increase in the Lake over the last decade, as reflected by increased stocking rates. It is relatively safe to assume that this increased predator biomass has most certainly impacted its prey species (e.g., alewife and smelt), which in turn has relaxed predatory pressure of these forage species on zooplankton. Again this pattern is suggested with the increased zooplankton number of the 1980s compared to those numbers for the early 1970s. In summary, decreases in phosphorus and phytoplankton, although corresponding to decreased phosphorus loading to the system, also appeared to mirror the increases in fish biomass and its effects on trophic structure.

In Lake Ontario, decreased phosphorus and reversal in eutrophication have been the result of an intensive abatement program. By this strategy *decreased system productivity* has occurred, as measured by improvements in water clarity, decreases in phosphorus and chlorophyll-a, and changes in phytoplankton community composition. Since the late 1970s millions of salmon and trout fry have been jointly stocked in Lake Ontario by Provincial, State, and Federal governments in Canada and the United States. The purpose of this stocking strategy has been to *increase system productivity* as reflected in the recreational harvest of large predatory fish. Is there a conflict in Great Lake resource management strategies that on one hand target nutrient control, limiting excessive productivity (one symptom of eutrophication), while on the other hand emphasizing extensive fish stocking, enhancing lake productivity (as reflected by higher trophic levels)?

Integrating several years of data on Lake Ontario, the presentation attempts to investigate the premise that current nutrient inputs may *not* be sufficient to support the food web. Using data for 1972, 1982, and 1987 (Table 1) provides some ability to evaluate the question of adequate food web support for fisheries production. In 1972, 51 percent of the phosphorus required to support phytoplankton production was derived from external loadings, and there was more than enough phytoplankton carbon produced to support estimated fishery production. In 1982, 46 percent of required phosphorus was derived from external loading, and again there appeared to be enough phytoplankton carbon production to support estimated fishery production in the Lake. The 1987 calculations, however, gave the first indication that there may be more fish production in Lake Ontario than long-term nutrient loading and phytoplankton production can support. Loading appeared to provide 51 percent of required phosphorus for phytoplankton while the fish production estimated for 1987 required 32 g $Cm^{-2}y^{-1}$ more than what was apparently produced by primary producers in the system.

Empirical models from other lake systems are used in this presentation to further corroborate the conclusions drawn from these evaluations (Figure 1). Based on the calculations presented here, it is suggested that Lake Ontario's present production dynamics are stressed by lack of sufficient nutrients available to sustain the primary production required to support observed yields of fish from the system. This presentation goes on to demonstrate, through examination of the nutrient recycle scheme for Lake Ontario, that recycling was also affected by nutrient abatement strategies and in 1987 contributions from all sources of phosphorus (external and recycled) appeared to fall short of requirements by phytoplankton to support estimated fish production in the Lake.

There are no simple distinctions between nutrient-loading dynamics to a lake, such as Lake Ontario, and the cascading trophic interactions that occur and also control productivity. Potential production at all trophic levels is set by nutrient supply. Actual production, however, depends on the recycling of nutrients and their partitioning among various trophic levels that vary in growth rates. Whereas changes in external nutrient loadings for Lake Ontario may affect composition and productivity of phytoplankton communities, examination of nutrient cycling from a biological rather than physiochemical perspective exclusively, suggests that there are also controlling factors within lakes that are generated by higher trophic levels.

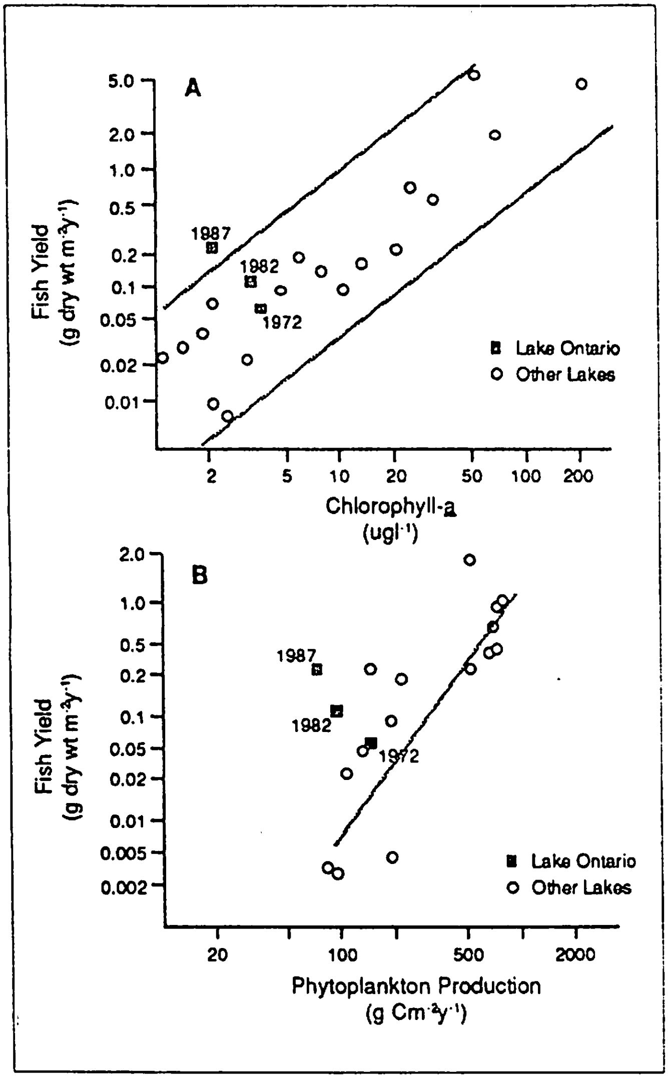
Implication for interconnections between nutrient cycles and food webs is not new and should be obvious. It is possible that because something is so

Table 1. **Results of simple simulation model comparing nutrient loading to Lake Ontario, phytoplankton production (PPr) in the deepwater environment, and fisheries production as reflected by fish harvest.**

YEAR	Estimated Phosphorus Loading ($g\ p\ m^{-2}y^{-1}$)	Total P Required to Support PPr ($g\ p\ m^{-2}y^{-1}$)	Mean Annual Phytoplankton Production ($g\ p\ m^{-2}y^{-1}$)	Total Fisheries Production ($g\ p\ m^{-2}y^{-1}$)	PPr Required to Support Observed Fish Production*
1972	0.86	1.68	178.0	0.04	40
1982	0.43	0.94	90.0	0.06	60
1987	0.38	0.74	78.0	0.11	110

* Assume a three trophic level transfer with each trophic level characterized by a ten percent transfer efficiency.

Figure 1. Comparison of Lake Ontario data for chlorophyll-a and fish yield (A), and phytoplankton production and fish yield (B) with empirical models developed by Oglesby (1977).



obvious it is overlooked. A paradox arises when one considers nutrient-loading control in the Great Lakes simultaneously with massive fish stocking. Decreasing nutrient loadings beyond some presently unknown threshold, for example, without knowing nutrient contributions from other sources to the system (e.g., recycling), may negatively impact a food web which is being heavily exploited by increases in top-level predatory fish. From this investigation, it is concluded that a clear understanding of Great Lake nutrient cycle/food web links must be holistic, incorporating trophic dynamics and nutrient dynamics in a single integrated framework, while acknowledging that one process cannot be investigated in isolation from the other.

BENTHIC COMMUNITIES AS INDICATORS OF ESTUARINE EUTROPHICATION

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The tenant that benthic communities are sensitive indicators of environmental disturbance caused by anthropogenic activities is generally accepted. These communities, which generally have longer life spans and are less mobile than pelagic fauna, integrate environmental conditions over periods of weeks to months and in some cases, years. Benthos are intimately associated with sediments, the environmental sink for many contaminants and the depository for many organics. Thus they are able to respond to changes in pollutant loadings, in some cases quite rapidly.

Studies of benthic community responses to organic enrichment resulting from eutrophication (nutrient stimulated in situ production) are rare. The commonly referenced studies of benthic responses to organic enrichment focus on enrichment from allochthonous carbon sources, such as paper pulp effluent and sewage treatment plants. Chief among these is the review of Pearson and Rosenberg (1978) who put together the general model for changes in benthic community parameters associated with a gradient of organic enrichment including number of species, abundance, and biomass.

Pearson and Rosenberg also identified certain benthic species associated with organically enriched areas. These species tend to be short-lived, rapid growing surface-dwelling and feeding forms, frequently referred to by benthic ecologists as opportunists (Grassle and Grassle,1974). Commonly found representatives of these species include *Capitella capitata, Polydora ligni, Streblospio benedicti,* and *Mulinia lateralis.*

Although the Pearson and Rosenberg model has been useful to understand benthic community changes along space or time gradients, the model has not been applied to assess the eutrophication status of estuaries. The problem is that the model is based on benthic responses to large external carbon sources that result in extreme changes in environmental conditions (anoxia, hypoxia) and benthic communities. Available data suggest benthic responses to nutrient-induced eutrophication are probably more subtle, resulting in increased secondary productivity and biomass of existing assemblages but rarely in major shifts in community composition or diversity. A further problem is that changes in benthic community parameters, including, abundance, biomass, and number of species, also occur with other forms of environmental stress including contamination as well as physical environmental conditions including salinity and sediment characteristics. Biological interactions (i.e., predation) may also affect these community parameters.

For the reasons above, classification of the eutrophication status of estuaries using benthic community data will not be easy. If used individually, benthic abundance, biomass, and number of species are unlikely to be good indicators of eutrophication. If *used together*, perhaps as a numerical index, these parameters may better identify eutrophic areas over broad scales. In addition, scientific knowledge about what should be expected, based on information from less eutrophic situations and the ecology of the dominant benthic species, will probably be helpful in identifying eutrophication status. For example, the proportion of opportunistic species or suspension feeding groups in benthic communities may increase as the degree of eutrophication increases. Preliminary analysis of data collected during a controlled mesocosm eutrophication experiment suggests that benthic communities in eutrophic areas have more widely fluctuating seasonal cycles. Data collected from estuaries in the National Estuarine Inventory might be used to test this suggestion. Regardless of which benthic parameters are most associated with eutrophic areas, ancillary data (e.g., sediment contaminants, sediment toxicity, bottom dissolved oxygen) will be needed before the effects of eutrophication can be partitioned from those due to other factors.

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NUTRIENT LOADING, PRIMARY PRODUCTION, AND EUTROPHICATION OF CHESAPEAKE BAY

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Since its inception in 1984, the Maryland Chesapeake Bay Monitoring Program has sought to provide a synoptic picture of the causes and consequences of nutrient enrichment of Chesapeake Bay. Component studies of this ongoing program have compiled annual records of nutrient loading, water quality, primary productivity, phytoplankton and zooplankton community structure, organic matter deposition rates, and benthic flux rates. Our purpose here is to present an overview of the progress we have made in synthesizing these data in the form of nutrient budgets, and to present evidence of a relationship between rates of total nitrogen (TN) loading and annually-averaged, depth-integrated chlorophyll-a (AADIC). We propose this relationship forms the basis for an empirical Vollenweider-type model of estuarine response to nutrient enrichment. The relationship between TN loading and AADIC emerged from comparisons of sections of the bay (those for which we have similar data sets) that were treated as essentially independent estuarine systems. These sub-systems of the Chesapeake are the Northern Mainstem Bay, Potomac River, Patuxent River, Patapsco River, and Choptank River. Our workshop presentation was structured in four sections covering: (1) spatial and temporal patterns in nutrient loadings; (2) spatial and temporal patterns in phytoplankton primary production and biomass; (3) a search for relationships between nitrogen loading rates and chlorophyll-a biomass; and (4) thoughts on the appropriate use and definition of "eutrophication" in the estuarine context.

Nutrient loadings and budgets. We have been developing budgets for several portions of Chesapeake Bay (Boynton et al.,1989) that summarize, on an annual basis, our ability to quantify nutrient inputs, losses (sediment burial, denitrification, fisheries exports), storages (in water, biomass, and sediments), and recycling rates. These budgets indicate that annual area-based loadings of total nitrogen (TN) and total phosphorus (TP) differed by factors of seven and 24 respectively, for the five sub-systems of Chesapeake Bay under consideration. From year to year, rates of nutrient loading of any one system typically varied by a factor of about two. The comparison among systems, using each year of data as a replicate for that system, therefore doubled the "dynamic range" of loadings for the suite of systems. This suggested that patterns linking loading and ecosystem-level responses could be detected against the background of interannual variability for an individual estuarine system by comparisons between systems. In other words, we used a "systems comparison" approach.

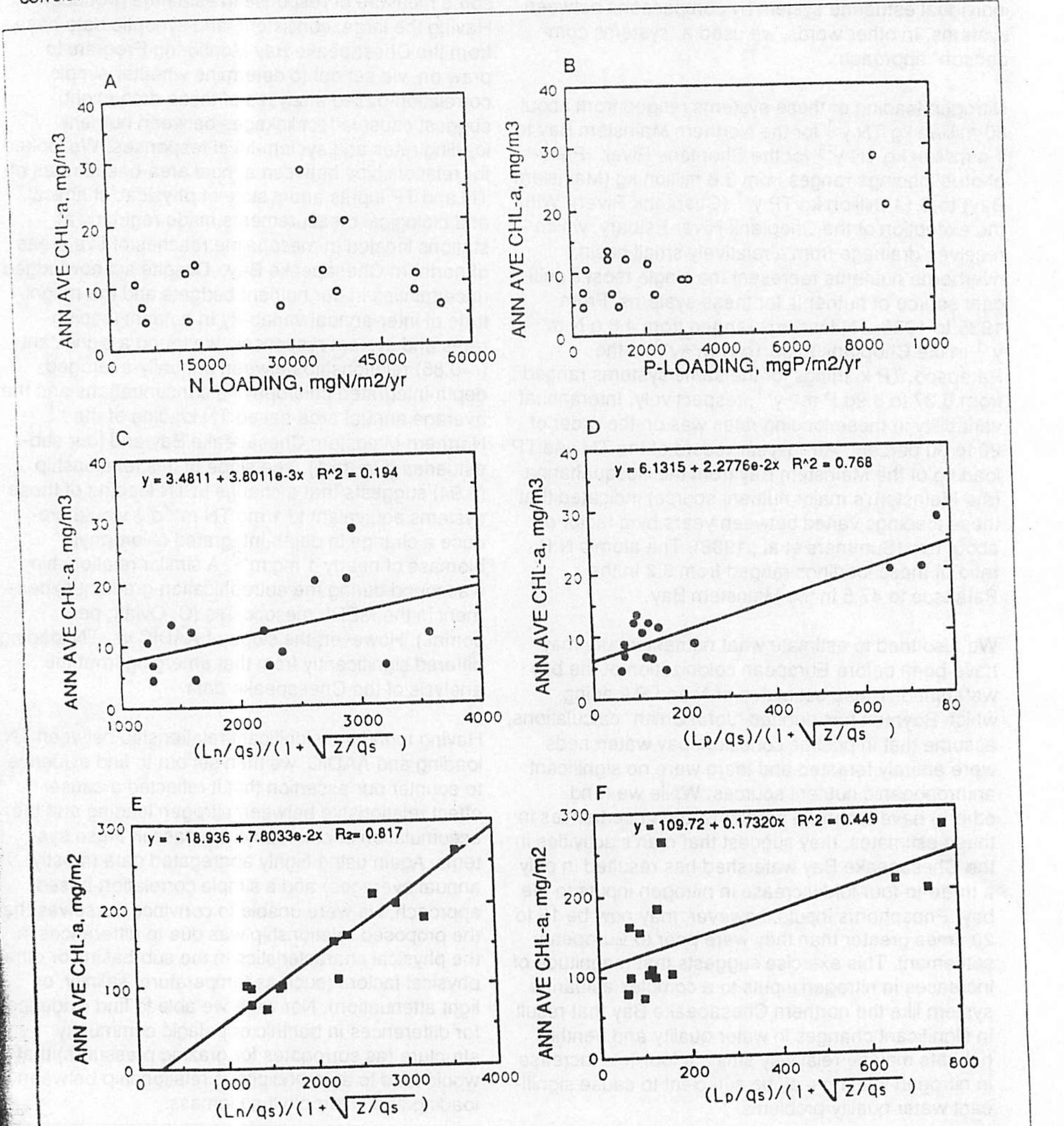
Nitrogen loading of these systems ranged from about 80 million kg TN y^{-1} for the Northern Mainstem Bay to 1.4 million kg TN y^{-1} for the Choptank River. Phosphorus loadings ranges from 3.6 million kg (Mainstem Bay) to 0.11 million kg TP y^{-1} (Choptank River). With the exception of the Choptank River Estuary, which receives drainage from a relatively small basin, riverborne nutrients represent the single most significant source of nutrients for these systems. From 1985 to 1986, TN loadings ranged from 4.8 g N m^{-2} y^{-1} in the Choptank to 33.1g N $m^{-2}y^{-1}$ in the Patapsco. TP loadings for the same systems ranged from 0.37 to 8.9g P $m^{-2}y^{-1}$, respectively. Interannual variability in these loading rates was on the order of 20 to 50 percent. An 11-year record of the TN and TP loading of the Mainstem Bay from the Susquehanna (the Mainstem's major nutrient source) indicated that these loadings varied between years by a factor of about two (Summers et al., 1988). The atomic N:P ratio of these loadings ranged from 8.2 in the Patapsco to 47.5 in the Mainstem Bay.

We also tried to estimate what nutrient inputs may have been before European colonization of the bay watershed. These estimates of N and P loading, which Boynton has dubbed "John Smith" calculations, assume that in pristine condition, bay watersheds were entirely forested and there were no significant anthropogenic nutrient sources. While we (and others) have technical problems and uncertainties in these estimates, they suggest that man's activities in the Chesapeake Bay watershed has resulted in only a three-to-four fold increase in nitrogen inputs to the bay. Phosphorus inputs, however, may now be 15 to 20 times greater than they were prior to European settlement. This exercise suggests that magnitude of increases in nitrogen inputs to a complex estuarine system like the northern Chesapeake Bay that result in significant changes in water quality and benthic habitats may be relatively small: a four-fold increase in nitrogen loading may be sufficient to cause significant water quality problems.

Relationships between loading and chlorophyll-a. We have suggested that if estuarine "eutrophication" is narrowly defined as an increase in loading rates or nutrient "supply" (Mortimer, 1969) then it becomes possible to formulate and test hypotheses about the causes and effects of that increased supply. That is, if water quality problems in Chesapeake Bay and other estuarine systems are due to increased rates of nutrient inputs, then there must be some demonstrable link between the rates of nutrient input and some measure of response in estuarine productivity. Having the large, consistent, and synoptic data base from the Chesapeake Bay Monitoring Program to draw on, we set out to determine whether simple correlation-based analyses of these data might suggest cause/effect linkages between nutrient loading rates and system-level responses. We looked for relationships between annual area-based rates of TN and TP inputs and a suite of physical, chemical, and biological measurements made regularly at stations located in mesohaline reaches of five areas of northern Chesapeake Bay. Despite acknowledged uncertainties in our nutrient budgets and the magnitude of inter-annual variability in nutrient loading rates and system responses, we found a significant (r=0.85) relationship between annually-averaged, depth-integrated chlorophyll-a concentrations and the average annual area-based TN loading of the Northern Mainstem Chesapeake Bay and four sub-estuaries (Figure 1). The slope of this relationship (0.94) suggests that a change in TN loading of these systems equivalent to 1 mg TN $m^{-2}d^{-1}$ would produce a change in depth-integrated chlorophyll biomass of nearly 1 mg m^{-2}. A similar relationship was found during the eutrophication-gradient experiment in the MERL mesocosms (C. Oviatt, pers. comm.). However, the slope of AADIC vs. TN loading differed significantly from that emerging from the analysis of the Chesapeake data.

Having found the significant relationship between TN loading and AADIC, we then set out to find evidence to counter our assertion that it reflected a cause/effect relationship between nitrogen loading and the accumulation of chlorophyll biomass in these systems. Again using highly aggregated data (mostly annual averages) and a simple correlation-based approach, we were unable to convince ourselves that the proposed relationship was due to differences in the physical characteristics in the sub-basins or other physical factors (such as temperature, salinity, or light attenuation). Nor were we able to find evidence for differences in benthic or pelagic community structure (as surrogates for grazing pressures) that would lead to a serendipitous relationship between loading and chlorophyll-a biomass.

Figure 1. Scatter plots showing the relationship between the annual average of depth-integrated chlorophyll-a and TN loading (Ln) and TP loading (Lp), for stations in the Northern Mainstem Chesapeake Bay, Potomac River, Patuxent River, Choptank River, and Patapsco River. There was little or no evidence of a relationship between TN and TP loading and average chlorophyll concentrations (A, B, C, and D). The best relationship emerges using annually averaged depth-integrated chlorophyll (E). The terms Qs (freshwater replacement time) and Z (mean depth) are derived from the fresh water analogy of this analysis, to help normalize the loading for differences in basin hydrography. Note that the relationship between loading and depth AADIC does not hold for TP, which indicates the relationship for TN is not a product of cross-correlation of depth in the x and y axes.



Some thoughts on "eutrophication." We have witnessed considerable confusion and debate over the meaning of the term "eutrophication." The exchange of information between estuarine scientists, managers, legislators, and the general public is difficult enough without the ambiguities of imprecise scientific language. In its report *Eutrophication: Causes, Consequences, Correctives*, the NAS (1969) itself failed to clearly define its subject when it stated that "the term eutrophic means well-nourished; thus, 'eutrophication' refers to natural or artificial addition of nutrients to bodies of water or to the effects of added nutrients." We suggest uncritical use of this oft-repeated phrase has not helped clarify thinking about nutrient-related problems in estuaries for two reasons: (1) the same term refers to process and its effects, and (2) it is difficult, perhaps impossible, to rank ecological systems along what was essentially an anthropogeneic scale of "nourishment." These criticisms are not new. Hutchinson's (1969) review at the same NAS symposium reveals that complaints over the use of the term began almost as soon as it was applied to ecological systems, and the debate was reflected in many papers included in the symposium volume. We suggest that use of the term "eutrophication" be restricted to Mortimer's (1969) definition which stated simply: "eutrophication is an increase in nutrient supply." By adopting this definition it becomes possible to pose and answer questions about "estuarine eutrophication " with less confusion ad greater precision. Thus, to the question "Is the Chesapeake eutrophic?" we reply that to the extent that we have evidence that the amounts of TN and TP entering the bay system have increased since European settlement, the answer yes.

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EUTROPHICATION AND BENTHIC INDICATOR SPECIES

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Reflecting on Pearson and Rosenberg's 1978 review, *Macrobenthic succession in relation to organic enrichment and pollution of the marine environment* (Pearson and Rosenberg, 1978) one is impressed that most subsequent studies of communities impacted by sources of organic enrichment, or organic inputs plus other pollutants, have sought novel numerical methods for comparing communities in space and time (Warwick, 1986) or have sought to simplify the task of enumerating and identifying benthic taxa (Gray and Pearson, 1982; Ferraro and Cole, 1990). Indices that use individual species abundances have ignored the repeatedly demonstrated fact that standard macrofaunal sieve sizes (1.0 mm, 0.5 mm) routinely miss a large fraction of the macrofauna (Grassle et al., 1985; Bachelet, 1990), and that, in subtidal communities, many O-group polychaetes and bivalves may take a year to recruit to a 1.0 mm mesh size (Buchanan and Moore, 1986). This is especially important in studies of the effects of pollution/enrichment since the dominant opportunistic species are notable for their small body size.

Some authors have recommended close attention to the natural history of individual benthic species that respond to enrichment. This is a review of current knowledge of the biology of certain enrichment opportunists: *Capitella spp.* (Grassle and Grassle, 1976) and *Mediomastus ambiseta* (Grassle and Grassle, 1984), and to a lesser extent the spionids, *Polydora ligni* and *Streblospio benedicti*, and the bivalve *Mulinia lateralis*.

Studies on many aspects of the biology of sibling species in the genus *Capitella* indicate that despite the morphological similarity of the adults, the species are genetically very distinct, stable entities differing in their allozyme mobilities, diploid chromosome numbers and karyotypes, many aspects of their reproductive biology, larval type and consequent dispersal potential, species-specific gregarine parasites, and functional properties of their intracellular hemoglobins. Moreover, although several sibling species occur sympatrically in almost all habitats examined, which constitutes a taxonomic problem, there is considerable fidelity between the species and particular habitats. Repeated collections in different years in the same habitat reveal the same *Capitella* species present, and often one species dominates from year to year. This is especially true in semi-enclosed estuaries and bays where there is a steady supply of organic matter (Tsutsumi, 1990; Tsutsumi et al., 1990).

In areas of intense point source or spatially circumscribed organic matter inputs (e.g., sewage outfalls and fish farms) the dominant *Capitella* species is often one with a lecithotrophic larva (i.e. potentially limited dispersal). Despite the general openness of marine systems the extent of the opportunists' distribution around such sources is very tight. Flume experiments on habitat selection on two such species, *Capitella* sp. I and II, indicate that this pattern is not only attributable to poor dispersal by the larvae but to active selection of sediments with high organic content (Grassle and Butman,1989).

Numerous other field and laboratory studies on *Capitella sp. I* feeding, growth, and population dynamics suggest that this species depends on high levels of available protein in sediments (Mayer et.al., 1986), and perhaps on micronutrients such as polyunsaturated fatty acids, sterols and pigments in newly-produced phytodetritus (Marsh et.al., 1990) and that standard measures of organic matter in sediments (CHN) will not explain the distribution and population dynamics of opportunists.

Eutrophication and sludge addition experiments in the MERL mesocosms also suggest that the different opportunistic and other dominant species vary significantly in their abilities and flexibility in exploiting new production vs. old (Oviatt et. al., 1987; Rudnick, 1989; Frithsen et.al., in preparation). These experiments also show that predicting long-term effects of widespread eutrophication on the benthos will not be easy because of the numerous subtle links between benthic and pelagic compartments. Predictions that eutrophication will leave the benthic species composition essentially unchanged while increasing species abundances and biomass (Beukema and Cadee, 1986) will seldom be fulfilled.

In areas subjected to intermittent hypoxia and occasional anoxia, the identity of species that persist will depend on their capacity for anaerobic respiration, capacities which may be very different in larval, juvenile and adult stages. When the benthic fauna is essentially removed by anoxic events or heavy winter mortality, the identity of successful colonists will depend on the species' capacity for reproduction in all seasons, larval dispersal and settlement behaviors, and hydrodynamic conditions. Attention to the reproductive biology and larval settlement behavior of "rare" species, which historically appear and dominate benthic communities for a period of several years and then disappear (Stull et al., 1984), as well as to the dominant enrichment opportunists, will enhance predictions about the probable outcomes of widespread eutrophication in estuarine and coastal waters.

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EUTROPHICATION AND MACROALGAE

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Three studies in two different ecosystems show: 1) effects of nitrogen and phosphorus enrichment on a coastal lagoon community (Rhode Island), 2) concentration of tissue nitrogen in response to seasonal changes in nitrogen loading in Rhode Island Sound, and 3) restoration of a eutrophic community (Mumford Cove, CT) as a consequence of nutrient diversion.

Study 1. Seagrass and algal beds showed a variety of responses when the water column was treated with low-level additions of ammonium, nitrate, and phosphate. The nutrients were added separately to three uniform seagrass beds of a temperate coastal lagoon during 1979 and 1980.

Ammonium caused the production of dense mats of free-floating green algae *Enteromorpha plumosa* and *Ulva lactuca*. It also stimulated growth in both the leaf and root-rhizome fractions of *Zostera marina*. This growth response in *Z. marina* was greater in the area where current reached 12 cm s^{-1} than in the area with little or no current. The concentration of nitrogen in the tissue did not change. In contrast, where current was lacking, *Z. marina* growth increase with ammonium was small, but the concentration of nitrogen in the tissue doubled over that in control plots. The growth of *Ruppia maritima* was inversely related to the growth of green algae in the same plots. The red alga *Gracilaria tikvahiae* did not grow better in ammonium, but its tissue reddened.

Nitrate additions enhanced the growth of the green seaweeds *Enteromorpha spp.* and *U. lactuca*, but not *Z. marina* or *R. maritima*. *G. tikvahiae*, when fertilized in isolation from other plants, showed a marginal response to this nutrient, and the tissue always reddened.

Phosphate enhanced growth in *Z. marina* and *R. maritima* exposed to moderate current. *G. tikvahiae* growing alone showed a small growth response to phosphate. The phosphate made no difference in the growth of the green seaweeds.

None of the nutrient supplements noticeably altered the species composition of either epiphytic or planktonic algae associated with the beds, although we did detect small increases in their numbers. The rapid and dense growth of green algae in nitrogen-enriched water probably limited growth of adjacent seagrasses and red algae. Because these seaweeds did not use the phosphate, it became available to other plant components. The overall floral response to nutrient addition in seagrass communities depends, therefore, upon the particular nutrient supplied, the ability of alternate species in the area to compete for that nutrient, and the velocity current in the specific area.

Study 2. Tissue nitrogen was assessed monthly for 16 months in five species of perennial macroalgae representing three phyla at one location in Rhode Island Sound. The species showed a remarkable similarity in their pattern of seasonal fluctuation in both nitrate and total nitrogen. The period of greatest accumulation (January through March) coincided with the period of highest concentration of inorganic nitrogen in the water, and for most of these algae it was also the time of least growth. Conversely, the period of lowest tissue nitrogen (less than 50 percent of the winter value, May through July) coincided with the period of lowest inorganic nitrogen in the water and highest algal growth. The greatest accumulation of nitrate was found in *Laminaria sacharina (L.) Lamour* (80 μmol g^{-1} dry wt), four times as much as that measured simultaneously in the other species and 560 times the ambient concentration. By April, the concentration of internal nitrate had dropped to nearly undetectable levels, but in August it began to accumulate again–a pattern that was repeated in *Chondrus crispus* Stakh. In *Ascophyllum nodosum (L.) Le Jolis, Fucus vesiculosus L. and Codium fragile subsp. tomentosoides (Van Goor) Silva*, the period of negligible internal nitrate level extended from March to December. The greatest concentration of total tissue nitrogen was measured in *C. crispus* (4.8 percent dry wt.), double the maximum in *L. saccharina* (2.3 percent dry wt.).

Study 3. Ulva lactuca had become a "virtual monoculture" in Mumford Cove, CT, prior to the diversion of sewage discharge in October 1987. A model developed for this cove predicted that nutrients in flux with the sediments would sustain this population indefinitely. Contrary to that prediction, within one year of sewage diversion total area covered by macrophytes dropped from 74 percent to nine percent of the cove. Of this value, *Ulva* fell to less than three percent. A monthly survey (May through November 1988) at seven stations on a distance gradient from the original discharge site, showed most *Ulva* biomass closest to the inlet and in the earlier sampling periods. Species number increased near the sound. Of 29 species tabulated: two were angiosperms, six chlorophytes, nine phaeophytes and 12 rhodophytes. A revised model developed from nitrogen flux measurements and C:N assessments of the seaweeds suggest that the macroalgae have removed the nitrogen from the sediment. And indeed, a second monitoring interval (May through October 1989) showed no spring bloom of *Ulva*. It is not expected that Ulva blooms will reappear. Zostera has not yet colonized the upper reaches of the cove, but there appears to be no reason why it would not.

EFFECTS OF WATER COLUMN PROCESSES ON DISSOLVED OXYGEN: NUTRIENTS, PHYTOPLANKTON, AND ZOOPLANKTON

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The mechanisms by which phytoplankton and zooplankton influence the relationship between nutrient loading and anoxia are the subject of this contribution. It is generally assumed that the spatial and temporal extent of anoxic bottom waters has increased in response to anthropogenic nutrient loading due to an interannual increase in phytoplankton productivity. However, synoptic observations of environmental factors, phytoplankton, and zooplankton in the mesohaline reach of Chesapeake Bay on seasonal to interannual scales from 1984 to 1988, indicate that oxygen depletion is not related to phytoplankton productivity on these scales. Rather, nutrient loading is linked to oxygen depletion via the accumulation and deposition of phytoplankton biomass.

The annual cycles of phytoplankton biomass and productivity are seasonally out of phase with biomass peaking during spring and productivity during summer. The accumulation of biomass during spring reflects the combined effects of high nutrient input, two-layered estuarine circulation, and low zooplankton grazing. The summer productivity maximum is driven by the annual cycle of incident solar radiation, and occurs as a consequence of the recycling of nutrients delivered to the Bay during the previous spring. Most summer phytoplankton production is consumed in the surface mixed layer. In contrast, a large fraction of spring production is deposited in the benthos.

The spring accumulation of phytoplankton biomass is more than sufficient to fuel oxygen depletion. The rate of oxygen depletion during April and May is a function of temperature, and shows little interannual variability. In contrast, the spatial and temporal extent of summer anoxia exhibits large variations from year to year, primarily due to physical processes governing vertical mixing. The magnitude of the summer productivity maximum does not appear to be a factor.

These observations lead to the conclusion that the seasonal cycle of oxygen depletion of bottom water in Chesapeake Bay is, to the first order, independent of nutrient loading under current conditions of climate and watershed development. Apparently, nutrient-dependent accumulations of phytoplankton biomass generate an aerobic oxygen demand that exceeds the oxygen assimilation capacity of the Bay. Consequently, the impact of management decisions that reduce nutrient loads to the Bay by a given percentage (e.g., 40 percent) will depend on the degree to which this assimilation capacity is exceeded as well as on climatic factors that govern nonpoint nutrient inputs from the watershed and mixing within the Bay.

SCALES OF EFFECTS AND RECOVERY OF BENTHIC AND DEMERSAL COMMUNITIES ASSOCIATED WITH REDUCTIONS IN WASTE EMISSIONS

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By the early 1970s nearly 200 km^2, or four percent, of the Southern California mainland shelf experienced the biological effects of sewage discharges from five major metropolitan areas. The kinds and scales of effects on transparency, kelp beds, benthic macrofauna, and demersal fish varied among the discharge sites and there was an exponential relationship between the size of bottom areas affected by changed benthic communities and the mass emission rate of suspended solids from the outfalls. Effects ranged from elevations in benthic biomass and decreases in benthic infaunal diversity, to alterations in fish community structure and high incidence of fin erosion, and several other abnormalities in bottom fish. Over the years, attempts were made to implicate DDT, PCBs, metals, petroleum hydrocarbons, grain size, and sulfides as causes of the disorders.

During the 1970s and 1980s, all dischargers spent billions of dollars implementing source controls for toxic chemicals and increased treatment (removal of suspended and settleable solids and some BOD) such that mass emission rates are now comparable to inputs 40 or more years ago. Sludge discharge into Santa Monica submarine canyon was discontinued. Intensive monitoring has continued and research on long-term patterns–such as between pelagic fish stock size, waste emissions, and climate change (NOAA), or between mass emissions and oxygen and transparency–are completed or nearing completion. Fin erosion has virtually disappeared, the total area of altered marine communities has shrunk dramatically, and the biomass of fish and infauna has declined. Nonetheless, public discharge agencies are being forced to implement additional treatment at costs equivalent to those already spent.

The exact size and total amount of the ecological changes (percent recovery) has not been determined because no one has used the abundant existing monitoring data for that purpose. But the data exist, and it is possible to make new computations to confirm the earlier relations between waste emissions and scales and intensity of biological effects. In short, information exists to confirm the assimilative capacity of a Pacific open coastal-shelf ecosystem for waste materials, including the possibility to distinguish effects of organic and nutrient enrichment from effects of toxicants. This should be done not only to confirm a potentially useful assimilative capacity forecasting tool but also to postulate what, if any, additional treatment is needed.

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ESTUARINE NUTRIENT ENRICHMENT AND PHYTOPLANKTON PRODUCTION: WHAT FACTORS REGULATE THE EXPRESSION OF EUTROPHICATION?

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Phytoplankton play a key role in the linkage between nutrient enrichment and occurrence of deleterious processes that are often characterized under the broad term of *eutrophication*. While both freshwater and marine mesocosm systems often display a fairly well-behaved, dose-response relationships between nutrient loading and phytoplankton production and biomass, this relationship is more variable in natural estuarine ecosystems. Herein, data from two estuaries–the Delaware and the Mobile–are used to discuss the role that natural physical and biogeochemical processes play in regulating the expression of eutrophication in estuarine ecosystems.

As used in this discussion, eutrophication refers to series of ecological changes initiated as a result of nutrient enrichment. These changes may be perceived as either beneficial or detrimental and include responses such as increased biomass, species composition changes, and dissolved oxygen depletion. Similarly, although nutrient enrichment is a necessary prerequisite to eutrophication, it does not ensure that an estuary will display a response. This discussion focuses on factors that regulate the expression of various ecological processes to nutrient enrichment.

Delaware Bay Estuary. The Delaware Estuary is one of the most enriched in the United States, with loadings of greater than $5x10^3$ mmoles m^{-2} y^{-1} for nitrogen (N) and $5x10^2$ mmoles m^{-2} y^{-1} for phosphorus (P). While these loadings, and the physical flushing characteristics of the system, result in high nutrient concentrations in the headwaters of the estuary, these nutrients are removed from the system as a result of phytoplankton production and sequestered in phytoplankton biomass in the lower estuary. However, this growth is not facilitated by noxious algae, nor does the biomass accumulation result in subsequent hypoxia in bottom waters such as is seen in many nutrient-enriched estuaries that are considered eutrophic.

Recent research has shown that the response of the phytoplankton community is moderated by elevated turbidity in the upper regions of the estuary that limit biomass levels (Pennock, 1985). One effect of light limitation in the upper region of the estuary is that it allows anthropogenic nutrients to be diluted before they are assimilated by the phytoplankton, a process that limits the overall accumulation of biomass if nutrients become limiting during bloom events. Recent experiments to test nutrient limitation in the Delaware indicate that despite high loading rates P becomes limiting to phytoplankton growth at the end of the spring bloom, while N may limit growth at the mouth of the estuary during summer. The relative importance of these limiting nutrients appears to be related to the geochemical sequestering of phosphorus in the upper regions of the estuary. Such alternation in the processes that limit phytoplankton growth are crucial regulators of the way in which the Delaware responds to anthropogenic nutrient enrichment.

Mobile Bay Estuary. Mobile Bay receives moderate loadings of both nitrogen and phosphorus, primarily from upstream nonpoint sources. Despite significant loadings, however, ambient nutrient concentrations at the headwaters of the bay attain concentrations of <20 µM inorganic N and <2µM inorganic P (10 and five times less than those for the Delaware, respectively) as a result of the relatively high flushing rate of the system. Although these characteristics suggest that the Mobile Bay Estuary should have a relatively low susceptibility to nutrient enrichment, several responses often associated with eutrophication are evident. For example, chlorophyll-a concentrations have been observed as high as 100 µg l^{-1} during the winter, and recurring hypoxic conditions on the eastern shore of the mid-bay region are seen during summer.

While organic loading to Mobile Bay is undoubtedly important to the observed response in chlorophyll-a and dissolved oxygen (these data are not yet available), it is evident that the Mobile Estuary responds differently to nutrient loading than the Delaware. This response is due, in part, to stratification that is set up as a result of salinity intrusions into the Bay through the shipping channel. Ongoing studies are expected to further explain the response of various components to nutrient enrichment.

Conclusions. Evidence from the Delaware Bay and Mobile Bay estuaries provides important insight into the roles of physical and biogeochemical processes in the regulation of phytoplankton growth in estuaries. These processes play a major role in regulating the ways in which estuaries respond to nutrient enrichment, and whether eutrophic responses are observed. Ultimately, factors such as light-limitation and biogeochemical processing must be incorporated into our models of how estuaries respond (either negatively or positively) to nutrient enrichment.

NORTHERN GULF OF MEXICO HYPOXIA: EFFECTS ON BENTHIC COMMUNITIES

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Hypoxia, or dissolved oxygen concentrations of less than 2 mg l^{-1}, regularly occurs in bottom waters of the continental shelf off Louisiana. On a shelfwide basis, the areas of oxygen-deficient bottom waters are large in area and volume. Off the southeastern Louisiana coast, hypoxia is regularly found over a large area from mid-June through mid-August, and intermittently found as early as mid-April and as late as mid-October. Hypoxia has been documented on the southwestern Louisiana coast, but the area is less well-defined and occurs ephemerally. Even fewer incidences of hypoxia are recorded for the upper Texas coast.

In conjunction with hydrographic surveys along a transect off Terrebonne/Timbalier bays in 1985 and 1986, the benthic macroinfauna and meiofauna were examined. Hypoxia was present in bottom waters from June through early August 1985, and from mid-April through mid-October 1986 (continuous and severe from mid-June through mid-August). Numbers of individuals and numbers of species of macroinfauna were greatly reduced during hypoxia. These communities were dominated by the polychaetes *Mediomastus ambiseta*, a motile subsurface non-selective deposit feeder; and *Sabellides sp. A*, *Magelona spp. H*, and *Paraprionospio pinnata*, tubiculous surface deposit feeders. There were few pericarideans or larger crustaceans. Non-polychaetes were bivalves, gastropods, nemerteans, phoronids, and sipunculids, which were present primarily during the spring recruitment period and, to some extent, during the fall recruitment period. Most of the individuals collected in this study were extremely small or juveniles. Seldom were any larger or mature individuals collected. The relatively small size of the individuals, the large number of juveniles present, and the dominance of the community by a few opportunistic polychaete species indicates an environment regularly stressed by hypoxic events.

No significant relationships were found between either the numbers of individuals or species and sediment texture characteristics. Bottom water oxygen concentrations at the time of sample collection (no temporal average) compared to numbers of species and individuals showed two populations of samples: those in extremely low oxygen concentrations (0.5 mg l^{-1} or less than seven percent oxygen saturation) and those in more oxygenated waters of 26 to $\geq$ 100 percent oxygen saturation. Numbers of individuals and species in samples where oxygen concentrations were below 0.5 mg l^{-1} were positively correlated with bottom water oxygen. In the more oxygenated waters, these parameters were correlated more closely with other hydrographic parameters such as bottom water temperature.

The results from this study are compared with others from the Louisiana inner continental shelf and the upper Texas coast, as well as preliminary data from 1990. Observational data from demersal fish trawls, shrimp tagging, diving, and a remotely operated vehicle (ROV) are presented.

EUTROPHICATION AND BIOGEOCHEMICAL SILICA DEPLETION IN THE GREAT LAKES

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Biogeochemical silica depletion is the long-term decrease in water column silica concentrations resulting from nutrient-induced increases in diatom production and sedimentation. In the Laurentian Great Lakes, this consequence of eutrophication is the direct result of increased anthropogenic loadings of the principal limiting nutrient, phosphorus. Enrichment of waters with phosphorus thus induces secondary silica limitation for diatom growth and affects the phytoplankton assemblage, favoring shifts in species composition from assemblages dominated by diatoms to those with greater proportions of cyanobacteria, flagellates, and other algal forms with no absolute requirement for silica.

Two stages of silica limitation have been identified as ecosystem responses to different degrees of phosphorus loading. Epilimnetic silica depletion, the first stage, occurred at total phosphorus concentrations of less than 10 µg l^{-1} P. In Lake Michigan, epilimnetic silica depletion developed rapidly between 1955 and 1970, when winter maximum silica concentrations decreased about 3.0 mg l^{-1} Si and summer minimum concentrations decreased from 2.2 to 0.2 mg l^{-1} Si. In Lake Ontario, epilimnetic silica depletion based on inferences from the paleolimnological record also developed rapidly within two decades. However, it occurred approximately 100 years earlier or soon after early European settlement in the drainage basin. The second stage of silica limitation occurred in Lake Ontario when total phosphorus concentrations increased to approximately 20 µg l^{-1} P. With this level of phosphorus enrichment, silica concentrations over the entire water column were reduced to 0.5 mg l^{-1} Si or less throughout the year. This stage is termed severe silica limitation because silica concentrations are always below optimal levels for diatom growth for many species of diatoms. The only areas of the lake that are not affected are nearshore areas near sources of silica loading.

Applications of the silica depletion paradigm for ecosystem assessment of eutrophication are that it: 1) represents a seasonal, integrated response to low levels of nutrient enrichment, 2) represents one of the most sensitive chemical measures of an ecosystem response to nutrient loading, and 3) also provides an historical (paleolimnological) assessment of nutrient enrichment. Because the application is based on the biological response of diatoms to nutrient enrichment a parallel and more detailed assessment of effects of nutrient enrichment can be discerned from analysis of diatom assemblages.

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EFFECTS OF EXCESSIVE NUTRIENT LOADING ON THE EELGRASS COMMUNITY

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As an estuary becomes eutrophic, the level of nutrient loading has a direct impact on the eelgrass *(Zostera marina)* community. The research reported here examines the effects of excess nutrient loading on eelgrass populations and the resulting changes in plant composition of the eelgrass community in both field and experimental mesocosm studies. The community dominated by eelgrass includes numerous species of both micro- and macroalgae which grow as epiphytes on the eelgrass blades, as well as species of unattached macroalgae. In addition to the plants, a host of animals are associated with the eelgrass community. These include many invertebrates (worms, snails, and crustaceans) and numerous fish species, many of which can be important in determining the effects of excess nutrient addition.

Excessive nutrient loading in an estuary can eliminate the eelgrass community by pushing the eelgrass system in one of three directions: 1) toward a plankton-dominated ecosystem, 2) toward a macroalgal dominated ecosystem, or 3) toward a system with excessive amounts of epiphytic algal growth. All three possible outcomes of eutrophication have been observed in the field and have been directly associated with documented eelgrass declines. Although nutrient uptake at high concentrations by eelgrass leaves has been well documented (Short, 1987), there are severe indirect impacts on the eelgrass community associated with eutrophic conditions. Nutrient loading can stimulate phytoplankton growth which shades eelgrass and can reduce its productivity and distribution. Benthic algal populations have been shown to overgrow eelgrass in situations of excessive nutrient loading (Harlin and Thorne-Miller, 1981; Neckles, 1989). Our observations of macroalgal, epiphytic algal, or phytoplankton inhibition of eelgrass communities along the East Coast have included Mumford Cove, CT; Charlestown Pond, RI; and Waquoit Bay, MA among others.

Our experimental mesocosm studies have quantified the effects of specific nutrient loading levels on eelgrass communities (Short, 1987). They have proven valuable in documenting the shift from an eelgrass-dominated community to situations dominated by various algal forms. Six mesocosms were planted with eelgrass and grown under the same conditions for two months, after which the tanks were paired according to eelgrass density and one of each pair was enriched with nitrogen and phosphorus at a level producing concentrations six times ambient nutrient concentrations. Although tanks received a relatively constant loading rate, the concentration of nutrients in the tanks decreased over time as a result of increases in plant uptake rates and incorporation of nutrients into biomass. Experiments were run for four months under continuous nutrient additions; plant characteristics, algal populations, and loading rates were monitored.

The overall effect of excessive nutrient loading on eelgrass populations was most evident in the reduction of shoot density (50-100 shoots m^{-2}) observed in the enriched tanks versus the eelgrass density (200-320 shoots m^{-2}) growing at ambient nutrient concentrations. Additional responses to nutrient additions included a reduction in eelgrass areal growth rate, and a decrease in leaf length by more than 20 cm in all enriched eelgrass tanks. Although eelgrass responded by becoming shorter and less dense, the specific growth rate of the enriched treatments was actually greater than that of the unenriched treatments. Stimulated specific growth rates coupled with reduced areal production, biomass, and plant size often indicate a stressed eelgrass population. The reduction in eelgrass abundance and health in the enriched treatments appeared to result from inhibition by various algal growth forms that were stimulated by the treatments.

The response of the algal community to the enrichment treatment reflected all three scenarios seen in field observations. Of the three enriched tanks, one became dominated by phytoplankton, with an intensive phytoplankton bloom maintained throughout the experimental period, microalgal epiphytic growth on the plants initially, and no macroalgal growth. The second "replicate" developed macroalgal mats floating at the surface which were dominated by the green alga *Enteromorpha*. In this treatment, extensive algal growth appeared to crowd out the eelgrass and block the amount of light reaching the plants. Some blades penetrated through the macroalgal mat while others died below the mat. Eelgrass in the third "replicate" became overgrown by epiphytic algae. The epiphytic cover was composed primarily of diatoms and small macroalgal forms.

The reason for the variation in response to identical nutrient enrichment treatments is complex and as yet not completely understood. The first enrichment "replicate" became dominated by a phytoplankton community which appeared to inhibit the growth of other algae. The second enriched tank initially showed a substantial increase in both epiphytic and macroalgal growth. However, the occurrence of an amphipod bloom within this treatment dramatically reduced the epiphyte population, allowing the macroalgae to dominate. The third enrichment "replicate" also began as a combination of epiphytic and macroalgal growth and had a population of amphipods develop. However, in this "replicate", the carnivorous fish (primarily sticklebacks and pipefish) that had been introduced to all tanks, were able to keep the amphipod population under control. The fish prevented extensive amphipod grazing on the epiphytes. As a result, the third treatment ended up being dominated by epiphytic growth, with the macroalgal component remaining small.

Thus, quite unexpectedly, the eelgrass mesocosm enrichment study has demonstrated the effectiveness of "top down control" of trophic levels for regulating macrophyte populations. Both the fish and herbivorous amphipods appeared to regulate the dominant form of primary producer within the experimental treatments. The factors determining the success of phytoplankton populations within experimental treatments of this kind are as yet unclear, and whether competition is important remains to be examined.

The effects of our nutrient enrichments have been an overall reduction in eelgrass health and biomass within the treatments and a conversion to algal-dominated communities. Examples of these same types of community changes have been documented over the last five years in estuaries along the New England coast and include the decline and loss of eelgrass in Waquoit Bay, as a result of eutrophication. Within the Waquoit Bay System, epiphytic algal populations became dominant in Eel Pond, macroalgal blooms covered eelgrass beds in Hamblins Pond, and phytoplankton populations have eliminated eelgrass in parts of Great River and Jehu Pond. Similar evidence of eelgrass loss has been documented in New Hampshire, Rhode Island, Connecticut, and elsewhere in Massachusetts.

It is clear that eutrophication of coastal areas contributes to the demise of eelgrass populations. Eelgrass populations are also declining from other factors including wasting disease, mechanical disruption, and increased construction within estuarine areas. However, eutrophication is the major threat to the long-term survival of eelgrass in coastal New England and elsewhere in the world.

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INCREASING WORLDWIDE FREQUENCY OF NUISANCE ALGAL BLOOMS

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Evidence was presented that long-term nutrient enrichment of coastal waters is accompanied by increases in phytoplankton biomass and primary production. Although there are no phytoplankton indicator species of incipient or advanced stages of coastal eutrophication presently identifiable, a significant shift in phytoplankton community structure at the phylogenetic level appears to be occurring globally in response to coastal nutrient enrichment. This phylogenetic shift has been towards increased abundance and seasonal predominance of flagellates and non-motile, nannoplanktonic chrysophytes and, in some cases, N-fixing blue-green algae. Many of these species increasingly exhibit noxious, harmful, and toxic blooms, contributing also to increased hypoxic/anoxic episodes. In fact, there is evidence that there is an ongoing global increase in the frequency, number of occurrences, and dynamics of these toxic and harmful phytoplankton blooms, based on evidence from the Baltic Sea, Kattegat, Skagerrak, Dutch Wadden Sea, North Sea, Black Sea, Tolo Harbour (Hong Kong), Korea, Seto Inland Sea, and northeastern coastal waters of the United States. A feature independent of the global-bloom epidemic is the apparent global spreading of certain species, some of which exhibit toxic and harmful blooms. Local stimulation of the indigenous "hidden flora" to bloom levels may also be increasing in response to elevated nutrient levels.

Evidence is presented that anthropogenic enrichment of N and P, leading to long-term increases in these nutrient loadings, has paralleled this global increase in toxic phytoplankton blooms. This increased nutrient loading has stimulated increased biomass concentrations and primary production levels. This increased nutrient N and P nutrient loading has also led to long-term declines in the Si:N and Si:P ratios. It is suggested that the decline in the Si:P and Si:N ratios has particularly favored non-diatom blooms in response to nutrification, and such ionic ratio regulation within nutrification is a key factor associated with the global epidemic of novel toxic and harmful phytoplankton blooms and phylogenetic shifts in phytoplankton biomass predominance in coastal seas. Experimental evidence is presented in support of this hypothesized role of silica and changing stoichiometric ratios with N and P. Using natural phytoplankton populations enriched with various N, P, and Si loadings, the dependence of N uptake on Si levels, increased biomass and production accompanying Si loadings, and the shift to elevated primary production of non-diatomaceous nannoplanktonic forms accompanying reductions in the Si:N uptake ratios was demonstrated. Experimental evidence from culture studies demonstrating the influence of the Si:P ratio on species selection and growth was also presented.

The notion that eutrophication is a dynamic process characterized by different stages, with differing ecosystem effects, was developed. The fundamental conclusion was that the phytoplankton niche structure accompanying ongoing nutrient enrichment of coastal waters appears to be changing in many coastal waters, increasing hypoxic/anoxic potential, and stimulatory to increased toxic and harmful phytoplankton bloom episodes disruptive to ecosystem pathways and posing human health problems.

EUTROPHICATION IN THE ALBEMARLE-PAMLICO ESTUARIES

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Information from previous and ongoing research and monitoring studies are used to illustrate three examples of hypothesized relationships between eutrophication "Factors" and "Effects" in three sub-estuaries of this large North Carolina system (Table 1).

Example 1. Blooms of blue-green algae have occurred during some, but not all, recent summers along the lower Chowan and Neuse River estuaries at the western ends of Albemarle Sound and Pamlico Sound, respectively (NC NRCD, 1986; Christian et al., 1986). The Chowan blooms are largely composed of the nitrogen-fixing species *Aphanizomenon flos aquae, Anabaena spiroides,* and *Anabaena flos aquae,* while in the Neuse, *Microcystis aeruginosa* has been the dominant blue-green (Paerl, 1982; Paerl, 1987). Fortunately both the Chowan and Neuse blooms have been limited to the riverine and freshwater tidal portions of the estuaries because the blue-green species comprising them cannot tolerate saltwater (Paerl, 1983). The Chowan blooms typically cover a 30 km stretch of the river, and the Neuse blooms extend 50 km from below Kinston, NC to the river's mouth at New Bern. Usually the blooms persist for several weeks to two months. Chlorophyll-a levels typically are several hundred $\mu g\ l^{-1}$, but during the 1983 Neuse bloom chlorophyll concentrations were over 600 $\mu g\ l^{-1}$ for a short period.

Table 1. Factors influencing the expression of eutrophication effects in the Albemarle-Pamlico estuaries, North Carolina.

Effect and Sub-estuary Where Expressed	Factor(s)
Appearance of Nuisance Algal Blooms (Chowan River) (Neuse River)	River Flow
Phytoplankton Primary Production (Chowan River) (Pamlico River) (Neuse River)	Seasonal N Loading Rates and Recycling
Phytoplankton P Primary Production (Pamlico River)	Annual N and P Loading Rates

It has not been shown that N and P levels in the Chowan and Neuse Rivers are substantially higher in bloom years than in years when blooms do not occur. For example, during the four years between 1979 and 1982 (two of which were bloom years) average dissolved inorganic nitrogen (DIN) and dissolved inorganic phosphorus (DIP) concentrations at 11 stations along a 400 km stretch of the Neuse were high and showed little year-to-year variability. Ammonium nitrogen levels averaged 3 to 9 μM, and nitrate nitrogen averaged about 30 uM. Orthophosphate levels were around 3 to 10 μM. Despite wide areal ranges in N and P levels in the Neuse, for any given point they varied only about 10-20 percent from year to year during this four-year period (Stanley, 1983).

Previous nutrient kinetics studies in the Chowan and Pamlico River estuaries have shown that these N and P concentrations should not limit phytoplankton growth (Stanley and Hobbie, 1981; Kuenzler, et at., 1979). Furthermore, one researcher concluded that a 50 percent reduction in DIP concentrations would be necessary to approach phosphorus-limiting conditions in the lower Neuse. He also recommended a 30 percent cutback in DIN in order to limit algal growth, but warned that reducing N without also limiting P loading might simply stimulate N-fixing algae like those in the Chowan to replace *Microcystis aeruginosa,* the non-N-fixing algal species in the Neuse (Paerl, 1987).

Occurrence of algal blooms in the lower Chowan and Neuse Rivers seems to coincide with low flow in the rivers (Figure 1). This observation led to an experimental study (Christian et al., 1986) to test the following two-part hypothesis: 1) summer algal blooms are prevented from developing in the Neuse River by intermediate-to-high flows because nutrient-rich water is carried to the estuary before algal densities can reach bloom densities; and 2) summer blooms do develop during sustained low-flow periods because the algae grow at a more rapid rate than when flow is high, and because water parcels remain in the river long enough for bloom densities to be reached. Two assumptions were made: 1) algal growth nutrients are sufficient every summer to allow bloom formation; and 2) blue-green algae are always present in the headwater tributaries in sufficient number to serve as an inoculum for bloom development. The hypothesis offers a mechanism to explain how river flow can act to control bloom development. Table 2 summarizes the effects that flow has on three key parameters which, in turn, influence blue-green algal growth rates. To test the hypothesis, researchers developed a mathematical model to simulate the effects these three variables have on the growth of blue-green algae in the Neuse. The model predicted algal densities under various flows. Information on the time-of-travel vs. flow relationship was generated by means of a series of dye tracer studies to measure river velocity. In the laboratory, studies were made to quantify the effects of light and temperature on growth of the dominant blue-green in the Neuse, *Microcystis aeruginosa*. Model predictions were in agreement with observations of bloom occurrence or non-occurrence over the past decade (Christian et al., 1986).

Example 2. The amount of nitrogen recycling in the lower Chowan River (northeastern North Carolina) was estimated from differences between inflow and assimilation (Stanley and Hobbie, 1981). Inflow was much higher in winter (e.g., 1,127 tonnes N in January) than in summer (185 tonnes in August), because of the wintertime high flow rates and high concentrations of particulate (PN) and dissolved inorganic nitrogen. About 69 percent of the total annual nitrogen inflow was dissolved organic N (DON), 13 percent was PN, and the remaining 18 percent was mostly ammonium. As much as 33 percent of the incoming DON may be broken down in this section of the river. Assimilation by the river plankton, computed from short-term ^{14}C and ^{15}N uptake measurements, averaged 100 g C m^{-2} y^{-1} and 32 g N. The seasonal patterns of inflow and uptake of DIN in the river were completely out of phase; during January inflow was 12 times higher than uptake, whereas during summer DIN uptake exceeded inflow by as much as 75 times. For the whole year, as much as two-thirds of the DIN assimilated may have come from regeneration within the river. Apparently, most of the DIN that was regenerated was ammonium. Thus, nutrient cycling seems to be important near the mouths of such coastal rivers (Stanley and Hobbie, 1981).

Figure 1. Relationship between flow and chlorophyll-a concentration at a station in the lower Neuse River, North Carolina.

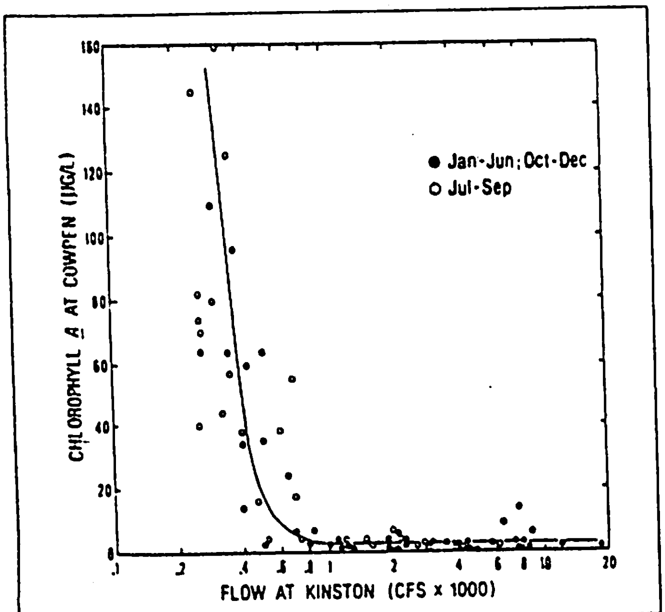

Table 2. Hydrologic parameters directly influenced by river flow.

Parameter	How affected by flow	
	Low Flow	High Flow
Velocity	Decreaces (+)	Increases (-)
Clarity	Increases (+)	Decreases (-)
Turbulence	Decreases (+)	Increases (-)

+ = favors bloom development
- = does not favor bloom development

Example 3. Trends in annual nutrient (nitrogen and phosphorus) production rates over the past century for each of the Albemarle-Pamlico sub-basins were estimated from historical population data, agricultural census statistics, and wastewater treatment data (Stanley, 1989). Nonpoint source production was computed by mass balance (for croplands) or by land-use coefficients (for other land use categories). Farm animals' N and P production was estimated by multiplying animal numbers times the estimated per animal nutrient release. Industrial point source production was calculated by multiplying discharges times the N or P effluent concentrations. Municipal production was estimated be multiplying the sewered population times the per capita N or P production times a "treatment factor," which ranged from one (untreated) to 0.47 (secondary treatment), depending on the type of wastewater treatment practiced by the municipal treatment plant.

Land use changed little during the past century. Cropland N mass balances (inputs minus outputs) rose from near 0 in the late 1800s to 54 kg ha^{-1} in 1974. The major cause for the rise was fertilizer application in excess of crop yield. The cropland P balance peaked at 31 kg ha^{-1} in 1969, and has declined since. N production from all sources nearly doubled over the past century, but there appears to have been little change in the past two decades. P production also doubled, rising gradually after 1920, with a peak around 1970 and little change since.

Agricultural fertilizer use, farm animals, and increased urban population have been the major contributors to the increases in total N and P production. Before 1950 there was little nutrient removal from wastewater discharged into the rivers and streams. Thus, N and P production grew at about the same rate as the sewered population. As secondary treatment came into use, the rate of increase in municipal nutrient production slowed. But there has been little improvement since 1975 because further increases in treatment efficiencies have not occurred, or have occurred more slowly than necessary to keep up with increases in urban population. Rapidly increasing swine and poultry production in the basin could lead to greatly increased N and P loading to the estuaries in the future. The above estimates of nutrient production seem to be about twice the estimates of nutrient actually delivered to the estuary. Presumably the difference is due to instream losses such as sedimentation and denitrification.

When twenty years of chlorophyll-a and nutrient concentration data for the Pamlico River estuary were analyzed for trends (by the Seasonal Kendall Tau Test), no consistent patterns emerged (Stanley, 1988). However, one cannot conclude from these results that the estuary does not respond to annual variations in loading. In fact, when a regression analysis was made between estimated N loading and chlorophyll-a, some positive relationships did emerge, depending on the time intervals chosen (Nixon, 1989). Specifically, data from 1980 through 1984 showed a five-fold increase in the mean annual standing crop of chlorophyll-a in the mid-to-lower estuary when the input of dissolved inorganic nitrogen from the Tar River increased almost four-fold. Further analysis showed there is a strong positive relationship between DIN input during January through June and the size of the winter-spring phytoplankton bloom in the lower Pamlico. The inverse relationship in the upper estuary may be due to higher turbidity and flushing rate in this region, and the response might be different if DIN input were increased without also increasing discharge significantly (e.g., through increased point source loading). During the summer there appears to be no relationship between nutrient loading and algal abundance in either the upper or lower Pamlico (Nixon, 1989).

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STRATIFICATION AND BOTTOM WATER HYPOXIA IN THE PAMLICO RIVER ESTUARY

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We have examined the relationships among bottom water oxygen, vertical stratification, and the factors responsible for stratification-destratification in the Pamlico River Estuary in North Carolina. The study is based primarily on a15-year set of biweekly oxygen, salinity, temperature, and nutrient-concentration measurements, but we also have incorporated some recent continuous-monitoring results.

The Pamlico is a shallow (2.7 m mean depth), oligohaline-mesohaline estuary extending 65 km from Washington, NC to the western edge of Pamlico Sound. The estuary varies in width from about 0.5 km near Washington to about 6.5 km at its mouth. The Pamlico "River" is actually an estuary of the Tar River, which drains most of the 14,000 km^2 basin area. Total freshwater flow into the Pamlico typically ranges between 28 m^3 s^{-1} in October and 112 m^{3}s^{-1} in February (Stanley, 1988; Giese et al., 1979). Freshwater flushing times corresponding to this flow range are estimated to be between 80 and 28 days. Lunar tides in the estuary are almost negligible (seven cm), due to restrictions imposed by the Outer Banks, a chain of barrier islands separating Pamlico Sound from the Atlantic Ocean. However, "wind tides" of 0.5-1.0 m are not uncommon, and are most likely following several days of sustained winds from directions approximately parallel to the estuarine axis (Giese et al., 1979). Prevailing summertime winds in the Pamlico region are from the SW and NE (Pietrafesa et al., 1986).

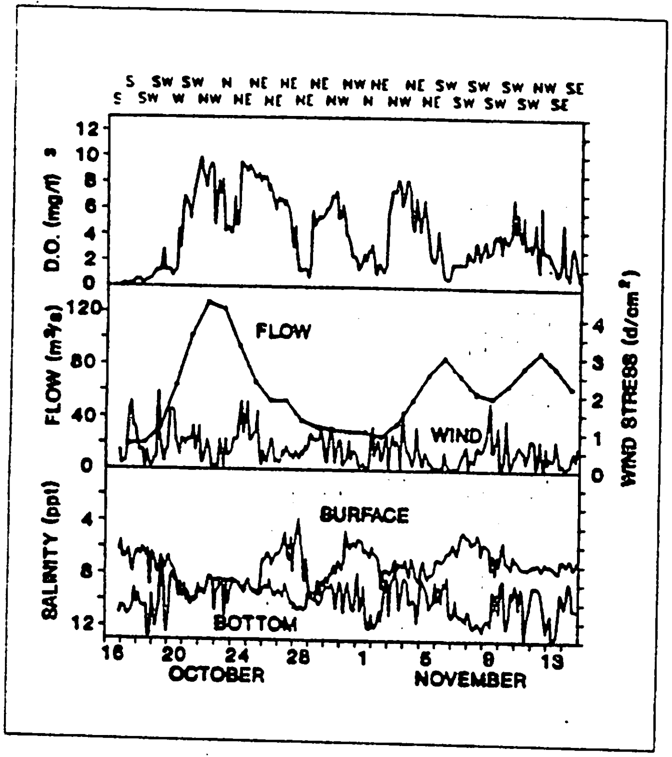
Seasonal salinity patterns in the estuary are set primarily by variation in Tar River flow. Typically, surface salinity is <8 ppt. during the late winter and early spring. The salinity increases to maximum values (10-15 ppt) during fall. However, there is considerable interannual variability. During drought years, the salinity may approach that of Pamlico Sound (20-24 ppt) (Stanley, 1988). Temperatures in the estuary typically range from 4 °C in January to 30 °C in August.

Hypoxia, or "dead water" as it is known locally, has become one of the most important environmental issues for the Pamlico. Hypoxia in the estuary was first documented twenty years ago (Hobbie,1970), and was investigated more thoroughly in the mid-1970s (Davis et al., 1978) but knowledge about it seems to have become widespread only in more recent times. A recurring theme in many newspaper articles, regulatory agency documents, and some of the scientific literature written during the late 1980s is that nutrient inputs promote large blooms of phytoplankton that eventually die, decompose, and contribute in a major way to low oxygen conditions during summer. In addition, most fish kills in the estuary in recent years have been attributed to hypoxia in the bottom waters (N.C. Division of Environmental Management data summarized in Stanley, 1985). Many citizens, and some scientists, suspect that bottom water anoxia and fish kills are more common in the estuary now than in the past.

Seasonal and Spatial Variability. Frequency distribution plots of all measurements made between 1975 and 1989 show a distinct seasonal pattern in Pamlico bottom water oxygen. Concentrations <5 mg l^{-1} are least common in the winter months (0-15 percent) and most common in July (75 percent). About one-third of the July measurements are <1 mg l^{-1}. This pattern is in part a reflection of the effect that annual water temperature and salinity cycles in the estuary have no oxygen solubility. But other factors must be involved, since the percent saturation frequency plot shows the same pattern. Instances of strong undersaturation (<40 percent) are rare in the winter but frequent in the summer months (39-61 percent).

A scatter plot of all the bottom water dissolved oxygen (DO) percent saturations against the corresponding water temperatures, reveals a sharp increase in the probability of hypoxia at temperatures >16°C. Below this temperature, only three percent of the measurements were less than 50 percent saturation: On the other hand, above 16 °C, there were both high and low saturation values. The lack of hypoxia in the winter is not due to a lack of water-column stratification because a frequency plot of delta Sigma-t (bottom-surface) indicates that stratification is even more common in the winter than in the

Figure 1. Tar River surface and bottom salinity, flow, wind stress and direction, and bottom water DO, October 16-November 14, 1989 (three hour intervals).



summer. Thus it appears that a combination of stratification and warm water temperature are required for bottom water hypoxia to develop in the Pamlico.

Severe hypoxia occurs more frequently in the upper half of the estuary than near the mouth. When data for all months are considered, around 15 percent of the upper- and mid-estuary measurements (Stations 10, 8, and 5) give oxygen concentrations <1 mg l^{-1}, while at Station 1 near the mouth only two percent of the values are below 1 mg l^{-1}. The percent saturations also show a spatial difference. From 18 to 23 percent of samples from the upper- and mid-estuary stations are less than 20 percent saturated, compared to only four percent at Station 1. A similar analysis of data from the summer months (June–September) shows that even though the frequency of low oxygen increases during warm weather, the spatial pattern does not change; i.e., low oxygen is still most common in the upper regions of the estuary. Concentrations less than 1 mg l^{-1} occur in one-third of the samples from the upper estuary, but in only four percent of the samples from near the mouth. The percent saturation data show the same pattern.

Short-term Variability. Unfortunately, the long-term monitoring data provide little insight into the short-term dynamics of stratification and hypoxia in the Pamlico, due to the relatively long sampling interval (two to three weeks). But data from a 1989 continuous monitoring study show that stratification/hypoxia events can develop and break down very rapidly. These data also strongly suggest that wind and freshwater flow into the estuary are important factors influencing the timing of these events.

One example of a continuous monitoring sequence, spanning the period mid-October to mid-November, 1989, is presented in Figure 1. This sequence includes large, short-term fluctuations in Tar River discharge and wind stress, which interacted to produce four distinct episodes of stratification. The first was in progress at the beginning of the sequence on 16 October. Tar River flow had declined from a previous peak to 20 m^3 s^{-1}, winds were blowing slowly from the south, and there was a six ppt difference between surface and bottom salinities. Also, bottom water DO was extremely low–well below 1 mg l^{-1}. The next day, a strong afternoon wind from the south eroded the salinity gradient, but was not sufficient to destroy it. Even stronger winds on the 19th temporarily broke up the gradient, and finally on the 20th it was destroyed following a third day of strong afternoon wind. At this time, the bottom water DO rose dramatically, reaching saturation concentration (9 mg l^{-1}) by 21 October. Subsiding winds on the 22nd and 23rd led to brief periods of stratification and lowered DO.

Meanwhile, in response to widespread precipitation over the Tar basin, a flow pulse had been building steadily for about four days, reaching a peak of 125 m^3 s^{-1} at Tarboro on 22 October. That pulse reached the estuary station three days layer, quickly reducing the surface salinity to five ppt, and setting up the second stratification event, which eventually amounted to a five ppt vertical gradient. Bottom water DO fell rapidly from 6 mg l^{-1} on 27 October to around 1 mg l^{-1} the following day. This seems to be a clear example of stratification caused by a moderate pulse of freshwater spreading out over the estuary surface under low wind-stress conditions. In addition, encroachment of saline Sound water, as evidenced by the slowly increasing bottom salinity, strengthened the density gradient even more. On the 28th, both the passing of the Tar River pulse and increasing wind stress combined to turn the water column over in a matter of a few hours during the evening.

Within 48 hours, another stratification event had begun to develop (30 October). This time, winds switched from the northeast to the northwest, and decreased in velocity. This event lasted about four days, with a vertical salinity gradient of about four to six ppt and bottom water DO reduced to around two mg l^{-1}. It ended late on 2 October following increased wind stress the previous night. The fourth episode began almost immediately, and for the next three days (4-6 November), there was weak stratification that was nearly broken on several occasions, but apparently did not completely disappear, since the bottom water DO continued to fall, reaching 1 mg l^{-1} on the sixth. The vertical salinity gradient strengthened on the next day, weakend on the ninth following stronger winds, and fluctuated between two and six ppt for the remainder of the sampling period. Bottom DO also fluctuated, mostly between 2 and 4 mg l^{-1}. In summary, this data suggest that stratification events and bottom water oxygen levels are tightly coupled with variations in freshwater discharge and wind stress. Stratification can form or be broken in a matter of hours, and episodes lasting from one to several days seem to be common.

Spearman Correlation Results. Results of Spearman Rank Correlation analyses tended to corroborate conclusions drawn from the frequency plots and the continuous monitoring data. Several variables were tested for correlation with bottom water DO concentration at each of the four long-term monitoring stations. Delta Sigma-t (bottom-surface), gave the highest correlation coefficient. The oxygen vs. delta Sigma-t relationship was inverse and was strongest at the three stations farthest up the estuary. The only physical variable showing a significant positive correlation to bottom water DO was wind stress lagged by one day, another indication of the rapidity with which stratification events are established and broken up. Tar River discharge, lagged 5, 10, or 15 days, seemed to be less important, as the only significant combination was the five-day lagged flow at Station 1.

Additional Spearman analyses were made to test for associations between delta Sigma-t, and two factors that could influence the strength of the stratification–Tar River flow and wind stress. Flows were lagged 0, 5, 10, and 15 days, and wind stress was lagged 0,1, and 2 days. The computed correlation coefficients between flow and delta Sigma-t were significant ($p<.05$) only for a station at the upper end of the estuary. There is a curious trend in the flow vs. delta Sigma-t coefficients, from negative in the upper estuary to increasingly positive at the lower station. This result could be interpreted to be a result of the salt wedge moving up and down the estuary in response to the strength of the mixing force exerted by freshwater inflow. Wind stress was significantly correlated with stratification at all stations when the previous day's wind was considered. In addition, the strength of these correlations trended upward toward the lower end of the estuary. This seems logical, since the shape and orientation of the Pamlico is such that fetch over which the prevailing southwest and northeast winds blow increases toward the mouth.

Event Frequency. Using hourly wind measurements collected during the summers of 1980 to 1985, we calculated the resultant daily vectors of the axial (along the channel, 295° NW or 115° SE) and coaxial (cross-channel, 25° NE or 205° SW) components of the relative wind stress on the Pamlico. Strong cross-channel and axial wind events occurred with frequencies such that on average, a vertical mixing and reoxygenation of the bottom water occurred approximately every 8.6 days during June, every 11.5 days during July, every 12.4 days during August, and every 6.5 days during September.

There is evidence that at this frequency of reoxygenation, oxygen demand by the sediments is not sufficient to lead to hypoxia or anoxia. The sediments could lower the oxygen content of the bottom water only by some 0.13 mg l^{-1} d^{-1}. At this rate, the total oxygen consumed by the sediments during the longest average interval between strong wind events (12.4 days in August) would lower the concentration by about 1.6 mg l^{-1}. Respiration by plankton and bacteria in the water appears to be much greater. Rates of water column respiration are 4.6 to 21.5 times greater than the five-day oxygen uptake by the sediments and are sufficiently great that hypoxia and anoxia could easily result if the water was only mixed every 6.5 to 12.4 days. It seems clear that it is the balance between water-column oxygen uptake and the frequency of strong wind events that largely determines the spatial extent and duration of the low-oxygen problem in the bottom waters of the Pamlico.

Effects of Hypoxia on Pamlico Biota. Short-term effects were documented in the Pamlico during the late 1960s by Tenore (1972), who found that macrobenthos in deeper waters of the estuary had low species diversity and density in the summer, and that variations in the density were correlated positively with anoxia/hypoxia. Large kills of the benthos occurred quickly in the affected areas following the onset of hypoxia. However, these areas were recolonized by the following winter. Data obtained from the North Carolina Division of Environmental Management show that low DO was suspected to be the cause of most fish kills investigated in the Pamlico during the past two decades (Stanley,1985; N.C. DNRCD, unpublished data). Most of the reported kills were not in the main stem of the estuary, but rather near the heads of relatively small tributary creeks. Menhaden were the species involved in most episodes, and the great majority of the kills were reported during the summer. In some cases, dissolved oxygen was measured and found to be low in the kill vicinity; in other instances low DO was inferred from circumstantial evidence (e.g., "sulfide-like odors") (Stanley, 1985).

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EUTROPHICATION OF MACROPHYTE-DOMINATED SYSTEMS

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Shallow coastal waters are among the most enriched ecosystems in the world, and most of the added nutrients are derived from human activities in the watersheds. In our studies of coastal bays in Massachusetts, we can show that nutrient accretion by forests takes up most of the nutrients provided to the watershed surface via precipitation and fertilizer use. On the other hand, septic tanks inject nutrients below the subsoil, and their nutrients find their way to the groundwater below. Although procedures to calculate loading rates to receiving waters are available, we need further information on whether there is attenuation of nutrient concentrations or transformations during transport through the aquifer and near-shore sediments. This information is essential before we can translate our findings into recommendations for managing land use in coastal watersheds.

Our work in Waquoit Bay shows some consequences of nutrient loadings from watersheds: a) nutrient and chlorophyll-a content of near-shore waters is somewhat elevated; b) macroalgal biomass on the bay bottom is greatly increased (macrophyte biomass increases three to four times in urbanized watersheds compared to non-urban watersheds); c) eelgrass biomass and cover decreased; and d) abundance and species richness of benthic invertebrates decreased.

The key change has been the dominance of the macroalgae; their biomass is sufficient to hold an amount of nitrogen as large or larger than the annual nitrogen loadings. Their biomass is also large enough that oxygen profiles are controlled by macroalgal photosynthesis and respiration. One consequence of this is that after a series of cloudy days photosynthesis does not restore oxygen consumed by respiration, and anoxic conditions result. The increased frequency of anoxic events, plus the disappearance of eelgrass combine to create the lower abundance of animals on the bay bottom. The Waquoit Bay ecosystem shows evidence of "bottom-up" controls; nutrient loading has repercussions throughout the entire food chain. The state of the coastal system greatly depends on what takes place in the watershed, since human activities there are the principal causes of changes in the aquatic ecosystem.

ANOXIA AND HYPOXIA IN LONG ISLAND SOUND, CHESAPEAKE BAY, AND MOBILE BAY: A COMPARATIVE ASSESSMENT

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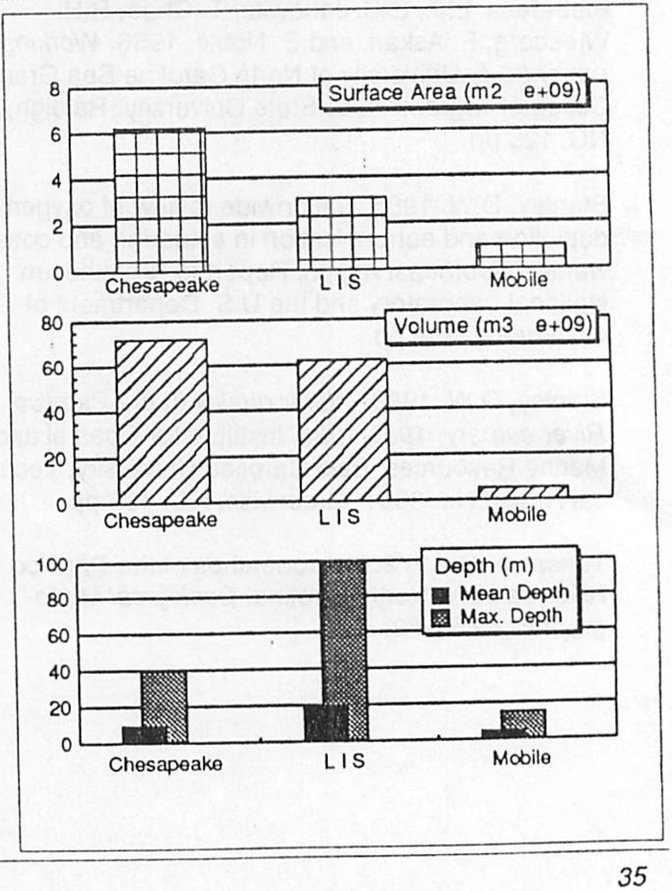
Introduction. The objectives of this paper are: (1) to compare the occurrence and persistence of hypoxia in Long Island Sound with that of Chesapeake Bay and Mobile Bay, and (2) to suggest some quantitative approaches for comparing and predicting the susceptibility of coastal systems to hypoxic development.

Eutrophication has resulted in seasonal oxygen depletion events which coincide with seasonal physical stratification in each of the three systems. There are differences, however, in the frequency, intensity, spatial and temporal patterns, and biological consequences of the oxygen deficit. This comparison was initiated because seasonal oxygen depletion in Long Island Sound (LIS) was found to be much more extensive and persistent than would be expected considering that, compared with Chesapeake Bay and Mobile Bay: (1) LIS lies well to the north along the latitudinal gradient, where summer water temperatures are relatively cool so that physical saturation levels for oxygen will be relatively high and respiratory demand should be relatively low; (2) LIS has a more robust tidal regime than the other two systems, and tidal turbulence would be expected to help maintain more turbulent mixing between the upper and lower strata. In subsystems of the lower Chesapeake, such as the York River, tidal mixing completely destratifies the system on a forthnightly cycle; (3) LIS is more weakly stratified than Chesapeake Bay or Mobile Bay, and thus was expected to be more susceptible to periodic mixing by wind as well as tides; and (4) LIS has a relatively much larger mass of deeper, cooler, water than Chesapeake Bay and Mobile Bay, which was expected to provide a large reservoir of oxygen to resupply seasonally-heightened respiratory demands below the pycnocline.

The major condition associated with the bottom water oxygen depletion is stratification of the water column, which physically isolates the bottom layer from the surface layer where oxygen is being supplied through plant production and atmospheric mixing. The degree of isolation depends on the balance between the strength of the buoyancy force (density stratification) resulting from vertical temperature and salinity gradients which stratify the system, and the strength of turbulent mixing forces, including vertical mixing by wind and current shear and lateral advection. The magnitude of depletion also depends on the magnitude of biological oxygen demand within the isolated water column and sediments. This paper compares characteristics of oxygen depletion events in these three systems with respect to their geography and geology and the strength, persistence, and seasonality of the pycnocline. It then discusses how the structure of the biological community may be responding to eutrophication and contributing to hypoxic and anoxic conditions.

Since my field experience has centered on conditions in Long Island Sound, that system provides the focal point for my remarks. Geographically, LIS is the most northern of the three systems. It has the coolest summer water temperatures conveying the advantage of higher saturation concentrations for oxygen and potentially lower respiratory rates. The main axis of LIS is oriented west-to-east, which provides a short fetch during prevailing southwesterly winds in summer, and thus makes it less susceptible to wind-mixing than Chesapeake Bay and Mobile Bay, where the main axes are north-to-south.

Figure 1. Comparison of the physical and hydrologic characteristics of Chesapeake Bay, Long Island Sound, and Mobile Bay.



With respect to area, these three systems form a fairly regular series with Long Island Sound forming the intermediate member (Figure 1). With respect to volume, Long Island Sound is closer to Chesapeake Bay than to Mobile Bay, and with respect to mean and maximum depth, LIS is clearly the deepest and Mobile Bay is very much shallower than the other two. This depth relationship becomes important when considering the stability of the pycnocline and the volume of water susceptible to depletion.

The importance of these relationships to living marine resources can be demonstrated by a cartoon using a cross-section in the approximate proportions of western LIS (Figure 2). With the pycnocline at 10 m, 75 percent of available benthic habitat lies in subpycnocline water and is susceptible to depletion, leaving only 25 percent on shallow, more oxygenated flanks. In terms of volume, 46 percent will be subpycnoclinic, and this proportion increases to 65 percent when considering only the waters beyond the 10-m depth contour.

Systems Compared. In 1987 hypoxic and anoxic bottom water covered about one third of the total area of Long Island Sound. Most severe conditions occurred in the far western Sound, where most of the sewage treatment plant discharges are located, and where the Sound is narrower and shallower so the volume of receiving water is relatively low and converging shorelines reduce effective wind mixing. The oxygen depletion was extensive and severe, despite the relatively weak physical stratification in this area. The Sound is not a typical coastal plain estuary like Chesapeake and Mobile Bays. The major fresh water inflows are the Housatonic, Connecticut and Thames rivers which lie along the northern shore, well down-estuary from the head. The East River, which enters at the western end, is a tidal strait which delivers saline water only partially freshened by the Hudson Raritan system. The vertical salinity gradient is extremely weak. Thus density stratification in the Sound is determined by the strength of the thermocline rather than the halocline, which controls stratification in Chesapeake and Mobile bays. The central axis is broad and deep throughout most of the Sound and, with the pycnocline located at six to 10 m below the surface, there is little flanking shoal area, as demonstrated by the two cross-sections. Thus a large proportion of water volume and benthic habitat lies below the pycnocline and is susceptible to oxygen-depletion events.

A contour plot of oxygen distribution along the central axis from Throgs Neck (East River) eastward (Figure 3) demonstrates the thickness of the depleted layer and the uniformly low oxygen below the pycnocline. This relative uniformity from the pynocline down, regardless of water depth, is evidence that oxygen uptake by the water column is an important factor in the drawdown and maintenance of the oxygen deficit during stratified conditions. Because of the shape of the basin, hypoxic conditions in the subpycnocline layer in LIS extend nearly shore-to-shore and involve a large volume of water. Some degree of cross-channel tilting has been observed in Long Island

Figure 2. Proportional cross-section of western Long Island Sound, showing phytoplankton respiration and decay susceptibility below the pycnocline.

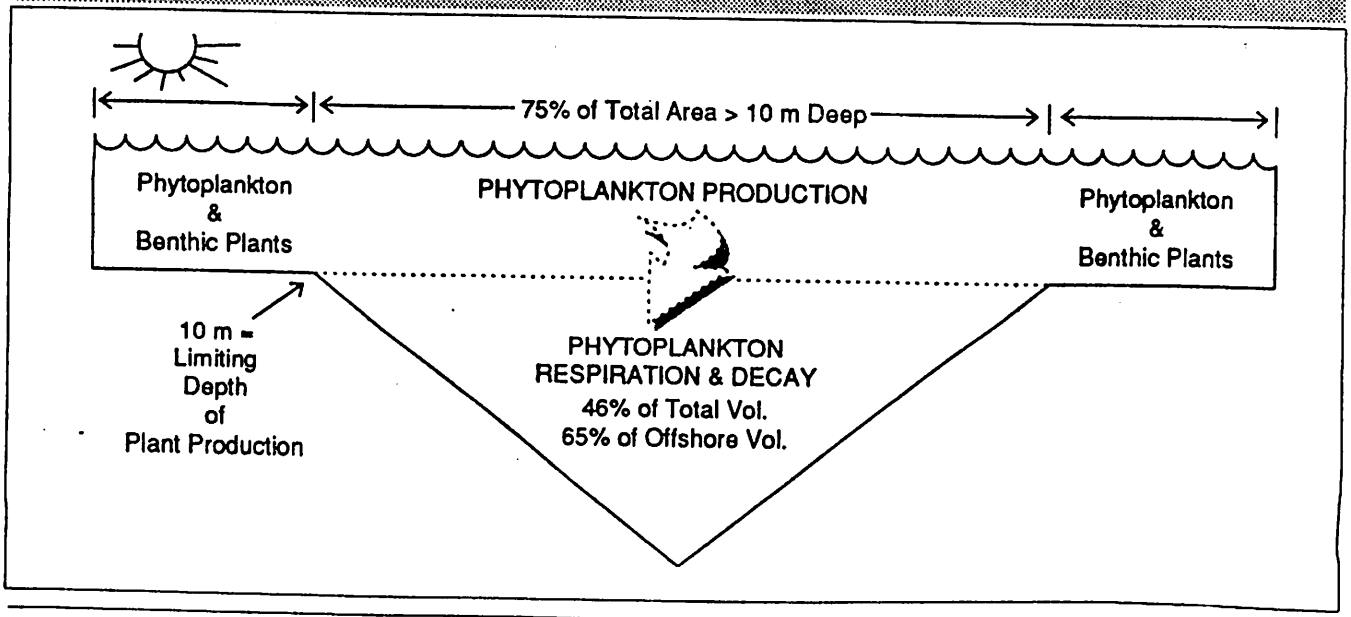
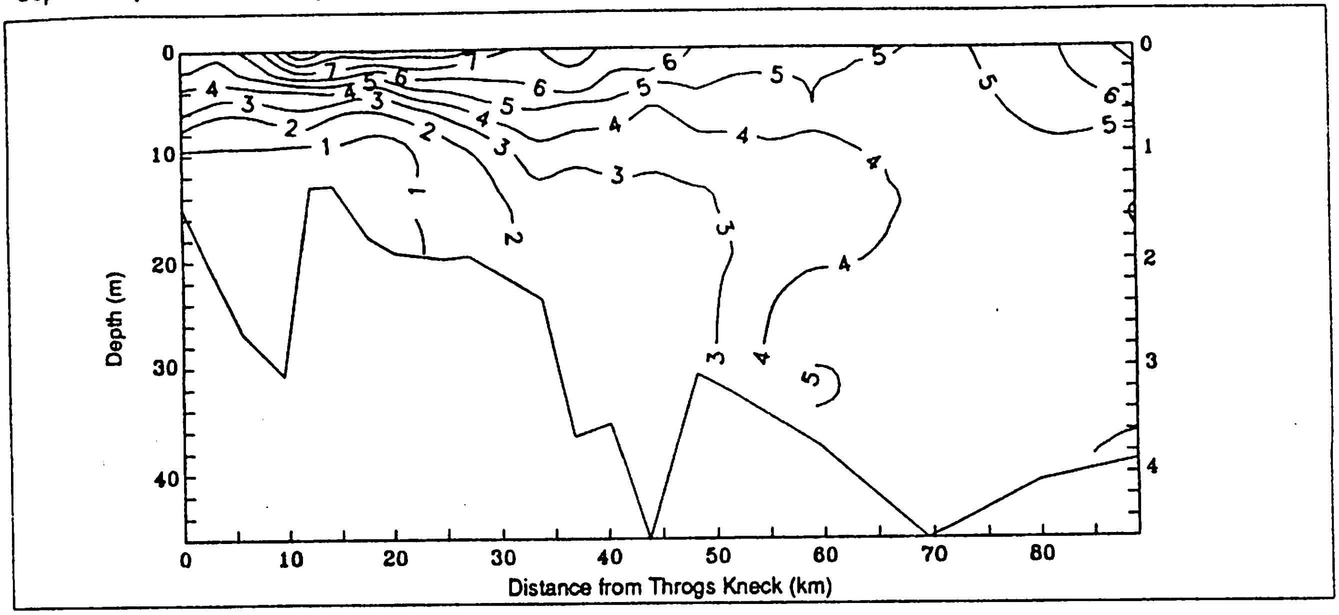


Figure 3. Contour plot of oxygen distribution along the central axis from Throgs Kneck (East River) eastward, showing depleted layer and uniformily low oxygen below the pycnocline.



Sound, but not to the degree reported by Tom Malone for the Chesapeake Bay. A rough calculation of the Kelvin Number for Long Island Sound indicates that the geomorphology of the western Sound renders it relatively unsusceptible to cross-channel tilting events.

Chesapeake Bay is a classical coastal plain estuary with its major freshwater inflow from the Susquehanna River at its head. Stratification within the Bay is strongly dominated by this freshwater inflow. Seasonal thermocline development adds, at most, about 20 percent of the buoyancy force. Seasonal hypoxia and anoxia occur in the deep central axis of the midsection of the Bay. A cross-section at Smith Point shows the relatively large proportion of flanking shoals relative to Long Island Sound. An axial profile (Figure 4) shows that the oxygen deficits develop within the deeper basins of the central axis, which occupy the midportion of the Bay.

The effects of pycnocline tilting on flanking benthic habitats were first described for Chesapeake Bay by Tom Malone. The geomorphology of the Bay makes it more susceptible than Long Island Sound to such events, and comparing the cross-sections, much larger areas of benthic habitat can be affected. Tilting events, which occur in response to wind or current oscillations, are therefore probably much more devastating to living marine resources in the Chesapeake Bay than they are in Long Island Sound. Simple calculations of Kelvin Number may provide a reasonably efficient parameter for comparing the susceptibility of systems to the tilting phenomenon.

Mobile Bay is an extremely shallow system compared with Chesapeake Bay and Long Island Sound. With most of the Bay, except the narrow ship channel, less than six meters deep, it is extremely susceptible to wind mixing. Over 20 percent of the nation's freshwater empties into the sea through the Mobile River system at the head of the Bay. Thus Mobile Bay is not only the most strongly stratified of the three systems, but its entire volume may be replaced within a few days by a single runoff event. The diel tidal cycle is extremely weak, and circulation is mostly controlled by winds and freshwater runoff. Hypoxia and anoxia occur as short-term events in masses of water that are moved about within the Bay but rarely occupy large proportions of the Bay at any one time. This pattern, controlled largely by meteorological events, differs substantially from the seasonally cumulative nature of the oxygen deficit in the relatively large masses of isolated deep water of Chesapeake Bay and Long Island Sound. The pattern in Mobile Bay is more similar to that in the Pamlico River estuary system.

Fewer records of oxygen depletion are available for Mobile Bay than for Chesapeake Bay and Long Island Sound. Those which do exist show more severe conditions in the upper Bay. There is sketchy documentation of the frequency and duration of events in Mobile Bay. This comparison demonstrates that not only strength, but persistence must be considered with respect to the pycnocline. Mobile Bay has the strongest pycnocline of the three systems, but its effects are modified because it is much less persistent.

The three systems, Long Island Sound, Chesapeake Bay, and Mobile Bay, constitute a series with decreasing proportions of water volume and benthic habitat below the pycnocline and increasing pycnocline strength. The large proportion of deep water in Long Island Sound undoubtedly contributes to the relative stability of the water column and persistence of stratification despite the relative weakness of the pycnocline.

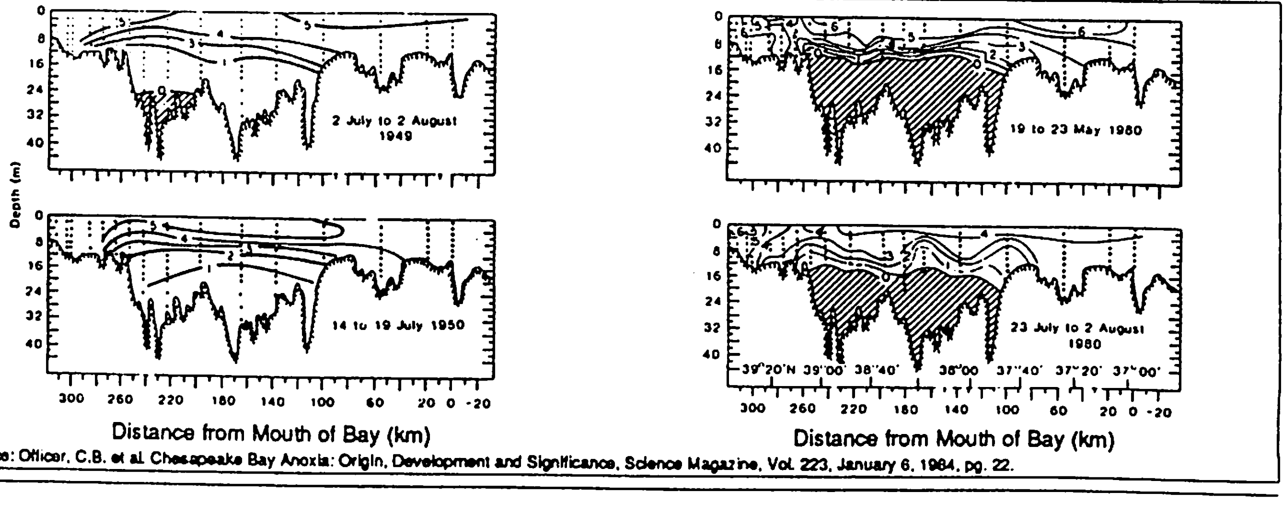
Differences in the seasonality of pycnocline development between systems influence the duration and intensity of the deficits. In Long Island Sound, permanent seasonal stratification begins about mid-June, when surface waters warm sufficiently to produce a seasonally permanent, thermally-controlled pycnocline. The rate of heating of surface waters outpaces that of bottom waters through early August, and during this period the pycnocline is persistent, even through heavy winds. Then, as nighttime cooling increases, the rate of warming of surface waters slows while that of bottom waters remains fairly constant. The vertical gradient (buoyancy force) gradually becomes reduced to the point where it can be overcome by the shear forces of wind and currents. Sandwiched between the cooling air and warming bottom waters, the narrow surface stratum mixes very rapidly. The timing of destratification is fairly consistent year-to-year.

Permanent drawdown of oxygen in bottom waters begins with thermal stratification, and the deficit accumulates in relatively linear fashion to a low in early August. Reventilation coincides with destratification and is relatively abrupt. Since the permanent drawdown does not begin until mid-June, it is questionable whether the spring phytoplankton bloom could have much influence on the oxygen demands creating the deficit, since most of the labile carbon from phytoplankton settling to the bottom in January or February would be expected to be remineralized before mid-June. It also appears that the severity of the deficit is a function of length of the stratified season. If the season is short enough so that bottom waters remain hypoxic, the effect on organisms is oxygen deprivation, which allows for adaptive behavior and should be less devastating on living marine resources. If the season is long enough so that the system becomes anoxic, the effect is hydrogen sulfide toxicity, an immediately lethal condition.

Stratification in Chesapeake Bay is controlled mainly by the salinity gradient, beginning with spring runoff from the Susquehanna River and ending with increased wind mixing around the fall equinox. Hypoxic development corresponds directly with stratification. The season is not only several months longer than that in Long Island Sound, but starts early enough for the winter-spring phytoplankton bloom to contribute heavily to a steep decline of oxygen at the beginning of the season. Thus Chesapeake Bay does become anoxic for extended periods in summer. The volume of depleted water is very large, and during anoxic periods reduced metabolites may accumulate so that chemical oxygen demand may combine with biological oxygen demand to quickly use up oxygen provided by episodic reventilation.

Importance of Water Column Processes To Oxygen Demand and Biota. In Long Island Sound, physical stratification of the water column leads to vertical structuring of metabolic processes, which contributes to the development and maintenance of hypoxia. During calm conditions, which prevail for extended

Figure 4. Dissolved oxygen levels (milliliters/liter) along the main channel of Chesapeake Bay in summer 1949 and 1950, and summer 1980. Dots indicate measurement locations.



Source: Officer, C.B. et al. Chesapeake Bay Anoxia: Origin, Development and Significance, Science Magazine, Vol. 223, January 6, 1984, pg. 22.

periods in summer, the pycnocline is broad and gradual, with sigma-t increasing in nearly linear fashion form near the surface to its lower inflection point, below which it is constant to the bottom. This shape indicates that there is little or no vertical mixing within the upper water column. Oxygen concentrations are characteristically greatest just below the surface, dropping off very rapidly, and frequently exhibiting a zone of minimum values near or slightly above the lower inflection point of the pycnocline. Below that, concentrations are homogeneously low to the sediment surface.

These calm-weather oxygen profiles reflect vertical metabolic structure in the water column. Net oxygen production is limited to a narrow surface layer of three to four meters. Below that is a zone of maximum net oxygen uptake coinciding with the oxygen minimum in the concentration profile. Below the pycnocline, oxygen uptake is low, but the volume of water is large, so that water column uptake accounts for 80 percent or more of total drawdown. The zone of maximum uptake is strategically located between that of net production and that of oxygen depletion, removing oxygen being dispersed downward from above and reinforcing the pycnocline as a barrier to reventilation of bottom water. Moreover, this metabolic pattern indicates that considerable remineralization must be occurring above the pycnocline, where it can be recycled to the producers.

Although wind mixing is ineffective in destratifying the entire water column in Long Island Sound, it is effective in mixing the surface layers. Sigma-t profiles then become homogeneous in the upper layer as in the lower layer, resulting in a narrow, sharp pycnocline. The shape of the oxycline mirrors that of the pycnocline as physical mixing masks the effects of biological processing. Presumably, microbiota and regenerated nutrients are similarly mixed within the upper layers.

The processes involved are better seen using gross production and respiration curves. The bulge of maximum net respiration does not result from larger concentrations of heterotrophs at that level, but rather from changes in the balance between total (gross) production and total respiration with depth. Direct count of bacteria showed that their vertical distributions follow the curve for gross production very closely. It is reasonable to find such tightly coupled distributions because bacteria are able to utilize the DOC produced by phytoplankton. The lower layer here should not show zero, but simply low oxygen uptake.

The correlation between oxygen consumption and bacterial abundance for stratified periods in August 1989 and 1990 supports the hypothesis that much of the oxygen uptake is associated with the bacterioplankton community. These data were obtained by David Drapeau for his MS thesis at the University of Connecticut.

A weaker but still significant correlation was obtained using data from June, July, and August 1989 and 1990. There is agreement of the slope of these data with that of the August data alone (data are from D. Drapeau).

Data from John Tuttle indicates a tight correspondence between respiration and bacterial abundance for a number of studies made year-round in Chesapeake Bay. These data and the Long Island Sound data are in remarkable agreement with respect to range of bacterial numbers as well as the actual slopes of the regressions.

Bacterioplankton are undoubtedly an important component in the pathway of organic cycling from phytoplankton, especially in their utilization of dissolved organic carbon and their tight spatial and temporal coupling with the primary producers so that nutrients are cycled within the upper water column. The extent to which they replace macrozooplankton as a trophic pathway and the extent to which they provide a link between phytoplankton and particle-feeding plankton is unknown. Some researchers regard them as a sink, transforming excess organic carbon to carbon dioxide through high respiration rates. Shunting carbon away from the macrofaunal pathway would presumably lower fish production.

The susceptibility of eutrophied systems to develop strong bacterioplankton pathways is logical, considering the large inputs of dissolved organic matter along with nutrients. Nutrient additions stimulate increased phytoplankton production as a primary effect, with direct secondary effects on the bacterioplankton community. Organic additions stimulate increased bacterioplankton as a primary effect, with direct secondary effects on the phytoplankton community. With STP inflows, these cycles are stimulated through both pathways, which might explain the extremely rapid response within the systems and the high respiration rates in the upper water column and down through the pycnocline. The good news is, if most of the response is in the water column, the effects of cleanup may be more rapid than if the loadings are deposited to the sediments and sequestered there for future remobilization.

Conclusions. Basic conditions for the development of bottom water oxygen deficits are isolation of the water column by physical stratification and biological oxygen demand. Geomorphology of the basin is a major factor in the persistence of isolation events, which, in turn, determines the temporal characteristics of hypoxic events. Shallow systems are dominated by meteorological events. Their stratified periods are short-term, episodic, and stochastic. Hypoxic conditions tend to be temporally intermittent and spatially patchy. Examples are Mobile Bay and Pamlico Sound (described in an earlier presentation). Deep systems are dominated by cyclic seasonal events. Their stratified periods are long-term and predictable. Hypoxic conditions are cumulative, persistent, and spatially and temporally continuous. Examples are Long Island Sound and Chesapeake Bay. The above dichotomy represents two points on a continuum. Examples of intermediate conditions might be found in the sub-estuaries of the lower Chesapeake Bay, not covered in this paper. The York River, for instance, which is destratified fortnightly on spring tides, represents a system with predictable episodic stratification where hypoxic conditions are cumulative but intermittent and spatially and temporally continuous with the period of stratification.

Within these broad characterizations, however, particular combinations of factors can have modifying effects. Of the two deepwater systems, Chesapeake Bay has a much stronger pycnocline but Long Island Sound is apparently more stable, probably due to the geomorphology of its basin which is deeper, oriented at right angles to prevailing winds, and has width and depth dimensions making it less susceptible to lateral tilting.

The intensity of the deficit is related to combinations of factors associated with the duration of isolation and biological oxygen demand. The longer the isolation, the more susceptible the system will be to extreme deficits and exceeding its anoxic threshold. The seasonality of the deficit will affect (1) the temperature-dependent saturation concentration and presumably the initial oxygen content of the isolated water, and (2) the extent to which drawdown is fueled by spring phytoplankton blooms.

Other characteristics of biological oxygen demand are important but their controlling factors are more speculative at this point. In Long Island Sound, development of vertical metabolic structure, especially during calm conditions, reinforces the barrier to downward diffusion and accounts for 80 percent or more of the drawdown in the lower water column. It appears that similar conditions hold for Chesapeake Bay. In systems dominated by water-column oxygen demand, larger subpycnoclinic water volumes would not be expected to provide greater buffering capacity for an area-based deficit. In systems dominated by sediment oxygen demand, larger subpycnoclinic water volumes may increase the buffering capacity against hypoxia. Sediment oxygen demand would be expected to dominate in shallower systems.

It appears that development of strong bacterioplankton pathways represents a system-level adjustment to nutrient and dissolved organic carbon loadings. This response may be a factor in shifting the dominance of oxygen uptake to the water column. It forms a short-cut for relieving the system of excess organic carbon, but it also perpetuates organic production by conserving and recycling nutrients within the upper water layer. Such a shift may inhibit macrofaunal pathways either directly through trophic redirection or indirectly through habitat exclusion as a result of exacerbated oxygen deficits or hydrogen sulfide toxicity.

THE MANAGEMENT OF EUTROPHICATION THROUGH AQUACULTURE AND NATURAL BEDS OF MARINE ALGAE

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The major primary producers in marine ecosystems are salt marsh grasses, seagrasses, phytoplankton, benthic macroalgae (seaweeds), and benthic microalgae. Nutrient additions to coastal ecosystems bring about changes in these communities. Of these primary producers, the seaweeds are an underutilized resource where intervention into eutrophication is possible. Macroalgal populations are major sinks of nutrients, are harvestable, and their harvest would least impact coastal ecosystems. The utilization of natural beds of seaweeds seems to present a unique opportunity for environmental management to enhance environmental quality by improving estuarine habitats for fish, invertebrates, infauna, epifauna, and waterfowl in these estuaries.

When shorelines experience eutrophication, certain algae greatly increase in abundance. Blooms of these "opportunistic algae" species, chiefly *Ulva*, *Enteromorpha*, *Cladophora*, *Agardhiella*, *Gracilaria*, and *Codium*, profoundly affect natural systems, overgrowing established plant and animal assemblages. Shorelines shift from diverse, stable communities to near monocultures. Blooms often occur to such a degree that the algal biomass becomes a significant problem, either through its direct fouling effects, or as a result of hypoxia produced from its eventual death and decay.

In many places direct action has been necessary to reduce or eliminate the damaging impact of other opportunistic algae. Such action typically consists of harvesting, or otherwise removing the offending macroalgal biomass. This type of harvesting has reached major proportions in some places, for example, Venice Lagoon where amounts of *Ulva* in excess of 200 tons per day are removed over extended periods through spring, summer and fall seasons. The optimal harvest of biomass is dependent on a working knowledge of the alga's physiology and ecology; and the history and current levels of pollutants. For example, analysis of plant tissue, when done in conjunction with growth analysis, allows one to determine a species' critical nutrient concentration. Information on internal supplies becomes a valuable management tool for predicting and preventing blooms, and providing critical information for developing an optimal harvesting strategy. The harvesting of macroalgae to remove pollutants is an ongoing part of an integrated management plan for a bay in Japan and, most recently, in the aforementioned Venician lagoon.

It is unfortunate that the algae typically associated with macroalgal blooms are ones for which there exists little market potential, for if these seaweeds had intrinsic value, their removal and disposal could be the basis of a viable maritime industry. Nevertheless, in Israeli fish ponds *Ulva*, is now used to remove all ammonia from fish pond effluents. The *Ulva* is harvested and used as a filler in sausage meats. The removal cost of macroalgae may also be mitigated by useful disposal of the harvested seaweed for fertilizers, animal food, or biomass substrates for methanogenic bacteria. Unfortunately, these uses have only limited market potential. Fortunately, however, it may be possible to introduce more valuable species into problem areas through aquaculture methods.

On a worldwide basis, many different species of macroalgae are cultivated. Most of the types cultivated successfully on a commercial scale are those used for human food, particularly in Asia. The best known examples include the red alga *Porphyra* ("nori"), and the brown alga *Laminaria* ("kombu"). Other species have been cultivated as sources for valuable extractive products, e.g., the red algae *Eucheuma* for carrageenan and *Gracilaria* for agar. A key consideration in most successful examples of seaweed farming is the selection and optimization of suitable habitats which have enough substrate for macroalgal attachment. In Asia, seaweed farm grounds are most often in areas where the same species would not otherwise occur due to the lack of suitable substrate materials at the appropriate water depth.

There is a widespread need for active intervention in eutrophicated marine ecosystems directly affected by nonpoint sources. A significant contribution to such efforts can be made by managing natural populations or through aquaculture techniques. By cultivating species with intrinsic economic value, the great costs associated with harvest and removal of undesirable species can be reduced or eliminated. The nutrients absorbed by the cultivated algae are removed from the ecosystem through harvest. An additional side benefit of such actions is the development of a new economic base for coastal fishing communities. When coupled with efforts at long-term source reduction, it should be possible to recover highly eutrophicated ecosystems.