This issue of the *Jersey ShoreLine* features one of New Jersey’s most important watersheds, Barnegat Bay. The state’s largest estuary, Barnegat Bay stretches from Little Egg Harbor Inlet to the Point Pleasant Canal, covering more than 660 square miles, and is a critical spawning and nursery ground for two-thirds of the fin and shellfish we eat. Its marshes and other vegetation protect marine life by filtering sediments and pollution and act as barriers against floods and storm damage. Barnegat Bay also provides major recreational, economic, and aesthetic benefits to the half-million people who live within the watershed, and the half-million seasonal visitors who vacation in the coastal communities that border the Bay. These benefits depend directly on the vitality of its ecosystem.

Because it is a coastal lagoon with a relatively large volume and few inlets, Barnegat Bay is characterized by poor flushing and long residence time of its water. As a result, the watershed is highly susceptible to environmental degradation. The State’s Comprehensive Conservation Management Plan (CCMP) for Barnegat Bay has recognized that information is lacking about basic estuarine processes and the cumulative environmental effects of pollutant loadings on these processes. As a means of meeting its objective of restoring and maintaining a productive ecosystem with no adverse effects due to pollution, the CCMP has made integrating scientific data to prioritize the focal issues of point and non-point sources of pollution, habitat loss/open space, water quality degradation, and the multiple interests in the watershed region one of the guiding principles of its Mission Statement.

In the last decade, the *New Jersey Sea Grant College Program* (NJSGCP) has dedicated nearly $1.5 million in funding to support projects that impact the Barnegat Bay watershed. Current research involves six projects directly related to water quality and habitat issues prioritized by the CCMP. The combined studies are integrated to address: residence time and subtidal circulation of the Bay; tidal exchange at the Barnegat Inlet and nutrient flux; atmospheric deposition to the watershed and directly to the Bay; tidal flows and inlet exchange as it affects recruitment of important shellfish; and impacts of chlorinated outfalls in adjacent ocean waters that may affect the Bay.

Other *NJSGCP* Barnegat Bay initiatives have included outreach programs like the Barnegat Bay Watch Monitoring Program launched in 1991, which provided ongoing monitoring of ecological conditions in the Bay, including episodic events and trends in habitat changes and water quality. Data collected by hundreds of volunteers provide a constant flow of information about the Bay’s health. More recent outreach projects involved partnering with the New Jersey Department of Environmental Protection to produce a public education fact sheet on Brown Tide in Barnegat Bay and working with various local and state agencies to create a *Boater’s Guide to Barnegat Bay and Little Egg Harbor*, designed to educate boaters about their responsibility to the watershed’s fragile ecosystem. The guide is part of a three-phase package that will include a companion video and teacher curriculum.

*NJSGCP* research and outreach efforts have already fostered a better understanding of the Bay and heightened awareness about the importance of sustaining this valuable natural resource. Ultimately, they will make an important contribution to the state’s Comprehensive Conservation Management Plan, with regard to the decision-making, policy planning, and public stewardship that will be critical to protecting the future of Barnegat Bay.

Michael P. Weinstein, Ph.D.
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As a result of brown tide blooms that occurred in 1999 and again this year in Barnegat Bay and Little Egg Harbor, the NJ Department of Environmental Protection is conducting the Brown Tide Assessment Project. The U.S. EPA and the NJDEP’s Bureau of Marine Water Monitoring are collecting water samples from 44 stations along the coast. For the first time, enumeration of the brown tide organism, *Aureococcus anophagefferens*, will be conducted using a monoclonal antibody technique developed by Dr. David Caron, of the University of Southern California. For a copy of the *Brown Tide Newsletter* or information about the Brown Tide Assessment Project, contact the Program Manager, Dr. Mary Downes Gastrich, in the Division of Science, Research and Technology at (609) 292-1895 or the website at [www.state.nj.us/dep/dsr](http://www.state.nj.us/dep/dsr).

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**Brown Tide Alga**

*Aureococcus anophagefferens*

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*The Boater’s Guide to Barnegat Bay and Little Egg Harbor* is a three-phase project developed by Dr. Eleanor Bochenek, which includes a waterproof map and guide, a 28-minute video and a teacher’s curriculum, featuring two of the state’s most important watersheds. The waterproof map, released just in time for prime boating season, provides users with a wealth of information about fishes and marine mammals and habitat, pump out facilities, pollution, weather tides, boating safety and more. The companion curriculum and video are scheduled for release in time for the fall school semester. For more information about the *Boater’s Guide* project contact New Jersey Sea Grant College Program Communications at 732-872-1300 ext. 18 or Email kim@njmsc.org.

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The Barnegat Bay region is one of the primary areas featured in *A Coastal Ecotourism Guide to Central New Jersey* just released by NJMSC and the New Jersey Sea Grant College Program. The guide focuses on natural and cultural attractions and activities in the central region of the state. The first guide in the series, which showcased ecotourism based attractions in southern New Jersey, was nominated for a 1999 Governor’s Tourism Award. With the printing of the second guide, more than 100,000 copies will have been distributed in an effort to educate residents and visitors about ecotourism and the importance of sustainability of the state’s natural resources. The final edition in the highly successful series, *A Coastal Ecotourism Guide to Northern New Jersey*, will be completed in the early winter of 2000.
A variety of factors have strong influence on these deposition processes including aerosol characteristics, weather patterns, rainfall rates, etc.

IMPORTANCE OF ATMOSPHERIC NITROGEN DEPOSITION

Atmospheric deposition has been identified as an important non-point source for nitrogen entering coastal waters along the US East Coast. There have been an increasing number of investigations of the atmospheric nitrogen input to estuaries and bays, including recent studies that estimate approximately 25% of the anthropogenic loading of nitrogen to Chesapeake Bay is in the form of atmospheric nitrate deposition. This source arises almost entirely from anthropogenic emissions of nitrogen oxides. Atmospheric ammonium deposition contributes another 14% of the total. Apparently atmospheric nitrogen input exceeds that from fertilizers, animal wastes, and point sources. In Long Island Sound, approximately 22% of the total nitrogen input are believed to be from atmospheric deposition. Recent NY-NJ Harbor Estuary Program studies (HEP 1995) suggest that nitrogen is the limiting nutrient in the Harbor Estuary/Bight system and atmospheric deposition plays a potentially significant role. As a result, the atmospheric inputs of nitrogen may play a key role in affecting coastal ecosystems and nutrient dynamics in continental shelf sediments.

ATMOSPHERIC NITROGEN DEPOSITION AND COASTAL EUTROPHICATION

The importance of atmospheric nitrogen deposition is its linkage to coastal eutrophication (Pear 1993). Dr. H. Pae rl and colleagues of University of North Caroline have demonstrated a link between atmospheric nitrogen input and biological activity through their studies conducted in the waters of Bogue Sound, North Carolina. Their results suggest that acidic rainfall with high nitrate concentrations results in increased plant chlorophyll-a production relative to neutral rainfall with lower nitrogen concentrations. Thus, episodic nitrogen inputs through precipitation have significantly stimulated microscopic plant growth in coastal North Carolina. These studies are consistent with those obtained in Chesapeake Bay and highlight the importance of the nitrogen input and its potential impact on coastal primary production. The US eastern seaboard may be particularly susceptible to the input of atmospheric anthropogenic nitrogen because a large proportion of atmospheric nitrogen input will be readily available for phytoplankton growth and its direct linkage to coastal eutrophication. Consequently, increased atmospheric deposition of nitrogen may lead to frequent harmful algal blooms, low oxygen conditions, and episodic mortality that contribute to the long-term decline of coastal marine organisms. In the New York-New Jersey Harbor Estuary, New York Bight, and Long Island Sound, seasonal episodes of low dissolved oxygen in the water and algal blooms persist over long periods, attributed to both point and non-point sources of nitrogen.

CONDITIONS IN THE BARNEGAT BAY ECOSYSTEM

Algal blooms such as "brown tides" (yellow-brown water discoloration caused by the blooms) have frequently recurred over a wide geographical area along the northeast coast in recent years. These blooms can markedly reduce water quality and decimate populations of commercially valuable shellfish. In the Barnegat Bay ecosystem, the first brown tide was recorded in early July in 1985 in the vicinity of Surf City at Long Beach Island that was mainly caused by minute coccoid algae. Densities of those algae reduced water
column visibility to as little as 0.25m (0.8 ft). In the late 1980s, similar brown tides occurred in the Intracoastal Waterway at least as far south as Great Egg Harbor south of Atlantic City. Since then, similar blooms have been recorded in Great Egg Harbor, Tuckerton Bay, Great Bay, etc. Sometimes the brown tides in Barnegat Bay occurred in two separate areas near opposite ends of the Bay where the strength of the water flushing is minimal, and the blooms appeared to be dependent upon prevailing winds. These brown tides may have negative impacts on Barnegat Bay habitats that are important as nursery and spawning grounds to a variety of finfish, shellfish and benthic infauna. They may pose threats to human use of Barnegat Bay for recreational and industrial purposes. Excessive discharges of nutrient nitrogen from both point and non-point sources are suspected to be contributing factors to harmful algae blooms, in general. Atmospheric deposition could contribute significantly to the total nitrogen entering the bay. However, until recently, in situ atmospheric nitrogen deposition measurements had never been made in Barnegat Bay and processes that control atmospheric deposition have not been addressed.

CURRENT RESEARCH ON ATMOSPHERIC NITROGEN DEPOSITION

“Atmospheric Nitrogen Deposition to Barnegat Bay,” a project jointly sponsored by the New Jersey Sea Grant College Program and New Jersey State Department of Environmental Protection (NIDEP) was initiated in late 1998. The primary goal of this research was to determine atmospheric deposition rates of nitrogen to the Barnegat Bay ecosystem and to investigate the processes that control atmospheric nitrogen deposition. The ultimate goal of this research is to better understand the relationships between atmospheric nitrogen inputs and coastal eutrophication at this specific location.

With a newly constructed platform, atmospheric nitrogen sampling (primarily precipitation collection) has been operating since early 1999 at Rutgers Marine Field Station (RMFS) at Tuckerton. RMFS is located at the tip of a salt marsh peninsula adjacent to Little Egg Inlet within the Great Bay-Little Egg Harbor estuary in the southern region of Barnegat Bay. The precipitation sampler deployed at this site collects rainfall for nitrogen analysis. Limited aerosol samples were also collected.

Preliminary results suggest that atmospheric wet-deposition fluxes of nitrogen are comparable to those obtained in New York Bight and Chesapeake Bay, but significantly higher than those over the open ocean. Both concentrations and precipitation rates affected the seasonal variations in the nitrogen fluxes. The data also suggest that atmospheric nitrate ($\text{NO}_3^-$) and ammonium ($\text{NH}_4^+$) contribute a major fraction of the total dissolved nitrogen in precipitation at this location. This research represents the first atmospheric nitrogen deposition measurements ever taken in the Barnegat Bay ecosystem, and will provide new information on the present level and magnitude of atmospheric nitrogen deposition to the Bay. This information will be valuable to decision-makers, leading towards improved management of non-point source emissions to Barnegat Bay. Results from this project will also make it possible to compare the current atmospheric nitrogen loading to Barnegat Bay with other regions of the East Coast, thus contributing to a regional atmospheric nitrogen deposition database. The combined efforts will be important to understanding the relationships between atmospheric nitrogen deposition and harmful algal blooms along the coastal United States.
ATMOSPHERIC DEPOSITION OF NITROGEN TO COASTAL ECOSYSTEMS

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INTRODUCTION
The overabundance of nutrients, especially nitrogen, is causing environmental damage on all of the nation's coasts, according to a recent report from the National Academy of Sciences (www.nap.edu/books/0309069483). Nitrogen inputs to coastal ecosystems originate from many sources, including runoff from agricultural lands, sewage treatment plants, urban runoff, and natural inputs (Figure 1).

Atmospheric deposition (e.g., rainwater and dry deposition) also is an important source of nitrogen to coastal ecosystems. Dr. Seitzinger and colleagues in a number of national and international workgroups are examining the contribution of atmospheric deposition to nitrogen loading to over 40 estuaries along the eastern and Gulf coasts of the U.S. (NOAA Atmospheric Deposition Workgroup; Scientific Committee on Problems in the Environment Nitrogen Project). While the results of those workgroups are still being finalized, initial analyses indicate that atmospheric deposition can account for between 5% to over 50% of the nitrogen inputs to the watersheds. A preliminary analysis of nitrogen inputs to Barnegat Bay, for example, indicates that over 75% of the nitrogen inputs are from atmospheric deposition (Seitzinger and Sanders 1999). Clearly, controlling coastal pollution requires controlling sources of atmospheric nitrogen.

However, not all nitrogen is the same. One of the difficulties facing scientists and managers in relating atmospheric nitrogen deposition to coastal eutrophication is that nitrogen occurs in many forms and not all forms contribute equally to eutrophication. Nitrogen, for example, occurs in both inorganic and organic forms. Inorganic nitrogen (ammonia and nitrate) is known to be rapidly used by algae and to contribute to coastal eutrophication. Almost 40% of the inorganic nitrogen load to Barnegat Bay is estimated to be from atmospheric deposition (Figure 2). However, organic nitrogen is often a major component of nitrogen inputs, including rainwater nitrogen, and little is known about its sources or its
contribution to eutrophication. Preliminary estimates by Seitzinger and colleagues suggest that rainwater accounts for over a quarter of organic nitrogen loading to Barnegat Bay.

Figure 2 — N-Loads to Barnegat Bay from the Watershed and Direct Atmospheric Deposition (Preliminary Estimates)

The three major categories of nitrogen in atmospheric deposition are nitrate and ammonium (i.e., inorganic nitrogen) and organic nitrogen. A considerable amount is known about the magnitude and sources of inorganic nitrogen in rainwater. Over 80% of the nitrate in atmospheric deposition globally is anthropogenic in origin, primarily from the combustion of fossil fuels (Table 1). Approximately 75% of the ammonium is anthropogenic in origin, primarily from animal waste and biomass burning. However, almost nothing is known about the source of organic nitrogen in rainwater, including how much of it is from anthropogenic sources and thus how much might be controllable by changes in management practices.

Table 1 — Sources of N in Atmospheric Deposition (global estimates from Galloway et al. 1995)

<table>
<thead>
<tr>
<th></th>
<th>Amount (Tg N/yr)</th>
<th>Major Sources</th>
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<tbody>
<tr>
<td>NO₃</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Anthropogenic</td>
<td>22</td>
<td>Combustion fossil fuels</td>
</tr>
<tr>
<td>Natural</td>
<td>5</td>
<td>Microbial, biomass burning</td>
</tr>
<tr>
<td>NH₄</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Anthropogenic</td>
<td>39</td>
<td>Animal waste, biomass burning</td>
</tr>
<tr>
<td>Natural</td>
<td>13</td>
<td>Volatilization, wild animal waste, biomass burning</td>
</tr>
<tr>
<td>DON</td>
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<td>Anthropogenic</td>
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<td>Natural</td>
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Another major challenge facing scientists and managers with respect to atmospheric deposition is that the sources of nitrogen to the atmospheric that are ultimately deposited in a watershed and that thus can enter the estuary are emitted to the atmosphere from hundreds of miles (kilometers) outside of the watershed. NOAA scientist Dr. Robin Dennis has been modeling and mapping airsheds for nitrate and ammonium for many estuaries throughout the eastern and Gulf coasts of the US. His analyses indicate that nitrate deposited to the Barnegat Bay watershed, for example, can originate from as far north as Canada, as far west as Ohio, and as far south as Virginia. This presents a particularly difficult challenge for managers in controlling the large inputs of nitrogen to Barnegat Bay from the atmosphere. However, because almost nothing is known about the
sources of organic nitrogen in atmospheric deposition, we have essentially no information on where organic nitrogen deposited in the Barnegat Bay watershed originates.

With funding from the New Jersey Sea Grant College Program, New Jersey DEP, and the National Science Foundation, Dr. Seitzinger and Dr. Mazurek are quantifying atmospheric deposition of inorganic and organic nitrogen over an annual cycle in the New Brunswick area. Samples collected during summer 1999 show that organic nitrogen often comprises 20–30% of the total nitrogen in rainwater (Figure 3).

They are developing novel analytical techniques to chemically characterize the organic nitrogen that will help in the identification of the sources (e.g., natural, anthropogenic and photochemical) of organic nitrogen in atmospheric deposition. Furthermore, they are conducting experiments to examine the effect of this nitrogen on estuarine eutrophication. Results indicate that rainwater organic nitrogen can stimulate bacteria and phytoplankton production in Barnegat Bay water, and that the organic nitrogen in rainwater may influence the phytoplankton species composition (Figure 4) (Seitzinger and Sanders, 1999).

**Figure 3** — New Brunswick NJ, Summer 1999 Precipitation, TDN Composition

**Figure 4** — Rainwater Dissolved Organic N Effects Phytoplankton Species Composition
Currently, Dr. Seitzinger and colleagues are incorporating the results of their work on nutrient loading from atmospheric deposition as well as from agricultural and urban sources with inorganic and organic nitrogen bioavailability into the first land use/nutrient loading model that will explicitly include a more detailed modeling of the forms and amounts of nitrogen inputs to estuaries (Figure 5). Barnegat Bay will be the first system in their analysis.

![Diagram of nitrogen flow through a watershed](image)

**Figure 5** — Next Step for Watershed N Management—Bioavailable Nitrogen Model

This model also will be applied to a range of watersheds throughout the east coast of the U.S. to gain a regional perspective. Tools such as these will be made readily available to environmental managers so that they can more accurately assess the consequences of various development and nutrient management scenarios on coastal eutrophication, including management scenarios that may need to extend to states many hundreds of kilometers away to reduce atmospheric nitrogen deposition.

*For more information on the research activities being performed by Dr. Seitzinger and her team at the Institute of Marine and Coastal Sciences, visit: http://marine.rutgers.edu/NPC*
INTRODUCTION

Several large shallow interconnected lagoon-type embayments lie along the United States East Coast. Examples are Great South Bay, New York; Barnegat Bay, New Jersey; Chincoteague Bay, Maryland; and Palmlico Sound, North Carolina. These embayments tend to have very long flushing times because exchange with the ocean is limited to flow through only a few narrow inlets. They have significant but often conflicting economic and aesthetic value. For example, the growth of human population and agriculture within the watershed of these systems may adversely affect commercial and recreational fisheries. Furthermore, these systems also provide important nursery habitats for a number of commercially important coastal fisheries that may be impacted by natural or anthropogenic change. Consequently, understanding circulation and exchange processes within these systems would benefit water quality modeling efforts and ultimately result in more effective management programs. To improve our understanding of circulation processes in shallow lagoon systems, instruments were deployed in the Barnegat Bay/Little Egg Harbor system as part of a New Jersey Sea Grant College Program study. Details of the field effort are depicted in Figure 1.

Figure 1 — Dots show locations of S4 current meter deployment. Instruments were deployed between April 8 and May 20, 1997, and were located in Barnegat Inlet (B1), near Little Egg Inlet (LE1), and just north of the Route 72 Bridge (RT72).
FORCINGS AND FREQUENCIES

Currents in Barnegat Bay are primarily affected by winds and tides. Astronomical tides occur with a periodicity of 12.42 h, generating two tides in 24.84 h, which in turn causes the time of high and low water to occur later each day. Wind forced water motion tends to occur at subtidal periods, defined as water motion occurring at periods longer than one day. By definition, subtidal motion is of a lower frequency than tidal period motion.

Tidal forcing occurs in the deep ocean and propagates as a wave onto the continental shelf, through the inlets and into Barnegat Bay. The wind forcing has two components; local winds and remote winds. Local wind forcing, as the name implies, is the response of the water in the Bay to the stress imparted to its surface by the local wind. Remote wind forcing is driven by meteorologically-influenced sea level fluctuations in the coastal ocean that are transmitted through the inlets and into the Bay. Although both the tidal and remote forced motions are associated with oceanic responses that propagate into the Bay, the Bay responds differently to these influences because of different frequencies. Because the Bay has narrow inlets, it responds sluggishly to coastal sea level fluctuations. For example, if the ocean suddenly rose 1 m (about 3 ft), it would take longer than a tidal cycle for the Bay to rise to the same height. meter. Thus, the Bay does not respond fast enough to be filled by tidal period forcing and tidal motion is strongly attenuated. In contrast, subtidal motion passes relatively unattenuated from the ocean into the Bays. The tendency for systems such as Barnegat Bay to act as a “low-pass filter” (admitting only low-frequency motion) is evident in sea level data (Figure 2).

Figure 2 — A) Sea level from Atlantic City during April–May 1997. B) Sea Level in Barnegat Bay at Waretown, April–May 1997. C) Lowpassed sea level at Atlantic City (thin line) and Waretown (thick line) April–May 1997.
The mean tidal range on New Jersey’s inner shelf is 120 cm (36.5 in) Figure 2A, while in the Bay it is less than 20 cm (6.1 in). In contrast, the lower frequency sea level motion passes relatively unattenuated into the bay. In general, the sea level data emphasizes the importance of subtidal motion in the estuary.

**TIDAL MOTION**

The mean tidal range in Barnegat Bay exhibits little spatial structure away from the inlet (Figure 3). For example, at Waretown the mean tidal range is 18 cm (5.5 in), whereas near the head of the bay at Mantoloking, the mean tidal range is 15 cm (4.6 in). Appreciably larger tides are evident in Little Egg Harbor; the tidal range exceeds 1 m (3.1 ft) near the inlet and attenuates gradually inside the estuary as the tidal wave propagates northward. The larger tidal ranges in Little Egg Harbor are likely due to the greater width of Little Egg Inlet relative to Barnegat Inlet.

**Figure 3-ABC** — 3A. Tidal range in study area. Tidal ranges obtained from NOAA tide tables. Crosses indicate location of tide gauges. Ranges are in meters. 3BC. Time of high water (central panel) and time of low water (right panel) relative to Sandy Hook. Tidal data obtained from NOAA tide tables. Time is in hours. Crosses indicate location of tide stations. Arrows indicate direction of tidal wave propagation.

The “signature” of the northward propagation of the tide from Little Egg Inlet is lost just south of Barregat Inlet. This is apparent in tidal phase propagation, presented as the time of high and low water in Figures 3BC. During high water, the tidal wave propagates at 2.2 m/s (5.5 mph), whereas at low water, the wave propagates at 1.7 m/s (4.5 mph). In the absence of function, a tidal wave would travel at a speed of about 4.5 m/s (11.1 mph) in Little Egg Harbor (mean depth about 6.2 ft). The slower phase speeds observed are due to the influence of friction that acts to slow wave propagation. The larger tidal range in Little Egg Harbor, relative to that in Barregat Bay, indicates that tidal flushing is more effective in Little Egg Harbor. Enhanced tidal flushing is evident in salinity levels in the Barnegat Bay/Little Egg Harbor estuary. Consequently, exchange between Little Egg Harbor and Barregat Bay acts to flush Barregat Bay with oceanic waters.
Figure 4 shows current meter data collected along the major axis of the estuary degrees east of north from the Route 72 bridge mooring. Both tidal and subtidal flows have velocities of approximately 10 cm/s (4 in/s). However, because subtidal motion has a period of 2–5 days, or 4–10 tidal periods, water movements excursions due to subtidal motion are 4–10 times greater than that of the tidal period motion. Subsequently, the subtidal motion will exchange water between the bays.

![Figure 4](image)

**Figure 4** — S4 data from Route 72 Bridge. A) Tidal period currents along the major axis of the bay from mooring RT72. B) Subtidal currents along the major axis of the bay from mooring RT72.

**SUBTIDAL MOTION**

The subtidal (low frequency) currents directed through the major axis of the Bay are shown in Figure 5A.

![Figure 5](image)

**Figure 5** — A) Lowpassed currents from all three moorings. B) CEOF Mode 1 time series along major axis. C) CEOF Mode 2 time series along major axis. In all three plots the thinnest line is the Barnegat mooring, the intermediate line is the RT72 mooring, and the thickest line is the LEH mooring.
Using a statistical technique, the remotely forced motion was separated from the locally forced motion (Mode 1). Most of the subtidal variability was contained in the remotely forced mode, which accounted for over 70% of the variability in the data. Twenty percent of the variability is due to local wind forcing (Mode 2).

**MODE 1—REMITELY FORCED FLOW**

The analysis suggests that low frequency flow is primarily driven remotely by coastal sea-level fluctuations, as shown in Figure 6A, which shows a high correlation between Mode 1 currents and the sea-level difference, or equivalently a pressure gradient, between moorings LEI and RT72. Figure 6B overlays the pressure difference between moorings LEI and RT72 and that between mooring RT72 and Waretown. Figure 6B emphasizes that the pressure gradient between mooring RT72 and Waretown tends to be of the opposite sign of that between moorings RT72 and LEH. Thus, when the ocean rises, sea level at RT72 remains lower than at either inlet, and the Bay fills. As the ocean drops, water levels at RT72 remains higher than at either inlet and the Bay empties.

![Figure 6](image-url)

**Figure 6** — A) Mode 1 time series along major axis for RT72 mooring (solid line) and lowpassed pressure difference between LEI and RT72 moorings (dashed line). B) Pressure difference between LEI and RT72 and RT72 and Waretown.

The character and dynamics of the Mode 1 flow are shown further in Figure 7(AB), which presents lowpassed oceanic sea level, pressure gradients, wind, and Mode 1 current vectors. During the positive phase of Mode 1 (Figure 7A) sea level in the ocean is rising. This produces the pressure gradient driving flow into the Bay. Note that the inflow from Little Egg Inlet continues northward past the RT72 mooring, despite the closer proximity of this mooring to Barnegat Inlet. Consequently, as sea level rises in the ocean (positive phase of Mode 1), water flows from Little Egg Harbor into Barnegat Bay. During the negative phase of this mode (Figure 7B), sea level drops in the ocean, which allows flow...
Figure 7 — Modal current vectors, pressure gradients, lowpassed coastal sea level (inserted graphic) and wind vector. Sea level record is shown as a two-day record centered around the time that the vectors are presented. Panels show conditions during A) positive phase Mode 1; B) negative phase Mode 1; C) positive phase Mode 2; and D) negative phase Mode 2.

During both phases of this mode, the flow between the bays (RT72) is in the same direction as the flow at Little Egg Inlet, and is probably due to the greater width of Little Egg Inlet that allows Little Egg Harbor to respond more quickly to coastal sea-level fluctuations. The faster response in Little Egg Harbor generates a pressure gradient between Barnegat Bay and Little Egg Harbor that is of the same sign as the gradient between the estuary and the ocean. Consequently, low frequency exchange between Barnegat Bay and the ocean occurs through Barnegat Inlet and indirectly through Little Egg Harbor.

**Mode 2 Local Response**

Mode 2 (Figure 7CD) is driven by local winds and characterized by a through-flow circulation; flow enters the upwind inlet and exits the downwind inlet. However, it only accounts for about 20% of the total variance in the data, thus local wind-driven response does not dominate subtidal flow. Note that currents associated with Mode 2 are at times inconsistent with pressure fields within the bay, further emphasizing that this mode is forced by local winds.
CONCLUSIONS

Analysis of current meter measurements in the Barnegat Bay-Little Egg Harbor estuarine system isolated remotely forced motion from the locally forced motion. The first mode explained 70-80% of the variance in the low-frequency flow. This mode, characterized by a simple pumping, is forced by coastal sea level and dominates subtidal flows at all moorings, even in the Bay’s interior. The second mode is forced by local wind and explains 20% of the variance in the low frequency flow. This mode is characterized by a through-flow mode, with flow entering the upwind inlet and exiting the downwind inlet.

The low frequency flow drives water exchange between Little Egg Harbor and Barnegat Bay. The exchange acts to increase flushing in Barnegat Bay. This exchange is driven by a pressure gradient between the bays setup because Little Egg Harbor responds more quickly to coastal sea level than does Barnegat Bay. The quicker response of Little Egg Harbor is likely related to the greater width of Little Egg Inlet. The faster response in Little Egg Harbor relative to Barnegat Bay is apparent at both tidal and subtidal periods. Consequently, as sea level rises in the ocean, a pressure gradient develops between the bays that drives water from Little Egg Harbor into Barnegat Bay. In contrast, as coastal sea level drops, the pressure gradient between the bays drives water from Barnegat Bay into Little Egg Harbor.

This description of subtidal motion in Barnegat Bay-Little Egg Harbor is consistent with that described by other investigators for Great South Bay, New York. Specifically these studies suggest that subtidal motion in large, shallow lagoon-type estuaries is primarily forced by coastal sea level. This study indicates that this mode dominates current fluctuations in Barnegat Bay’s interior.
Restocking Estuaries: Predicting Bivalve Larval Transport

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INTRODUCTION

Historically, Barnegat Bay and its southern extension, Little Egg Harbor, have had important shellfisheries associated with them. From the late 1800s to the 1920s, large numbers of oysters were harvested from these estuaries. Beginning in the 1920s after a decline in oyster numbers, the hard clam industry increased in importance and continues as the main bivalve species harvested in the Bay. In recent years, the numbers of hard clams have declined in the wild, leading to the need for programs to maintain the fishery.

One method proposed to help maintain populations of estuarine bivalves (clams, oysters, scallops, etc.) is to increase the number of spawning adults by reseeding depleted habitats. The idea behind this type of project is that the addition of breeding adults will increase the number of larvae available for settlement and recruitment to the next generation. The success of reseeding efforts in estuaries depends, in part, on whether there is a spawner-recruitment relationship. In other words, a substantial portion of the larvae spawned in an estuary would need to be retained within that estuary. For instance, if most of the larvae spawned within the estuary were flushed out of the estuary then planting spawning adults would be useless. The spawner-recruitment relationship has been documented for the bay scallop in North Carolina and Florida and the oyster in Chesapeake Bay. This suggests setting up “spawner sanctuaries” may be effective for certain species.

Spawner-transplant programs have been tried in the past for estuarine bivalve species in several locations along the east coast of the US with variable success. A program for transplanting spawning adults of the hard clam (*Mercenaria mercenaria*) was carried out in Great South Bay, Long Island, NY. The program was thought to be unsuccessful because the scale of the transplant project was small relative to the native populations. Another program on Long Island sought to reseed populations of bay scallops (*Argopecten irradians*) that had been decimated by brown tide algal blooms. While the program showed some success and increased recruitment to bay scallop populations, reoccurrence of the brown tide blooms led to the ultimate failure of the program. A more successful attempt to restore bay scallop populations was carried out in Bogue Sound, North Carolina, where reseeding of adults led to significant increases of larval recruitment in scallop populations. Another successful broodstock-replacement program for the American oyster (*Crassostrea virginica*) was carried out in the Great Wicomico River in Chesapeake Bay. Transplanting adult oysters into the estuary resulted in significant increases in larval recruitment in that estuary. The variability in the success of these
programs demonstrates the need for further research on the factors influencing larval retention and ultimate recruitment to populations of estuarine bivalves.

To predict the effectiveness of adding spawning adults to estuaries, we need to have basic knowledge of how bivalve larvae interact with tidal and wind-driven currents in estuaries and how this influences their movement. With support from the New Jersey Sea Grant College Program, we are studying how tidal currents influence the vertical distribution and behavior of bivalve larvae. We are using both modeling and observation to determine the role of physical and biological processes in the transport of bivalve larvae. With this knowledge, we will be able to implement more complex models in the future and to make better predictions of where to place spawner sanctuaries.

**BIVALVE LARVAE AND THEIR TRANSPORT IN ESTUARIES**

Several species of bivalves have been harvested from Barnegat Bay including oysters, bay scallops, and hard clams. The hard clam (*Mercenaria mercenaria*) is the most heavily-fished species in the bay during recent years and therefore, the life cycle of this species will be discussed. The life-cycles of many commercially harvested bivalves, including the oyster, bay scallop, and soft-shell clam, are similar and the processes affecting their populations will be similar.

The hard clam has a sedentary adult stage that remains buried in sediments with only their siphons extending into the water column for feeding. During reproduction, the adults release eggs and sperm into the water where fertilization occurs and development of a planktonic larval stage begins. Development of the non-shelled stage from fertilized egg to the naked veliger occurs in about 16 h. After this time the larvae begin to secrete the shells characteristic of bivalves and are called strait-hinged veligers. Larvae remain in this stage from one to three days and range in length from 90–140 µm (1000 µm is equal to 1 mm). From 3–20 days into development (depending on temperature and available phytoplankton food), the larval shell develops an umbo, which gives it the general appearance of an adult bivalve. The larvae range in size from 140–220 µm during this stage. During both the strait hinge and the umbone stages, bivalve larvae swim using ciliated vela (hence the name veliger). The final stage before the larvae settle to the bottom and metamorphose into juvenile clams is called the pediveliger, meaning the larvae have both a foot for movement on the bottom and a velum for swimming in the water. The pediveliger stage ranges from 170–220 µm in length. At the pediveliger stage the larvae may spend part of the time exploring the bottom and part swimming in the water.

The small size of the hard clam larval stages, and how long it takes them to develop, is important because it is during this period that they are in the water column at the mercy of the currents. Current velocities in Little Egg Harbor may range from several centimeters per second (0.22 mph) to greater than a meter per second (2.2 mph). Hard clam larvae, on the other hand, have swimming speeds of only 1–5 mm/s (0.002 mph). This great difference in velocity means the larvae cannot control by horizontal swimming where the currents take them. It is at this time the larvae may be transported to unfavorable habitats or even flushed out of the estuary into the open ocean, making them unavailable for recruitment to the estuary.

Another factor complicating this model of larval transport is vertical mixing. The vertical
shear in current velocity observed in estuaries also causes turbulent eddies that may mix particles and larvae throughout the water column much faster than by simple sinking and swimming by the larvae themselves. Consider current velocities over a tidal cycle in an estuary; current velocities reach a maximum during the flood and ebb tides and go to nearly zero during the slack. Mixing will be at a maximum during the flood and ebb tides and it will be near zero during the slack. When mixing is at a maximum, suspended material, including larvae, may be mixed higher into the water column. So, larvae may be mixed into the water column on both flood and ebb tides, and transported out of the estuary on ebb tides. However, if a larva has a behavioral adaptation to sink or swim all the way to the bottom, then it may not be resuspended by turbulence and mixed into the water.

Do bivalve larvae, even though they cannot fight the relatively strong currents, have some control over the direction they are transported in an estuary? One of the theories describing how bivalve larvae are able to control the direction they are transported is by the method of vertical swimming. It is well known that as water flows over sediment on the bottom of the sea-bed, estuary, or river, there is a frictional drag between the water and the sediment. The influence of this friction extends upward some way towards the surface of the water. The effect of this friction is to slow down the water flow, so that the actual velocity of the water at the surface is much greater than that near the bed. This is known as a vertical shear. A particle suspended in the water close to the bottom will not be transported as far as a particle close to the surface because of the difference in velocity. These principles apply to living things such as bivalve larvae. If a bivalve larva possessed a behavioral adaptation to keep it in an estuary, then it would swim up toward the water surface during the flood tide, and sink or swim towards the bottom during the ebb tide. This behavioral adaptation would give the larva a net transport into the estuary, thus keeping it in a favorable habitat.

While the vertical movement of bivalve larvae may be important, there are other factors that influence larval transport in an estuary. The time for development from the larval stage to the adult stage varies among different commercially important bivalve species. Larvae of oysters (Crassostrea virginica) may remain in the plankton from 14–25 days. The soft-shell clam (Mya arenaria) has a larval development time of 14 days, and the development time of the bay scallop (Argopecten irradians) is 5–11 days. Species with a shorter larval development period have a greater chance of retention within an estuary. Other factors, such as water temperature and food availability, will influence the amount of time a bivalve larva spends in the plankton. All of this leads to year-to-year variability in the number of larvae that develop into adults and contribute to growth of the populations and should be considered when planning a broodstock replenishment program.

MODEL OF PROCESSES INFLUENCING VERTICAL POSITION OF LARVAE

How do we determine whether bivalve larvae are using behavioral adaptations to control their vertical position in the water column? The approach we are taking is to make a model that simulates the sinking and mixing of non-living particles in the water column over a tidal cycle. The results of the model with actual observations of the changes in the vertical distribution of larvae in Little Egg Harbor can then be compared. If there are differences between the vertical distribution predicted by the model with those of larvae collected in the field, then we assume these differences are due to larval behavior.
The mathematical model can be described in words by the following relationship:

Concentration of a tracer at each depth = sinking by the tracers + mixing by turbulence.

This relationship states that a tracer's position in a column of water is determined by a balance between sinking of the tracer and mixing by turbulence. The force of gravity acting on the tracer is constant, so tracers will always have a tendency to sink. Turbulent mixing is related to the velocity of the currents. Therefore, during the flood and ebb tides, when current velocities are greatest, mixing will overwhelm the tendency of the tracer to sink and particles will be distributed throughout the column of water. During slack tide, the tracers will sink at a greater rate than they are mixed because currents are relatively low or zero. Figure 1 shows the results of a model simulating mixing of a tracer over a tidal cycle. Notice that during the flood and ebb tides particles are mixed into the water column, but during the slack tide the particles are mostly concentrated at the bottom.

![Figure 1](image_url)

**Figure 1**—Results of a model simulating the change in concentration of a tracer at different depths in the water column over a tidal cycle. The numbers on the contours are the relative concentrations of the tracers: purples and blues denote lower concentrations and reds and oranges denote higher concentrations.

We can deduce from this model that the tracers will tend to be transported almost equally during the flood and ebb tides. Therefore, if bivalve larvae have no behavioral adaptations, we would expect them to be flushed out of the estuary on the ebb tide.

A sampling program was conducted to determine if bivalve larvae were being transported passively by currents or if they used behavioral adaptations to control their vertical
distribution. Sampling was conducted by anchoring at a point in the inlet to Little Egg Harbor and using a pump to collect bivalve larvae at several depths: at the surface, the middle of the water column, and near the bottom. This sampling regime was repeated once every hour over a 12-h period so that sampling occurred during flood, ebb, and slack tides. Results from the sampling are shown in Figure 2. Notice that the densities of bivalve larvae are highest during the flood and slack tide and drop to very low numbers during the ebb tide. This result is quite different from what is predicted from the model, which simulates passive tracers. It can be inferred from the observations that bivalve larvae are indeed using a behavioral adaptation to influence their transport into the estuary.

![Figure 2](image)

**Figure 2**—Change in bivalve larval concentration over a tidal cycle from samples collected in Little Egg Harbor. White bars = samples 0.5 m below the surface; red bars = samples 5 m above the bottom; blue bars = samples 1 m above the bottom; EDT = Eastern Daylight Time.

The hypothesis we are using to explain how bivalve larvae control their vertical position over a tidal cycle is as follows: The larvae swim intermittently during the flood tide, at which time they are also mixed throughout the water column by turbulence. As slack tide approaches, the larvae spend more time sinking than swimming, but they are still present in the water column. The larvae either sink exclusively or swim downward during the ebb tide until they reach the bottom. This behavior would facilitate transport into the estuary and, overall, would promote retention within the estuary.

**IDENTIFYING BIVALVE LARVAE**

One difficulty in studying bivalve larvae is their small size and lack of distinctive shell characteristics—it is hard to tell species apart. In addition to the modeling and observational studies, we are also developing techniques using DNA technology to identify bivalve larvae. We are able to take advantage of each species’ unique genetic code to identify it. Identifying an organism using DNA has several advantages: the DNA makeup
of an organism does not change with variable environmental conditions, and it does not change as an organism develops. Compared to using the body shape of an organism, DNA provides a more objective diagnostic character that can be used for reliable identification.

The development of DNA methods for identifying unknown larvae requires two steps. First, regions of DNA must be found that are unique to the group of organisms we are interested in, such as a species of bivalve. This is done by obtaining adults of the species, then extracting their DNA and finding a region that is unique to that group by determining the DNA sequence. Once we have found a unique sequence of DNA, we manufacture a DNA “probe,” a small strand of DNA that is complementary to the region we have identified. A “marker molecule” is then attached to the strand of DNA, making it easy to detect by a simple chemical reaction.

Once the DNA probe has been manufactured, it is used to identify unknown larvae by a procedure called DNA dot blotting. We start by taking a single larva, extracting the DNA from it, and making many copies of its DNA by a procedure called the polymerase chain reaction (PCR). The amplified (or copied) larval DNA is then bound to a piece of filter paper and washed with a solution that contains the DNA probe. If the DNA sequence from the larva matches the DNA sequence of the probe, then the probe will bind to the larval DNA on the filter. If the DNA sequence of the probe does not match the larval DNA, then it will not bind and will be washed away. The filter is then soaked in a chemical solution that reacts to the marker molecule attached to the DNA probe. If a colored mark is formed, then the larva that the DNA was extracted from is probably the species from which the probe was designed. Figure 3 shows a schematic of the procedure for identifying bivalve larvae.

One of the advantages of using DNA techniques for identifying organisms is it can be used on any type of organism at almost any stage of its life cycle. Since all living things possess DNA, the same identification procedures can be used for many different types of organisms. This makes accurate identification of species possible even in laboratories that do not have special training in the identification of certain groups of organisms.
There are disadvantages in using DNA techniques for identifying bivalve larvae. Specialized and sometimes costly equipment is needed for manipulating and analyzing DNA. Making the initial investment may be more than some laboratories can manage. These techniques can also be laborious and time-consuming, making it impossible to identify a large number of individuals. The use of DNA techniques presently may not be feasible for large-scale ecological surveys where collections may have many individuals. However, for studies investigating small-scale problems that have very specific goals, DNA techniques are a valuable tool. Figure 4 shows the developmental stages of a bivalve.

![Figure 4 — Schematic diagram of the developmental stages of a bivalve larva. A= Straight hinge veliger; B= Veliger; C= Pediveliger; V = velum; F = foot; U = umbo.](image)

We have designed DNA probes for two species of clams that occur in New Jersey; the hard clam (Mercenaria mercenaria) and the surfclam (Spisula solidissima). The probes have been used in studies tracking the abundance of the larvae of these species. These studies are part of ongoing projects to investigate the ecology of these bivalves.

**SUMMARY AND CONCLUSIONS**

Populations of commercially harvested bivalves in estuaries have declined in recent years due to natural and human causes. The programs designed to mitigate these impacts have met with varying degrees of success. Further study is needed to better predict the outcome of these projects. Our study focused on the interactions of physical processes and the behavior of bivalve larvae. A model was developed to predict the vertical distribution of bivalve larvae if they were passive particles. Observations of vertical distribution of bivalve larvae in Little Egg Harbor indicated that they were probably using behavioral adaptations to control their distribution. This approach, using both modeling and observation, can be extended to other parts of Little Egg Harbor and used as a tool to better predict the placement of spawner sanctuaries.

A second portion of this study dealt with the problem of identifying bivalve larvae using DNA techniques. Further development of these methods will allow researchers to identify the larval stages of certain bivalves. Furthermore, similar methods may be used not only on other bivalve species, but any other organism for which identification is a problem.
NITROGEN FLUX THROUGH BARNEGAT INLET: THE OCEAN AS SOURCE AS WELL AS SINK

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INTRODUCTION

Eutrophication of estuaries is a common water quality problem. Increases in nutrient loadings and primary production associated with eutrophication sometimes result in low oxygen concentrations (hypoxia or anoxia) that are detrimental to fisheries, sea grass beds, animal and plant communities, and recreational uses of coastal bays. The degree of eutrophication is affected by nutrient inputs, residence time of water, light limitation, and other factors. An understanding of the relative nutrient loadings from different sources can lead to better management of those inputs and help reduce eutrophication. Nutrients may enter coastal water through direct runoff, groundwater seepage, and atmospheric deposition. Nutrients may be buried in or released from bottom sediments. These sources or sinks of nutrients have been or are being studied by several university investigators. At the bay-ocean boundary, it has heretofore been assumed that the ocean is a sink for nutrients and that flood tides assist in the dilution of nutrient levels in the estuary. However, measurements of nutrient flux (net exchange) at this boundary have been difficult to develop because the only way to determine whether a particular substance is imported to or exported from an embayment over a long period of time and to detect the effects of episodic events (such as storms) is to conduct continuous long-term monitoring. This New Jersey Sea Grant College Program-supported project is the first study that monitors the nitrogen import and export at an estuary-ocean boundary continuously and over a long-term, using a piece of newly-developed in situ equipment.

PROJECT GOALS

Barnegat Inlet (Figure 1) is the primary pathway through which water and materials are exchanged between the ocean and Barnegat Bay. Nitrogen exchange through Barnegat Inlet has never been measured, and it is critical to do so for any comprehensive eutrophication study of Barnegat Bay. To quantify relative nitrogen loadings to Barnegat Bay and to develop a eutrophication model for the Bay, nitrogen flux through Barnegat Inlet must be quantified. In addition, it is probable that ammonia discharged from nearby coastal outfalls along the New Jersey shore may enter the Bay on the flood tide. However, no data have been collected in New Jersey to assess the potential significance of influx of ammonia from the coastal zone. It is the intent of this research project to collect a time-series of nitrogen flux data at Barnegat Inlet and to identify the variability of that flux. The results obtained from this project can be used to refine existing nitrogen loading estimates. The results will also fill critical data gaps in the knowledge required for dynamic water quality modeling of the Bay.
STUDY APPROACH

To quantify nitrogen exchange at Barnegat Inlet, both water flows and nitrogen concentrations were measured. Current meters that record velocity and water depth as well as water temperature, salinity, dissolved oxygen, and turbidity were deployed at a monitoring station located in the Inlet. A newly developed, highly accurate in situ nutrient analyzer was also deployed to record concentrations of two important forms of nitrogen, nitrate plus nitrite. The device measures the light absorbance of a colored dye produced by a chemical reaction and is capable of recording and storing data for up to two months. It is essentially a miniature underwater chemistry laboratory. A fluorometer is also attached to the nutrient analyzer. Water samples were also collected for laboratory measurements of concentrations of all forms of nitrogen: nitrate (plus nitrite), ammonia/ammonium, dissolved organic nitrogen, and particulate nitrogen. Laboratory values were used to verify the in situ measurements.

The monitoring station was installed in a long, shallow and straight portion of Barnegat Inlet where water was well mixed and current flow was uniform. In the channel, the values of current velocity and concentration were assumed to be representative of the entire cross-section of Barnegat Inlet. Only a long-term continuous monitoring methodology as proposed will generate a time series of nitrogen exchange data that will fully capture the variability in nitrogen exchange. To identify causes for the variability of nitrogen flux, the time-series will be analyzed for its correlation (or coherence) with a number of existing environmental data sets.

RESULTS

The in situ nutrient analyzer and current meter were successfully deployed from March 17–April 11, 2000. A set of long-term continuous nitrate concentration and flow data were recorded (Figures 2–5).
Figure 2 — Variation of Nitrate, Salinity and Fluorescence at Barneget Inlet

Figure 3 — Variation of Temperature, Water Depth, Oxygen and Salinity at Barnegat Inlet
A major event was detected and recorded during this monitoring period. High concentration of nitrate occurred in water that was exchanged through Barnegat Inlet from April 18–24, 2000. This concentration was two to ten times greater than normally recorded levels. This six-day high-nitrate concentration event appeared to be associated with rainfall that occurred on March 16 and March 21 (0.90 and 2.24 in, respectively), as indicated by low salinity during this period. At the early stage of this six-day high concentration period, high nitrate concentration was associated with low salinity confirming that the Barnegat Bay watershed was the likely source of nitrate. However, during the late stage of this six-day period, high nitrate concentration was associated with high salinity, indicating the coastal zone/ocean as the potential source of nitrate. A south and southwest wind prevailed from March 14 – March 17 immediately before the observed high nitrate concentration. This would indicate ocean upwelling as a possible source. However, because low salinity was recorded during this same period, ocean upwelling is probably discounted as the likely cause. A north and northeast wind occurred from March 17–23. This wind could have held watershed-derived and nitrogen-enriched freshwater within the nearshore zone and kept it available for tidal exchange returning to the Bay.

Prior to this recent successful field deployment, water samples were taken manually over a tidal cycle on two separate days. One set of samples was taken on July 19, 1999 and another on October 11, 1999. Data taken from both days indicated that high concentrations of nitrate, ammonia, and phosphate were associated with high salinity, indicating the coastal zone/ocean as the source of these constituents in the bay. These results were presented on September 29, 1999 at the 15th Biennial International Conference, Estuarine Research Foundation ‘99 in New Orleans.

**CONCLUSIONS**

Preliminary results indicate that nutrients can enter Barnegat Bay from the ocean as well as from the watershed and that continuous long-term monitoring is necessary to detect episodic events such as freshwater flow from the watershed or ocean upwelling. It is also necessary to quantify net import and export of nitrogen to an estuary across its several boundaries (inlets). Future water quality/eutrophication modeling must consider these dynamic boundary conditions.
INTRODUCTION

Although the ocean is a valuable resource for recreation and wildlife habitat, it is also a significant resource for the disposal of treated wastewater. There are 14 ocean outfalls that discharge treated municipal wastewater into the coastal waters off New Jersey. A total of approximately 220 million gallons per day of treated wastewater is permitted to discharge into these 14 ocean outfalls. The outfalls vary in length, discharging from 1600 feet to 7500 feet offshore. The wastewater treatment plants that these outfalls serve use chlorination for disinfection. Since none of the treatment plants have dechlorination facilities, the wastewater being discharged through these outfalls at times may have a high chlorine concentration, which, in turn, may result in concentrations in the receiving waters, exceeding criteria established by the New Jersey Department of Environmental Protection (NJDEP) to protect aquatic life.

A paradox exists concerning chlorination. Wastewater treatment plants chlorinate to kill the pathogens that are contained in the wastewater. Pathogens are organisms that are capable of infecting or transmitting diseases to humans, making disinfection necessary to protect human health. The problem with this approach is that the residual chlorine and/or the chlorine-produced oxidants (CPO) may be toxic to marine organisms even at very low concentrations.

Several approaches are available to address this chlorine paradox. One solution is to dechlorinate (i.e., add chemicals to the wastewater to remove the chlorine after it has destroyed the pathogens). The problem with this procedure is that the chlorine needs to be in contact with the wastewater for an extended time period to effectively destroy the pathogens. The chlorine disinfection process is not instantaneous. A typical wastewater treatment plant without an ocean outfall pipeline would have a chlorine contact tank. The chlorine is injected as the wastewater enters the tank. As the wastewater travels through the tank (typically 30 min or more travel time), the chlorine destroys the pathogens. This makes it simple to dechlorinate at the end of the tank before the wastewater is discharged from the plant (unlike chlorination, dechlorination is virtually instantaneous). The problem with the treatment plants discharging to the ocean is that they use the outfall pipe as their chlorine contact tanks and, thus have no means to dechlorinate the effluent. Because wastewater has to travel between one to twelve hours from the treatment plant to
the ocean before it is discharged (depending on the discharger), the wastewater pipe makes an ideal chlorine contact chamber.

Another possible approach to the problems associated with chlorination is to use an alternative means of disinfection, such as using ultraviolet or ozone disinfection. For the ocean discharges, alternative methods of disinfection are very expensive and in some cases unfeasible to implement due to discharge conditions such as high-suspended solids concentrations in the wastewater or high flow rates. Since chlorination is the most viable and cost effective alternative for disinfection for the ocean discharges, there is a clear need to determine with some precision the effects of CPOs in the marine environment. Two problems exist concerning chlorine discharged from the ocean outfalls. The first problem is the difficulty of predicting the chlorine concentrations in the vicinity of the ocean outfalls. The second problem is the difficulty of determining how these chlorine concentrations affect marine life.

Since chlorine concentrations are only measured at the wastewater treatment plant, an initial mixing model must be used to determine chlorine concentrations in the ocean near the discharge. The NJDEP regulations provide for acute\(^1\) and chronic\(^2\) mixing zones within which the chlorine water quality criteria can be exceeded. An initial mixing model must be used to determine the dilution factor at the edge of these mixing zones. These dilution factors can then be used to calculate the chlorine concentration at the edge of the acute and chronic mixing zones if the concentration at the discharge point is known.

The impact on aquatic organisms of the chlorinated wastewater being discharged from the ocean outfalls has never been thoroughly examined. Under present regulations the ocean dischargers perform whole wastewater toxicity testing, but these tests are performed on wastewater samples taken prior to chlorination. Also, whole wastewater toxicity testing has never been performed on samples taken in the ocean from the acute or chronic mixing zones. Therefore, potential impacts have been based on laboratory-derived data and not on field-testing.

The present method of determining the concentration of chlorine at the boundaries of the mixing zones is by application of numerical models to worst-case scenarios. A more realistic approach is to determine concentration levels that would only be exceeded a specified percentage of the time, say, for example, once in 1000 days. This concentration would be determined from a long-term simulation of expected discharge and receiving water conditions.

**PROJECT GOALS**

In this multidisciplinary three-year study, two goals were established: 1) to develop a dynamic mixing model that can be used to perform a probabilistic analysis of ocean outfall mixing, and 2) to evaluate the impact of CPOs on the marine environment. The model will be used to predict dilution factors for a variety of time intervals and to predict

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\(^1\) Acute concentrations are those that result in mortality of organisms.  
\(^2\) Chronic concentrations are those that may not kill organisms outright, but rather may affect life processes such as growth and reproduction.
probabilities of exceeding water quality criteria. The model will be validated by comparison with field measurements. The effect of the discharge of CPOs on marine organisms will be determined through toxicity testing and transplant studies.

**METHODOLOGY**

The three New Jersey outfalls selected for this study represent the three types of outfalls that exist along the New Jersey coast. The studies included long-term data collection, dye-release experiments, acute and chronic whole wastewater toxicity testing, and caged animal experiments for two of the outfalls. The South Monmouth Regional Sewerage Authority’s (SMRSA) outfall is representative of the long ocean outfalls (5400 ft) that discharge into water depths of approximately 50–60 ft. During summer months, these waters form two layers, with an upper layer that can be 10 °C warmer than the lower layer. The Cape May County Municipal Utilities Authority’s (CMCMUA) Seven Mile Beach outfall is representative of the long ocean outfalls (5900 ft) into relatively well-mixed (nonstratified) waters approximately 26–33 ft deep. The City of Asbury Park’s ocean outfall is representative of the short ocean outfalls (2000 ft) in less than 33 ft water. Only long-term receiving water data will be collected off Asbury Park. This allows for running the model to predict wastewater movement and dilution for the near shore areas.

**TOXICOLOGY STUDIES**

In the first year of the project, toxicity studies using American oyster larvae (*Crassostrea virginica*) and mysid shrimp (*Mysidopsis bahia*) were carried out on the wastewater from the SMRSA outfall pipe and the CMCMUA Seven Mile Beach outfall pipe, prior to entering the ocean. The American oyster has been used for monitoring purposes around the world and has been used by the United States Environmental Protection Agency (USEPA) for monitoring ocean outfalls. The purpose of our studies was to evaluate the toxicity of the wastewater after chlorination and to establish dosage response curves for both acute and chronic effects. The former usually tests rates of mortality, while the latter often measure growth and reproduction. Using the results from these tests, dose-response curves were developed to evaluate the impacts of chlorinated wastewater in the ocean. The tests included dechlorinated samples, which served as controls.

A single-day, dye release was performed at the SMRSA and CMCMUA outfalls. The dye enabled the wastewater plume to be tracked and the dilutions to be measured at various locations. The dye concentration in the discharged wastewater was maintained at a relatively high concentration to allow dilution measurements as high as 1000:1. The dye tracer also allowed for the collection of ocean samples with known concentrations of CPOs to be used for acute and chronic toxicity testing on the two species. The resulting data was used to develop dose-response curves that could be compared to those developed earlier from the wastewater samples collected at the land end of the wastewater pipe.

The second year of the study included long-term dye release experiments in conjunction with caged animal experiments. Using American oysters and eggs from the common killifish (*Fundulus heteroclitus*), dye concentrations in the water column were measured to estimate exposure concentrations for the caging experiments. The purpose was to evaluate if shellfish placed close to an outfall near the ocean bottom were adversely affected. Two hundred individuals were examined prior to deployment to establish a baseline of their health and condition. Half the animals were retrieved after 15 days and
the other half were retrieved 30 days after placement near the outfalls. Oyster growth, gonad (sexual organ) development, and the presence of lesions were evaluated in retrieved specimens. The survival of the developing fish eggs was also evaluated.

**DILUTION STUDIES**

During the first year, a numerical model was developed to predict the distribution of the discharged wastewater in the ocean receiving waters. This new model accounts for the hour-to-hour variability in the rate of discharge through the outfall, for the variability of the currents and temperature distribution in the receiving waters, and for the different configurations of the outfall pipes (e.g., ports along either a straight or Y-shaped diffuser). Because this newly developed model responds to variations in discharge and receiving water properties, it has been labeled a “dynamic mixing” model to distinguish it from earlier models that were for “steady-state” conditions, in which all properties of the discharge and receiving waters are held at constant values. This new model relies on a previously developed, well-verified initial dilution model to determine the height to which the wastewater plume will rise above the outfall and its vertical thickness. Measured or simulated currents at the height of rise of the plume are then used to move the wastewater plume away from the ports on the outfall diffuser. The numerical model also includes a component that provides for the prediction of the lateral spread of the wastewater plume due to other processes as it is being transported by the currents.

In order to use this dynamic mixing model for long-term simulation of discharge through ocean outfalls, it is essential that its predictive capabilities be carefully assessed by comparison with field observations. The second year of this study included the collection of data designed to both calibrate (fine-tune) and verify (does it accurately reproduce real-world conditions?) the dynamic mixing model. Fifteen-day dye release experiments were conducted at both the South Monmouth Regional Sewerage Authority outfall off Belmar, NJ and the Cape May County Seven Mile Beach outfall off Avalon, NJ. The distribution of dye, representative of the distribution of wastewater discharged through the outfall, was measured throughout the receiving waters from within the immediate vicinity of the outfall to several miles away from it. The distribution of current speed and direction at 1.6 ft intervals throughout the water column were measured using state-of-the-art, acoustic Doppler current profilers. The distributions of salinity and temperature were determined at 0.3 ft intervals in the vertical using a very high-resolution salinity, temperature, and water-depth recorder.

The combination of the observations of dye distributions by continuous sampling and the nearly complete characterization of water column properties provided ideal information for verification of the dynamic mixing model. To our knowledge these data are among the most comprehensive ever acquired for studies of the performance of ocean outfalls. In addition, three months of long-term current data (four weeks during the long-term dye release in the summer and six weeks during the fall) were collected at the South Monmouth outfall using a bottom-mounted acoustic Doppler current meter. Additional long-term current data for all seasons will be collected at South Monmouth and at the other two outfalls.
RESULTS TO DATE

Toxicological Studies

Shrimp bioassays were performed in the laboratory using wastewater samples from the outfall pipe at the beach and ocean samples of known wastewater concentration for both outfalls. The results of these tests showed high shrimp mortality and low egg reproduction in the wastewater before discharge at both plants. The ocean samples of diluted wastewater from both plants showed no effect on the survival or egg production of the shrimp at wastewater dilutions of 77:1 and 54:1 for South Monmouth and Seven Mile Beach facilities, respectively.

The oyster and fish egg caged tests were run concurrently with the 15-day dye studies. At the near-field sites, cages were placed as close to the diffuser pipe as possible (50-100 ft) to insure that test animals would be exposed to high concentrations of the wastewater discharge. The far-field cages were placed approximately one mile downstream of the wastewater plume to serve as controls.

The results from these studies revealed interesting patterns. Juvenile oyster growth initially was increased significantly for the near-field outfall site when compared to the far-field site off Seven Mile Beach, but over 30 days the far-field oysters increased in growth to significantly larger size than the near-field oysters. This suggests that over-time the wastewater may inhibit the growth of the oysters by bioaccumulation of contaminants or other physical factors such as temperature or salinity changes resulting from the wastewater plume. The oysters at both the near-field and far-field South Monmouth sites did not grow over the 11-day deployment. This lack of growth is possibly due to the cold near-bottom water temperatures, lack of nutrients, or other factors. At all locations there was no mortality of the oysters and sexual organ development was normal. Water collected in the near-field (acute) mixing area caused no mortality in the laboratory shrimp bioassays. This result was expected based on the previously developed dose response curves.

Dilution Studies

As expected, the wastewater plume from the South Monmouth outfall became trapped well below the ocean surface (typically, below 33 ft). The plume had a thickness of approximately 2 m. Although dramatic temperature differences were noted in the vicinity of the South Monmouth Outfall (e.g., a 10°C change from surface to bottom with an 8°C change occurring in the lower half of the water column (see Figure 1)), there was a negligible change in surface-to-bottom salinity. At this location, the stratified water column significantly reduces dilution by preventing the wastewater plume from rising to the ocean surface. During the experiment, the trapped, submerged wastewater plume was mapped as it traveled along the coast.
Figure 1 — South Monmouth Outfall; July 23, 1999 Temperature Profile.

Although the wastewater plume from the Seven Mile Beach outfall was found to be trapped below the surface during the morning hours, the wastewater plume typically traveled through the water column and reached the surface later in the day. Whereas stratification was strong during morning hours, afternoon winds shifted and mixed the water column to the extent that the temperature stratification disappeared (Figure 2). Similar to conditions at South Monmouth, there was very little difference in surface-to-bottom salinity concentrations.

Figure 2 — Seven Mile Beach Outfall; August 9, 1999 Temperature Profile
The data collected during the long-term dye release experiments have been used as input to the near-field module of the dynamic mixing zone model. This model predicts dilution, height of plume rise within the water column and plume thickness. Preliminary model simulations displayed good agreement with field data. For example, on July 23, 1999 at SMRSA, the plume was observed trapped between 35 and 45 ft from the surface (i.e., plume thickness of 10 ft). The model predicts a trapping level of 30 ft with a plume thickness of 12 ft (Figure 3). The average dilution at the trapping level was observed to be 276:1, while the model predicted the dilution to be 250:1, or very close agreement.

The far-field module of the dynamic mixing zone model was also used to simulate the ocean outfalls. The far-field portion of the model showed excellent agreement with the measured dye concentrations. For example, on the July 23, 1999, the wastewater plume from SMRSA was mapped to a distance of approximately one mile south of the outfall. Mapping was performed by navigating the research vessel along tracks that are parallel to the outfall pipe while continuously measuring dye concentrations in the trapped wastewater plume. At the maximum distance from the outfall, the peak observed concentration was measured at 0.28 parts per billion. Preliminary model results predicted dye concentrations in this vicinity to be 0.27 parts per billion (Figure 4).

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3 A part per billion is the equivalent of placing one black marble in a barrel containing 999,999,999 white marbles.
CONCLUSIONS

A series of toxicology studies have been performed to examine the effect of the chlorinated wastewater on the marine environment. A variety of local species have been used for conducting toxicity testing on both wastewater collected in the outfall pipeline and in diluted wastewater collected in the open ocean. Caged animal experiments have also been conducted in the vicinity of two ocean outfalls. The results from these studies suggest that the wastewater may over time inhibit the growth of the oysters by bioaccumulation or other physical factors such as temperature or salinity changes from the wastewater plume.
The diluted wastewater that was collected from the near-field mixing area was not acutely toxic or deleterious to shrimp reproduction. Additional studies will be performed this year to verify some of the data collected thus far in this project.

A dynamic mixing zone model was developed to simulate the mixing of discharged wastewater in the near-field and far-field. Preliminary modeling results suggest that the model is capable of accurately predicting dilutions in the immediate vicinity of the outfall and at distances of up to two miles from the outfall. The model allows for the wastewater flow rate to be varied over time, as well as the ocean receiving water conditions, such as surface-to-bottom salinity and temperature and current speed and direction. The model is currently being calibrated and verified with the long-term data collected for this project. These data include two four- to six-week measurements of currents, two 15-day dye release experiments, and extensive data collected on the salinity and temperature stratification in the vicinity of the ocean outfalls.

With the additional toxicity studies and the refinement of the model based on the long-term data, the results of this study will provide useful information for managers to predict the effects of the chlorinated wastewater on marine organisms. The model will provide a better means of predicting residual chlorine concentrations in the near-field and far-field. When the mixing of the wastewater with the receiving water is thoroughly understood, methods to minimize the impact of the ocean outfalls can be developed.
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