6. IMPROVEMENTS TO THE ASSESSMENT

IMPROVING THE METHOD

- The accuracy of the methods used in the NEEA continues to improve for the future.
- Linkages between EPA's National Coastal Assessment and the NEEA program were examined.
- An indicator for socioeconomic/human use impacts to Barnegat Bay is described.
- An estuarine classification scheme, or typology, is under development.
- The method of evaluating eutrophic condition is being improved, especially for SAV and macroalgal abundance.

Jane Hawkey, University of Maryland Center for Env. Science

Developing a socioeconomic indicator will help researchers and managers understand how eutrophication impacts human uses such as commercial fishing.

Developing methods which accurately assess the eutrophic conditions of the nation's estuaries is a significant challenge, especially considering the huge diversity of estuaries present, their varying sensitivities to nutrients, and their diverse functional characteristics. With assistance from U.S. and international eutrophication experts, a set of methods has been developed over the past 16 years (see Bricker et al. 1999, 2003, 2004, 2006; Scavia and Bricker 2006; see *www.eutro.org* for more details), leading to those included in this assessment. While the established methods have provided a relatively reliable assessment of the Nation's estuaries, the NEEA continually seeks improvement.

Since the first NEEA assessment in 1999, two workshops have been held, with over 40 experts from across the nation participating in each (Bricker et al. 2004, this study). These workshops have provided an excellent opportunity to seek recommendations on how the methods can be improved. This chapter highlights some of the main recommendations made at the workshops and by survey participants.

Recommendation #1: The overall recommendation from both workshops was to develop a long term, coordinated eutrophication monitoring and assessment program to help managers address problems in coastal water bodies on a national basis.

Response #1: At present there is no comprehensive, national monitoring program which samples the same eutrophication indicators in all U.S. waterbodies. This fact makes a national assessment such as this one difficult to achieve. The NEEA team has worked with state, federal, academic, and non-governmental organization experts for the past sixteen years to identify the appropriate indicators to be used for this assessment. Mechanisms for coordinated acquisition of pertinent data from existing programs as well as national overarching data collection programs (e.g., Integrated Ocean Observing System, National Water Quality Monitoring Network, EPA National Coastal Assessment) are being discussed and developed. The team will work further to influence the development of standardized methods of indicator measurement.

Recommendation #2: Develop a strategy of reporting and meeting time frames, with the specific recommendation of providing periodic updates of the assessment.

Response #2: The NEEA team is reviewing the options for providing more frequent updates. Challenges currently being addressed include determining: (i) the most appropriate frequency for which the assessment should be repeated, (ii) how to make the assessment program sustainable when repeated at shorter time frames (the online survey form developed for this update is one example already undertaken), and (iii) the appropriate mechanisms for reporting assessment results when conducted at greater frequencies.

Recommendation #3: Develop a framework for increasing the accuracy and reliability of data entered into the survey.

Response #3: An inherent challenge of surveying the Nation's estuaries using data from multiple sources is accounting for the diversity of data quality entered. Quality and completeness of data entered is not only dependent upon the monitoring data available, but also upon the diligence of the survey respondents. The NEEA team will continue to improve data quality by developing the recommended framework. This framework will include factors such as: (i) providing detailed guidelines and protocols, (ii) an opportunity to obtain training and support, (iii) improving the methods used to assess eutrophic symptom expression (see below), (iv) developing tools for managers, and (v) making all of this available online.

Recommendation #4: Improve the accuracy of the macroalgae indicator by: (i) requiring a spatial coverage assessment, (ii) defining the thresholds at which macroalgal abundance is considered a eutrophic symptom responding to excess nutrients, and (iii) developing standardized monitoring protocols to enable better comparison of results.

Response #4: The current set of characteristics used to assess macroalgae symptom expression is insufficient to differentiate between naturally occurring levels of macroalgal abundance and those signifying eutrophication. The NEEA team, with regional and national experts, will work to improve the survey methods and elucidate eutrophic responses. This process includes the development of a list of macroalgae nuisance species that when present are indicative of eutrophic conditions.

Recommendation #5: Improve the submerged aquatic vegetation (SAV) indicator by including both spatial coverage and biomass, basing values on the absolute value of change in area rather than on percent change in area. The indicator should be able to account for losses before the survey period. For those systems which have not historically had SAV, a different indicator should be developed.

Response #5: With regional and national SAV experts, the NEEA team will refine the SAV indicator to address the shortcomings identified by survey respondents. Some methods under development look promising and will be considered as a starting point (e.g., using linear measures of shoreline with SAV in place of traditional area measures, Latimer et al. 2006).

Recommendation #6: Develop a classification of estuaries using physical and hydrologic characteristics to describe and group systems by their susceptibility and the type of eutrophic conditions expressed.

Giancarlo Cichetti, U.S. EPA

Abundant green algae in a shallow bay. One suggestion for improving the method is adding a spatial coverage component to the macroalgae indicator.

Response #6: NOAA has commissioned the development of a type classification as the first step in improvements to the method. A useful and functional typology appears achievable (see below for details about the development of this classification).

Recommendation #7: Establish the link between eutrophication symptoms and the loss of beneficial uses/aquatic life use through development of an economic/human use indicator.

Response #7: Few studies have linked human dimensions or socioeconomic cost to nutrient impaired coastal water quality. In response to this recommendation, NOAA commissioned the development of a socioeconomic indicator. Its application in Barnegat Bay is described in this chapter. Complementing the existing eutrophication indicators, the socioeconomic aspect will illustrate the impacts and potential economic losses to human uses of coastal systems as a result of nutrient related water quality degradation.

Recommendation #8: Establish a link between the NEEA eutrophic symptom indicators and EPA's National Coastal Assessment Water Quality Index indicators. Work with EPA on the establishment of nutrient criteria (particularly biocriteria) for estuaries.

Response #8: The NEEA team, with EPA, is exploring linkages between the two national assessment programs with the aim of identifying potential collaborations. An intensive comparison between the programs is planned for 2007–2008, whereby a full set of recommendations will be provided. Currently, the two assessments have been compared and contrasted (See next page).

COMPARING THE EPA NATIONAL COASTAL ASSESSMENT WITH THE NEEA

- Two programs (NEEA and NCA) assess estuarine condition at a national scale.
- Both programs indicate a similar present condition of moderate or fair, but NCA shows improvements since the early 1990s while NEEA shows no change in condition over the same time period.

Adrian Jones, University of Maryland Center for Env. Science

The NEEA uses information from a variety of sources, ranging from large-scale, fixed-monitoring stations to manually deployed instruments.

Given the widespread problems and possible long-term impacts of eutrophication in U.S. and global coastal water bodies, it is not surprising that two national assessments have evolved, one conducted by the Environmental Protection Agency (EPA) and the other (the NEEA) by NOAA. The goal of the EPA National Coastal Assessment (NCA) is to document ecological conditions and trends throughout the Nation's estuaries through assessment of water and sediment quality, benthic community and coastal habitat health, and fish tissue contaminant concentrations. This is a broader goal than that of the NEEA, which evaluates the status and trends of nutrients only in coastal systems, the causes of observed impairments, and predicts future conditions based on demographic changes and management implementation. The intent of the NEEA is to provide a basis for the development of management measures to protect water bodies from further nutrient-related degradation. The EPA's NCA includes a nutrient related index, the Water Quality Index (WQI), which is comparable to the NEEA overall eutrophic condition component. Since both assessment frameworks are national in scope, it was recommended at the 2006 NEEA workshop that a link be established between NEEA and NCA WQI, while still recognizing the different goals of each program. This chapter highlights the first step in establishing this link by comparing results of the NCA WQI and results of the NEEA assessment of overall eutrophic condition.

Table 6.1 compares the results of the NCA water quality index and the NEEA overall eutrophication condition rating. The time frame represented by the NCA comparison is approximately 5 years (1990–1996 vs 1996–2000 USEPA 2001, 2005) while that of the NEEA is 10 years (early 1990s vs early 2000s; Bricker et al. 1999; this study). The NCA results are based on a comparison of both regional and national area-weighted WQI scores using five component indicators: dissolved inorganic nitrogen (DIN), dissolved inorganic phosphorus (DIP), water clarity, chlorophyll *a*, and dissolved oxygen (Table 6.2). The indicators are all given equal weight in an index formulation. The NCA WQI data were collected at stations that were randomly selected, using the EPA Environmental Monitoring and Assessment Program's (EMAP) probabilistic sampling framework. Data were sampled

Table 6.1. Comparison of trends in nutrient related conditions for NCA WQI and NEEA overall eutrophic condition. Scale is 1 - 5 (1 = poor and 5 = good). Changes are reflected as (△) for improvements and (◯) for no change.

	NCA Water Quality Index*			NEEA Overall Eutrophic Condition**		
	1990–1996	1996–2000	Change	Early 1990's	Early 2000's	Change
National	2	3	△	3	3	◯
Northeast***	1	2	△	3	3	◯
Southeast	4	4	◯	3	3	◯
Gulf	1	3	△	3	3	◯
Pacific	1	3	△	3	3	◯

* NCA WQI methods changed between the 1990-1996 and 1996-2000 groups; the 1990-1996 scores were recalculated using the current methods.
**Note that NEEA uses only 90 of 141 systems for analysis; the 51 systems with unknown conditions in either or both years were not included in this comparison.
***NEEA results from the North and mid-Atlantic regions combined to calculate the Northest region.

once per year during a summer index period (June to October) which typically represents the time period of greatest observed nutrient-related impacts (USEPA, 2001a). The NCA sampling regime provides 90% confidence in the results for its condition indicator for the U.S. and subregions (i.e., states). The NCA water quality index for U.S. estuaries (national scores) typically includes the Great Lakes and Puerto Rico, but for this assessment only contiguous U.S. water bodies are included for direct comparison to those in the NEEA.

The NEEA results are based on regional and nationally weighted averages of the number of systems assigned a particular overall eutrophic condition level. This level is based on annual data for five indicators:chlorophyll *a*, macroalgal abundance, dissolved oxygen, nuisance/toxic blooms, and loss of submerged aquatic vegetation (Table 6.2). The dissolved oxygen, nuisance/toxic blooms, and loss of submerged aquatic vegetation results are given a higher weight as a precautionary measure, recognizing that they are indicative of more well developed nutrient-related degradation. While the North and mid-Atlantic are considered separate regions in the NEEA, they have been combined here and called the Northeast region for comparison to the region boundaries used by the NCA.

The national NCA survey results indicate an overall improvement in estuarine condition for the 5-year change analysis while the NEEA shows no changes for the 10-year change analysis. However, the most recent results for both surveys are the same, indicating moderate level conditions nationally. A detailed and statistically rigorous assessment of the WQI change is included in the National Coastal Condition Report III (in review). Regionally, the NCA results suggest improvement in all but the Southeast region, which remains unchanged. In contrast, the NEEA results suggest that conditions in all regions have remained the same since the early 1990s. Both NCA and NEEA identify the Northeast region (i.e., Chesapeake Bay and tributaries as the southern boundary and Maine systems as the northern boundary) as the most highly impacted region. The most recent NCA results report that 61% of the Northeast coastal area is rated as fair to poor (19% as poor, 42% as fair). Comparable NEEA results report that 79% of Northeast systems are rated moderate to poor (47% poor, 32% moderate).

Table 6.2. Indicator variables used for assessing the NCA Water Quality Index and NEEA overall eutrophic condition.

Indicator variable	NCA[1]	NEEA[2]
DIN	X	
DIP	X	
Water clarity	X	
Dissolved oxygen	X	X
Chlorophyll *a*	X	X
Macroalgae		X
Nuisance/toxic blooms		X
SAV loss		X

[1]The NCA does not weight the variables in the formulation of the WQI.

[2]The NEEA weights dissolved oxygen, nuisance/toxic blooms, and loss of SAV more heavily than chlorophyll *a* and macroalgae (*see text for explanation*).

While the comparability of the recent national results is encouraging, the variation in regional results and trends suggests that the differences between methods should be investigated further. In addition to the different indicators used by these methods, there are differences in sampling time frames. Furthermore, the NCA random stratified sampling program is designed to evaluate conditions for 100% of the estuarine water area, reporting results on a regional basis, while the NEEA is designed to evaluate conditions within individual water bodies, representing greater than 90% of continental U.S. estuarine area and greater than 90% of freshwater discharge to the U.S. coastal zone. These results can be summarized into regional and national perspectives.

This brief comparison of the national results is encouraging. A more detailed comparative study planned for 2008 should elucidate reasons for differences between the programs, and an approach for using the best of both for future assessments.

DETERMINING TYPOLOGY

Robert W. Buddemeier, Kansas Geological Survey; Stephen V. Smith, CICESE; Suzanne B. Bricker, NOAA;
Dennis P. Swaney, Cornell; Susan D. Dunham, UNC; Bruce Maxwell, Swarthmore College

> Typology is a type classification of systems, determined by and grouped according to their sensitivity to nutrients and functional characteristics.

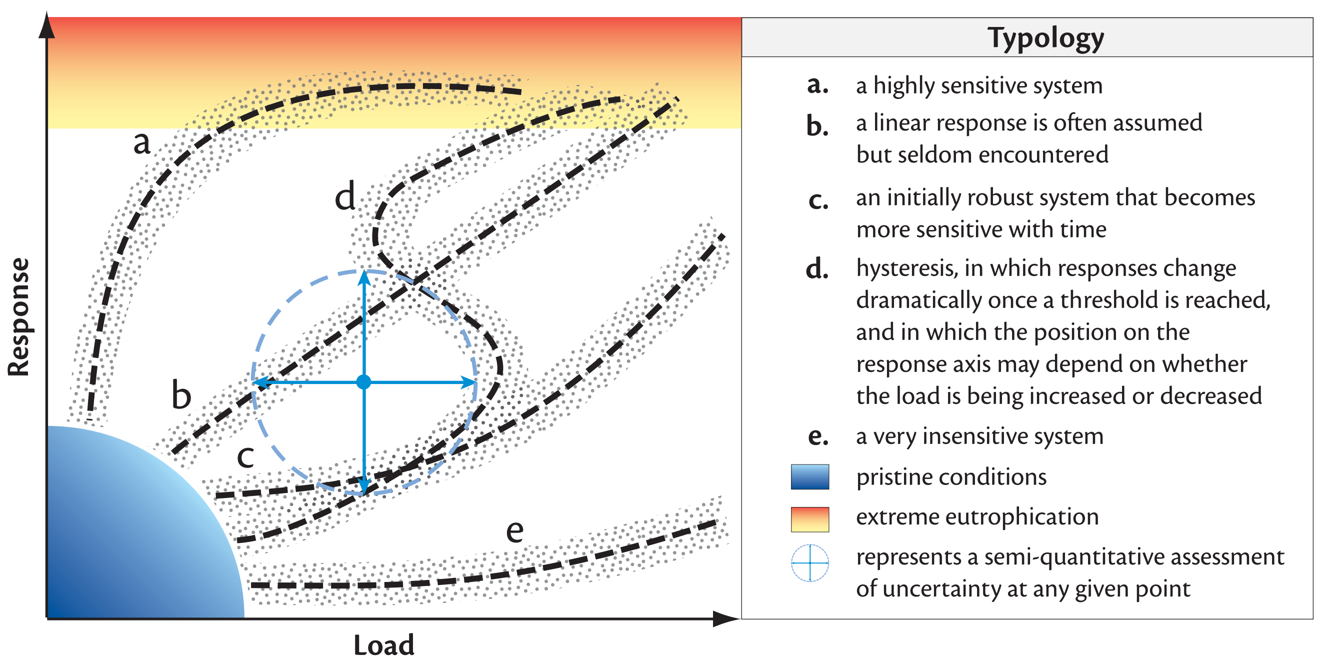
Type classification of U.S. estuaries is motivated by the need for two types of information — the sensitivity (or vulnerability) of specific estuaries or classes of estuaries to nutrient addition and the similarities between estuaries. Sensitivity reflects the degree of eutrophication (or the severity of eutrophication symptoms) to be expected for a given nutrient load. Similarity analysis identifies groups of estuaries that are similar not only in their sensitivities, but also in the functional characteristics contributing to this sensitivity. Such groups will have members subject to a variety of conditions and stressors, providing a broader perspective on the range and nature of responses. Also, similar systems can presumably be addressed with similar management approaches, permitting the transfer of knowledge and experience, and economies of scale — the primary goal for type classification in the NEEA.

The eutrophication assessment results presented in this report are a classification — a typology of eutrophication symptom intensity, categorized in the form of the classified variable, overall eutrophic condition (OEC). These scores, or classes, represent the system's observed nutrient-related water quality conditions. The OEC itself is a composite index, based on scores assigned to five eutrophic symptom variables. This index is very informative with regard to communicating the status of U.S. estuaries, and particularly in making useful comparisons among systems that may express a similar level of degradation, but with a different composition of symptoms.

Figure 6.1. shows a conceptual model illustrating the need for a functional typology. It illustrates various pathways that an estuary might follow in moving from an undisturbed natural state to a highly eutrophic condition as nutrient load increases. The blue arrows are intended to represent a semi-quantitative assessment of uncertainty at any given point on the graph. Sensitive estuaries are represented by the nearly vertical lines at the left side of the plot, while resilient estuaries with a high and robust assimilative capacity follow semi-horizontal pathways near the bottom of the plot. The highly curved paths illustrate possible situations for estuaries with critical thresholds — a phenomenon described in text box 1.

For management purposes, it is desirable to identify types including all of the estuaries likely to follow a particular envelope of eutrophication trajectories, regardless of where they are in terms of load or response when assessed. Ideally, this

Figure 6.1. Conceptual model of a few possible eutrophication trajectories as a function of nutrient load.



would help to identify critical thresholds before major transitions occur, managing systems to avoid more complicated, less easily reversed problems. An example in Figure 6.1. is system d, which reaches a threshold after which even reduced nutrient loads continue to drive increasing eutrophication. Similarly, a system following path c or e into the extreme eutrophic zone might return along path a or b as the load is reduced, requiring nearly pristine conditions and a substantial amount of time to recover.

A complication in the analysis is that our estimates of load necessarily contain a substantial uncertainty, and the measures of status ("response" axis position) do not necessarily identify consistent or calibrated differences. This is illustrated in Figure 6.1. by a data point with x- and y-error bars (blue arrows), showing a possible range of uncertainty in the plotted position of an observed estuary. Not only do an unlimited number of trajectories fit through the circle of uncertainties, but successive observations do not reliably define a path unless they are widely separated on the graph — usually not a desirable occurrence! Even with uncertainties narrowed, successive points can only support a linear extrapolation, which would probably miss upcoming thresholds. Details such as the duration and temporal variation in load are important to a comprehensive analysis, but it is not practical for a manager to get involved at this level.

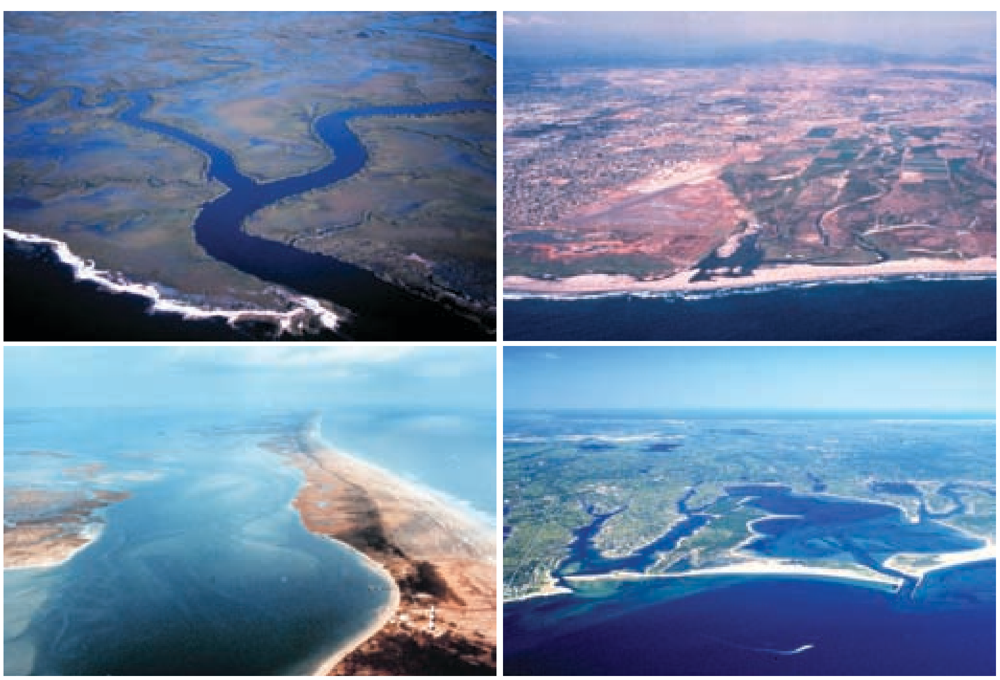
These complications and uncertainties illustrate the need for a predictive classification system for

Text Box 1: Critical thresholds and other factors effecting the response of coastal waters to nutrient loads[1]

Nonlinear responses and critical thresholds — Typically, nutrient budgets give a linearized picture of system response to changes in flows and loads, which can potentially predict how systems respond to relatively small changes. However, some systems respond in a highly nonlinear fashion, such as the loss of keystone species, with modest changes in load. Once a state change occurs, restoration can be extremely difficult.

Load per unit area of receiving waters — The capacity of a coastal system to process nutrient loads is related to the surface area of receiving waters. For systems with long residence times, load per unit area of receiving waters tends to determine ecological impact. Thus, small, poorly flushed coastal systems with small catchments and low runoff but large point source inputs are particularly vulnerable. Coastal lagoons often have these characteristics, as runoff is typically low and exchange with the ocean is restricted or intermittent. Urban sprawl with high loading is likely to increase the number of such systems. Small systems with large catchments may be vulnerable if flow is highly seasonal or diverted. Conversely, very large systems (coastal seas and large embayments) may show little broad-scale impact if loads per unit area are small and oceanic exchange is significant.

[1]Adapted and expanded from Le Tissier et al. (2006)



National Oceanic and Atmospheric Administration

The wide variety of estuary sensitivities and influences (physical, chemical, and hydrologic) calls for a determination of typology in order to better assess systems in a comprehensive, large-scale manner.

a diverse assemblage of systems, where detailed mechanistic understanding and the supporting databases are generally lacking. Systematic efforts to achieve this within the NEEA are summarized below.

Estuarine typology development

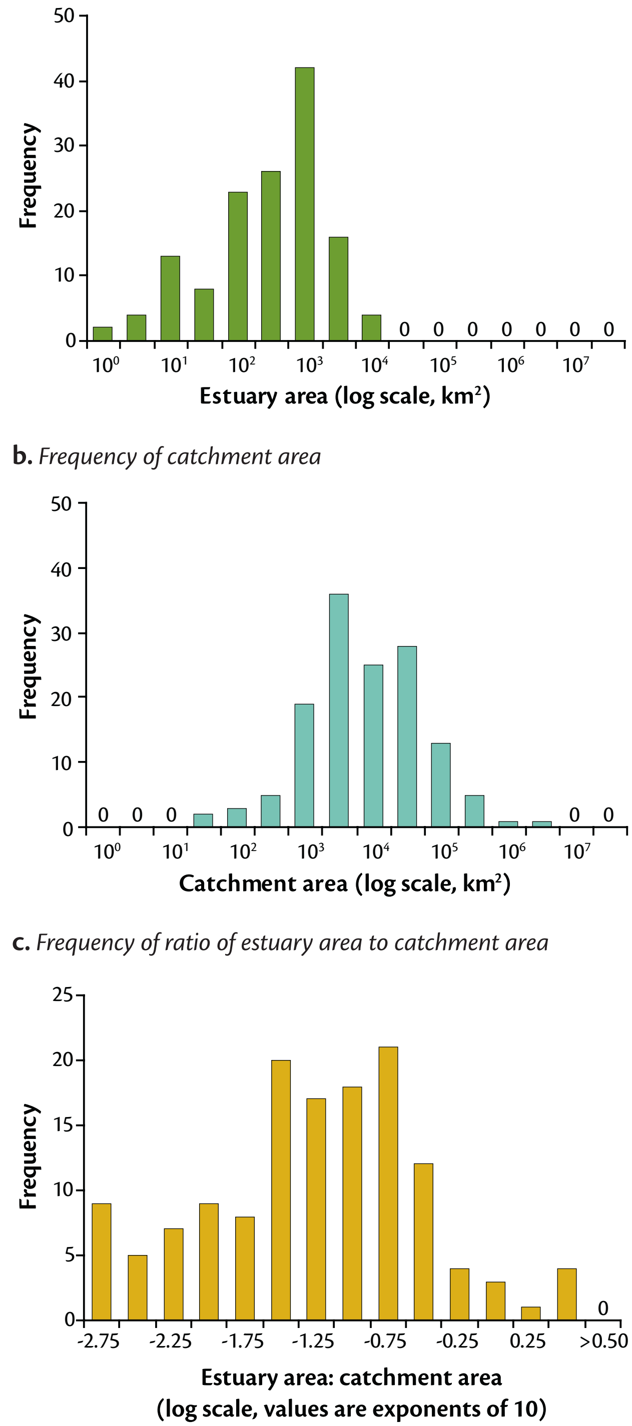
In order to meet the rigorous demands of developing a classification system to serve as a proxy for critical differences between systems, a process of geospatial clustering based on a broad spectrum of environmental variables has been adopted. The basic typology approach used is that employed by the Land-Ocean Interactions in the Coastal Zone (LOICZ) project — a joint effort of the International Geosphere-Biosphere Program (IGBP) and the International Human Dimensions Program (IHDP). The clustering tool groups systems based on their similarity with regard to selected biogeochemical characteristics. LOICZ aims to understand the role of the global coastal zone in natural biogeochemical cycles of the planet, and the degree and significance of its alteration by humans.

Due to the need to compare and integrate information across diverse coastal systems ranging from well-studied to essentially unknown, LOICZ-related tools and a linked global typology database (http://hercules.kgs.ku.edu/hexacoral/envirodata/hex_modfilt_firststep3dev1.cfm) have been developed. The tools, WebLOICZview (palantir.swarthmore.edu/loicz) and DISCO (narya.engin.swarthmore.edu/disco), are web-based geospatial clustering applications. DISCO, a second-generation application, offers a variety of user-controlled options including: (i) supervised and unsupervised k-means clustering, (ii) fuzzy k-means clustering, (iii) dataset manipulation, (iv) cluster comparison and stability evaluation, and (v) plotting of color-coded cluster points by geographic coordinates or on any two-variable plot from the dataset. Developments, applications, and findings of the first phase of the LOICZ project are described in Crossland et al. (2005).

The original LOICZ approach was designed for global application, with the recognition that it would be applied to many data-poor regions. The NEEA effort had both requirements for more refined and specific assessment products, and the advantage of working in a relatively data-rich region. This made possible a U.S. database more detailed than the LOICZ global database, and a prototype database of the U.S. estuaries and their watersheds was developed. The database consists of estuary and catchment variables assembled from available data sources, plus systems specific indices and composite variables created

Figure 6.2. Size distribution of estuarine and watershed areas included in the NEEA.

a. *Frequency of estuary area*



a. Histogram of $\log_{10}$ of estuary areas (km²); **b.** Histogram of $\log_{10}$ of catchment size (km²); **c.** Histogram of $\log_{10}$ of the Estuary/Catchment area ratio.

specifically to support the functional typology effort. Text box 2 (*see next page*) presents examples and discussions of some of the key factors known to influence estuary response to loading. In addition to calculated estimates of some of these factors (load/area and exchange time), the database offers a selection of geomorphic, hydrologic, and other variables which influence characteristics such as load and exchange time.

Workshops were held to evaluate and upgrade both the database and the DISCO tool, and to enlist the expertise of the estuarine scientific community in developing and testing a methodology for the group of systems included in the NEEA (Figure 6.2. shows some physical characteristics of these systems). A number of promising formulations of an estuary classification system were developed; one example is shown in Figure 6.3. In this case, variables used were estuary depth, percent of the system's mouth that is open to exchange, freshwater input, tidal range, and average temperature. These factors are directly relevant to residence time and are significant components of the factors listed in text box 2 (*see next page*). A large majority of the estuaries were contained in only six clusters, with groupings appearing reasonable to expert judgment. However, there was no reliable way to tell whether or not this classification actually reflected functional, mechanistic similarities suitable as a basis for management.

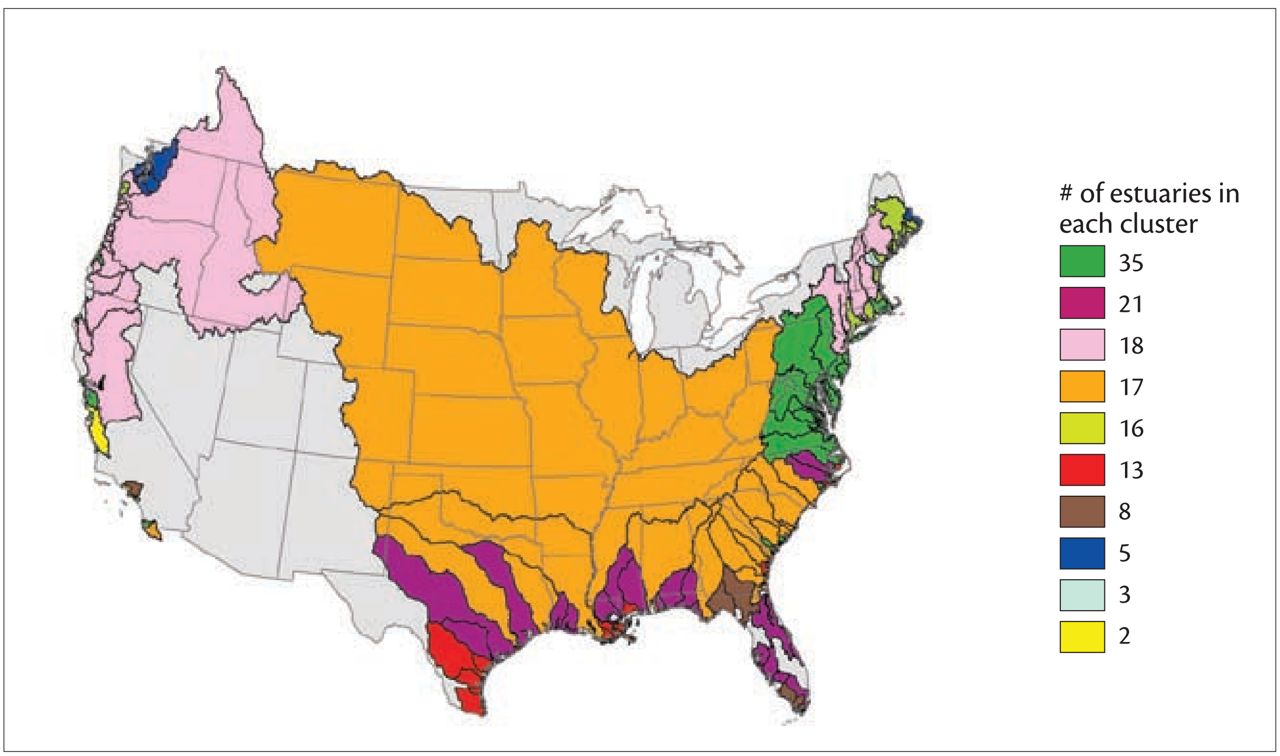
Needs and challenges

The efforts named above have revealed (or reinforced) several critical issues. To evaluate clusters in terms of function, a larger amount of quality data is needed for several classes of features. Some of these, such as the variability or seasonality of freshwater inflow, are partially addressed by the existing database and can be improved with only modest effort. The four categories that appear most important for improvement are:

- Ecosystem and biogeochemical function indicators;
- Measures of stratification;
- Characterization of system response to load; and,
- Interaction between changes in hydrology and load history.

These variables would permit consistent and more precise placement of datapoints on a practical version of Figure 6.1, and would provide more focus on the critical time dimension.

Figure 6.3. Example of estuarine classification. Estuaries were classified based on depth, percent of mouth open to exchange, freshwater input, tidal range, and average temperature.*



*Note geographic groupings, and that 85% of the systems are described by six clusters.

Text Box 2: Key factors affecting the response of coastal waters to changing nutrient loads[1]

Residence time — vulnerability to nutrient loads increases with longer residence times, determining the capacity of internal biogeochemical processes to transform and retain nutrients. Residence time is determined by the relative interaction of riverine flows and flushing by marine exchange. Systems with short residence times (days) tend to reflect the biogeochemical state of the dominant boundary (river or marine). If river flows dominate, most of the load is exported to the adjacent sea. With long exchange times (weeks to months), inorganic nutrients can be transformed into organic matter (autotrophic), or conversely, can transform organic matter to inorganic nutrients and carbon (heterotrophic). With little exchange, internal nutrient sinks and carbon may dominate (through denitrification or burial). Such systems are likely to be sensitive to changes in loads. Exchange times may change from days or less during floods, to months during the dry season.

Vertical Stratification — Vertically stratified systems are more likely to show adverse eutrophic symptoms from positive feedback. Stratification inhibits vertical mixing, restricting oxygen supply to bottom waters and sediments, and increasing nutrient availability (positive feedback). As nutrient supply and organic matter increase, sediment respiration also increases, further depleting dissolved oxygen. As bottom waters become hypoxic or anoxic, changes in sediment chemistry and microbial processes lead to reduced denitrification efficiency and desorption of phosphate bound to sediments, resulting in further increases in nutrient supply, or a reduction in nutrient sinks.

Relationship between nutrients and freshwater flow — The degree of correlation between load and freshwater flow is also crucial to estuarine response. If load is disconnected from riverine flow (e.g., sewage, large atmospheric deposition, or oceanic inputs), estuarine response is expected to be different from cases where loads vary closely with flow (e.g., river dominated systems). Estuarine responses are frequently coupled to residence time through several mechanisms, both biotic and abiotic (Nixon et al. 1996; Howarth et al. 2000; Smith et al. 2005a; Swaney et al. in press). When river discharge controls residence time, it affects the estuarine response indirectly through these mechanisms as well as by the nutrient load. Low flows can result in longer processing times of lower loads; high flows in shorter processing times of higher loads. When nutrient loads are independent of discharge, processing time is independent of load, and therefore intermediate responses could be expected (greater processing at high loads, less processing at low loads).

[1]Adapted and expanded from Le Tissier et al. (2006)

Ecosystem and biogeochemical function

Biogeochemical function (e.g., nitrogen fixation, denitrification) can be estimated using the LOICZ biogeochemical budget methodology (Smith et al. 2005a); some estuaries have already been characterized in this fashion. Completion of the estuarine budget dataset would require additional effort, and in some cases, probably additional data. For example, in some arid systems, the cap on salinity at oceanic values, with no allowance for hypersaline (net evaporative) systems, needs to be replaced with actual values. Information on communities whose responses may be particularly telling (e.g., macrophytes) is probably available for many of the Nation's estuaries, but has not been collected in a consistent format or location.

Stratification

The EPA has supplied a database of georeferenced surface and bottom salinity measurements, which permits direct determination of stratification; these data are being evaluated and processed for inclusion in the database. In addition to the stratification itself, it appears that dataset may also support classifying the estuaries by salinity zone, which will help provide more precise system characterizations.

System response to load

While the NEEA provides a large national dataset concerning the overall status of eutrophication in U.S. systems, this typology is not a particularly useful basis for further statistically-based mechanistic typologies. Although the component symptoms all reflect estuary conditions resulting from nutrient loading, these symptoms are not necessarily equivalent or interchangeable in their relationships to estuary function. The relative intensity of the symptoms reflects functional and structural differences between how systems condition their responses to changing nutrient loads. The five-class composite index (OEC) is derived by a quantitative (although subjective) method of combining the symptom scores (*see Chapter 2: Approach*). The OEC probably cannot be treated as a well-defined continuous variable, however, as it is made up of non-equivalent component scores in varying proportions. This means that it can have multiple non-unique relationships to environmental characteristics, depending upon how the score is achieved. Its usefulness is very limited because combining the symptom scores tends to blur functional distinctions between systems.

Ideally, the typologies for the symptom expressions would be developed on the basis of system responses and estuary characteristics. This would leave open the options of combining individual typologies into a master classification system, or classification based upon the nature and intensity of dominant symptoms. However, basing typology upon these characteristics has not been practical for three reasons:

- The lack of quantitative, continuous measures of response which are comparable among systems;
- The small number of credible categories (three) into which the symptom scores are classified translates to little discriminatory power; and
- The number of total systems is too small for statistically robust analysis, and even smaller for systems which exhibit a useful signal for any individual symptom.

The last point (sample size) could partially be addressed by treating defined salinity zones as individual systems. This would not only permit more precise assignment of some of the characteristics, but it would also at least double the total number of (sub)systems considered, although zones may be small or lacking data in some estuaries. Because a few systems (e.g., Chesapeake Bay) are estuary complexes, they may have multiple salinity zones of some or all three types. In these cases, individual zones of the same type will almost certainly differ among themselves in terms of response.

As a basis for developing an analytical typology of eutrophication, desirable characteristics of a response variable include:

- A direct and mechanistically understood relationship to nutrient loading;
- Quantitative measurements;
- Comparability and availability of data for all estuarine systems;
- Extended time series, so that responses can be evaluated with respect to averages, trends, and variability of environmental factors; and
- A wide range of observable values (i.e., sensitivity and discrimination power).

When combined with ground-truth measurements, remote sensing observations of chlorophyll *a* have the potential to address the last four of the desired characteristics. Enhanced primary productivity is one of the major and generally well-understood outcomes of excessive nutrient loading.

Monthly composites of estimated chlorophyll *a* concentrations and turbidity values based on SeaWiFS images have been obtained from the NOAA Center for Coastal Monitoring and Assessment. The dataset covers the period September 1997 through November 2004, and provides a reasonably complete data series for 107 of the 141 estuaries. The satellite data are based on 1100 x 1100 meter pixels, so that small estuaries or estuarine sub-systems with one dimension less than several kilometers are typically lost when the images are masked to avoid land contributions. In addition, systems routinely obscured by clouds may not be reliably characterized.

Although the number of estuaries with satellite chlorophyll *a* coverage is a smaller subset of an already small sample, concentration estimates can be evaluated and assigned at the level of the salinity zones within the estuaries. If the zones (tidal fresh, mixing, and seawater) can be treated as separate systems, then the total number of systems is significantly expanded.

The satellite-derived estimates are in general agreement with the classifications derived in the original NEEA assessment. The chlorophyll *a*, overall eutrophication condition, overall primary symptom, and overall secondary symptom expression scores all vary in the same sense as the concentration groupings, and differences between scores tend not to be statistically significant. The reverse is also true; when systems are sorted by the chlorophyll *a* or eutrophication variables, the average satellite chlorophyll *a* values of the groups vary in the same sense, but with large, overlapping standard deviations.

Some of the reasons for the weak positive relationships among ostensibly comparable variables (e.g., chlorophyll *a* measured *in situ* and estimated from SeaWiFS remote sensing color data) can be


SeaWiFS, NASA

SeaWiFS imaging is a world-wide data resource, useful for scientists interested in observing global primary production and phytoplankton patterns.

identified from Figure 6.4, which also illustrates the recently acquired datasets and some challenges faced when using them in conjunction with the NEEA results.

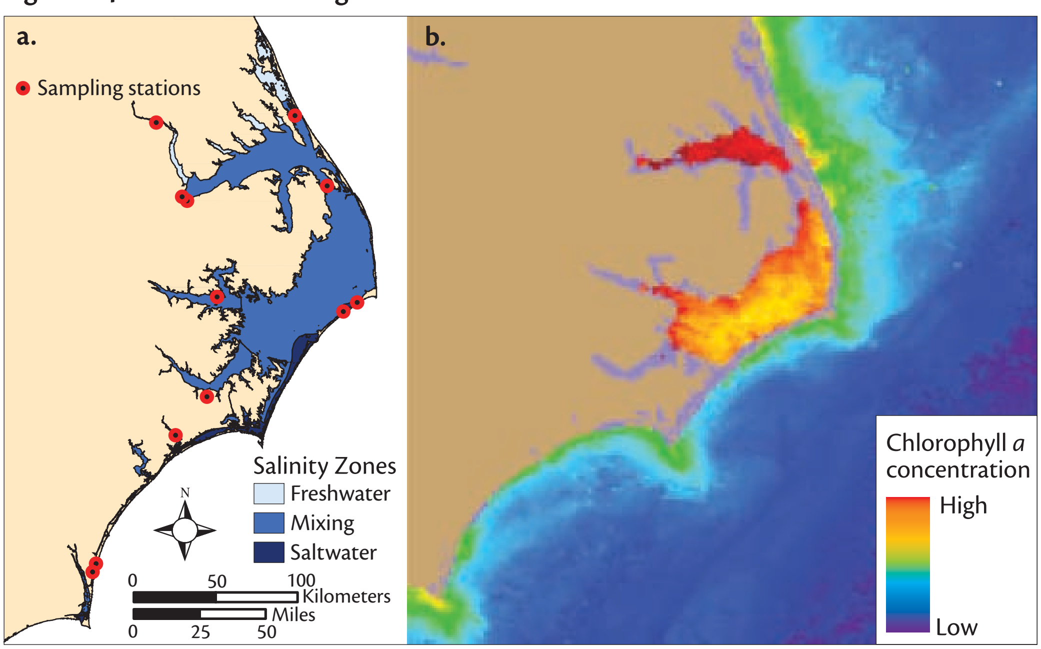
Figure 6.4.b demonstrates a tendency of chlorophyll *a* concentrations to increase as the shoreline is approached, and also that a substantial extent of the nearshore water is masked out of the chlorophyll *a* analysis. This implies (1) there is not a reliable overlap between the parts of the water bodies reported on by the two methods, and (2) that the assessment efforts do not include consistently georeferenced field locations of chlorophyll *a* determinations (these would permit straightforward geographic comparison with the satellite data). The salinity measurement points tend to be close to land and/or a zone boundary, so that stratification estimates are probably more relevant to the NEEA observations than to the satellite data. Bringing these three datasets together for combination or comparison could be addressed with higher resolution, large-area satellite data (if available), but the nearshore chlorophyll *a* estimates are more likely to be influenced by signals from the shallow bottom. Available bathymetric data combined with turbidity estimates would make it possible to exclude areas of probable interference, but this would require a substantial effort in data acquisition and processing. Greater attention to *in situ* chlorophyll *a* analyses from the areas of satellite coverage would provide important comparisons between the two approaches and help to identify the sources of present discrepancies.

The NEEA classifications are based on extreme conditions (e.g., lowest value dissolved oxygen, highest chlorophyll *a*), the spatial area of a salinity zone over which those values are observed, and the frequency of occurrence (Bricker et al. 1999). Location is not considered, and there is no quantitative standardization of the areas and durations of occurrence within or between estuaries. These factors complicate their comparison with the standardized remote sensing determinations.

Figure 6.5 graphically compares satellite and assessment score values for three groups of estuaries (identified by clustering mean maximum monthly satellite concentrations), with both exponential and linear data models. For this comparison, data were grouped into three best-fitting categories according to the magnitude of their concentrations. As the chlorophyll *a* concentration levels of SeaWiFS data only go up to about 21 µg L^{-1}, the difference in magnitude between these data and that of the NEEA is large. This makes finding a significant relationship difficult. While it is evident that a common signal is being communicated, the lack of significant difference is clear in the results of figure 6.5, as standard deviation ranges overlap.

Figure 6.5a compares the NEEA overall eutrophic condition (OEC) with the satellite-determined

Figure 6.4. Pamlico Sound region of the North Carolina coast.



a. Map of Pamlico Sound sampling stations **b**. A processed SeaWiFS image of the same region, September 1997. Note the masking (brown) that extends into the water bodies, and the smaller, unanalyzed estuaries. The salinity zones shown in **a.** are generally outside the region that SeaWiFS can measure.

chlorophyll a results, and figure 6.5b compares the OEC with the NEEA Chlorophyll a index. The plots show both exponential and linear data models. Because the OEC reflects responses other than chlorophyll *a* concentration (i.e., spatial coverage and occurrence frequency of macroalgae, dissolved oxygen, nuisance/toxic blooms, and SAV), there is no real justification for forcing the curve through the origin; a positive intercept on the OEC axis would be reasonable. Figure 6.5.c shows the NEEA chlorophyll *a* index compared to the SeaWiFS mean maximum monthly value; this relationship is forced through the origin on the assumption that zero chlorophyll *a* would be a common point. This assumption should be treated with caution, since the SeaWiFS chlorophyll estimates are not corrected for turbidity contributions, so the curve might have a positive x-axis intercept. The ranges of values correspond to about half of the index range and two thirds of the satellite value range, with the low concentrations not represented in either case.Although the correlation is apparently strong and positive, statistical significance and the sensitivity of the relationship between symptom expressions and satellite data, and between OEC and either SeaWiFS or NEEA chlorophyll *a* estimates are all low.

The positions of the data ranges on the plots tend to support the viewpoint that most systems are significantly impacted—there are clearly relatively few points falling in the low-concentration, low-impact ranges. This enables the separation of systems with probable recovery or prevention potential from those most likely to be irretrievably altered. Separation with the goal of prioritization is an important management tool, and one that may be addressed by a mechanistically-oriented typology.

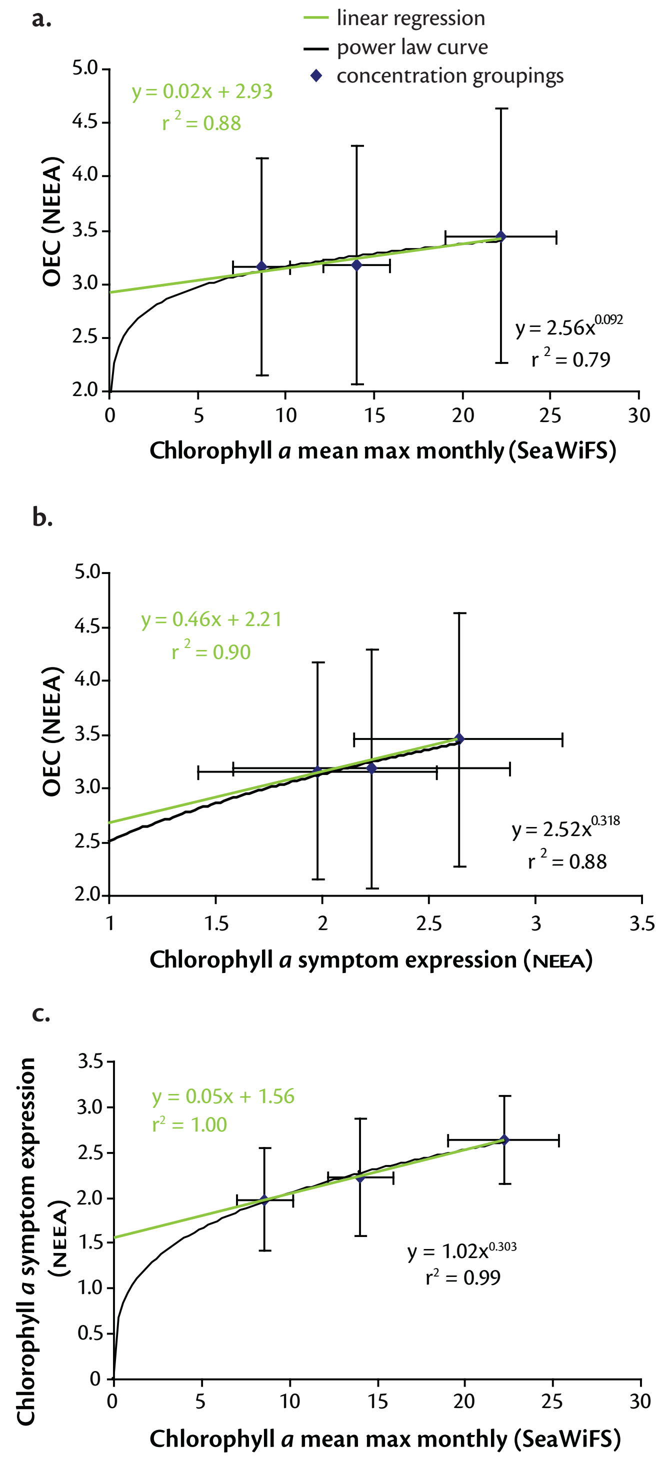
Discussion

Considerable progress has been made in classifying the estuarine systems, and the goal of a useful typology appears achievable. It will, however, require continued effort in terms of scientific analysis, data acquisition, and agency cooperation.

Biogeochemical budgeting or similar process assessments can be developed to provide additional data for vulnerability assessment, and to provide intermediate functional analyses, such as those at the global scale by Smith et al. (2003, 2005a). Those studies addressed the issue of loads rather than of sensitivity, and developed bivariate equations (Log Runoff and Log Population) that described nutrient loads with relatively high r^2 values.

Remotely sensed extreme chlorophyll *a* values appear to make a significant contribution to the assessment of eutrophication level and sensitivity, but more work is required to relate these data to existing

Figure 6.5. Comparison of chlorophyll *a* data, and estimates obtained from NEEA and SeaWiFS. Overall eutrophic condition ratings and symptom expressions are compared to clustered data values.



a. Comparison of OEC and SeaWiFS chlorophyll *a* concentration estimates. **b.** comparison of the NEEA OEC index (max. value = 5) with the chlorophyll *a* symptom expression. **c.** comparison of the NEEA chlorophyll *a* index (max. value = 3) with the SeaWiFS-estimated concentration. Error bars represent standard deviation from the mean.

assessment indices, or to modify their formulation in order to improve sensitivity and comparability.

It is noted that smaller estuaries are systematically excluded from the satellite chlorophyll *a* database because their dimensions are small relative to the pixel size. This needs to be compensated for in any management-oriented typology, since the small systems are more variable and vulnerable (Smith et al. 2005b), and thus more likely to be poorly characterized. Their high between-system variability is a significant management (and perhaps classification) issue, as it implies increased difficulty when extrapolating from one small system to another without a robust (and physically understandable) basis for the classification. As to the edge effects in the larger estuaries, this can be addressed either with higher resolution chlorophyll *a* determinations, or by better calibration of analytical results against the satellite estimates (so that small systems can consistently be assessed on a basis comparable to that used for the larger ones). Figure 6.2 illustrates catchment and estuary size distributions for the U.S. systems; a large fraction of the systems are below the 10^4-10^5 km^2 threshold identified by Smith et al. (2005b) for small catchments. These issues are the subject of continuing efforts to develop a typology which may be used to discriminate estuary and coastal water body types for use in the transfer and application of successful management approaches.

Acknowledgments:

DPS would like to acknowledge support from NOAA award NA05NOS478120 from the NOAA Center for Sponsored Coastal Ocean Research, Coastal Hypoxia Research Program, to D. Scavia, G. Helfand, R. Howarth, R. Alexander, and D. Breitberg.

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DEVELOPING A HUMAN USE INDICATOR FOR BARNEGAT BAY

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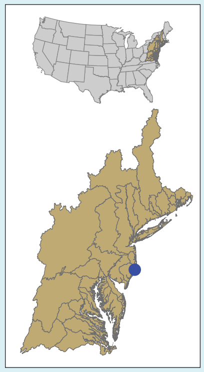
- As a further way to enhance the understanding of how eutrophication affects ecosystem health, a human-use indicator is tested in Barnegat Bay.
- This indicator allows for analysis of how eutrophication affects human populations, whereas most impact assessments consider only the human impact to an environmental system.



The traditional approach to assessing coastal eutrophication and related water quality issues has focused on causes stemming from human activities. Recently, there has been great interest in looking at the issue from a different perspective: documenting how eutrophication and water quality affect human uses of coastal waters and estuaries (USEPA 2005). This chapter highlights progress in the development of an indicator for one of the many possible impacts to human uses of an estuary. This indicator complements the NEEA method and provides a more complete picture of the system.

Given the complex nature of eutrophication, there are a variety of potential human-use impacts, including impacts to commercial and recreational fishing, fish consumption, swimming, boating, aesthetics, and tourism (Bricker et al. 1999, EPA 2005). Recreational fishing is an important activity in most estuaries and one that is often directly impacted by eutrophication. Lipton and Hicks (1999, 2003) demonstrated that recreational fishing for striped bass in Chesapeake Bay and the Patuxent River sub-estuary was negatively impacted by low bottom dissolved oxygen levels. Another recent study linked changes in recreational fish catch rate for three species (bluefish, striped bass, winter flounder) to changes in bottom water dissolved oxygen in 12 Gulf of Maine and mid-Atlantic systems, with striped bass being the most affected of the three (Bricker et al. 2006).

Through the Marine Recreational Fisheries Statistics Survey (MRFSS), the National Marine Fisheries Service regularly conducts surveys of recreational fishing activity and success in most U.S. estuarine systems. This fishing data can be combined with water quality monitoring data and analyzed to determine whether recreational fishing catch rates are related to eutrophic conditions within particular estuarine systems. When a significant relationship is found, recreational fishing catch rates, with appropriate adjustments for other influencing factors, can be used as an indicator of human use impairment due to eutrophication. With additional data and analysis, a dollar value estimate of lost economic welfare can be estimated directly using techniques such as travel cost and random utility models (Herriges and Kling 1999). Alternatively, with a large number of recreational fishing value studies available in the literature, an approximation of lost economic value can also be determined using benefits transfer (Walsh et al. 1992).

Barnegat Bay is an excellent candidate for the application of recreational fish catches as a human use indicator. Surrounded by a large population center, Barnegat Bay sees a lot of recreational fishing activity. Barnegat Bay is also frequented by a variety of recreational species targeted by fishermen. According to MRFSS data, the three species most targeted on Barnegat Bay fishing trips are summer flounder (42%), striped bass (19%), and bluefish (7.5%). The following analysis focuses on these three species.

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Recreational fishing is just one activity to be incorporated into a human use indicator for estuaries.

Methodology

Individual recreational fishermen can be thought of as biased samplers of the estuarine fish population. They are biased in that they are not randomly sampling the population, but using their knowledge of past fish catches, seasonality, weather conditions, and other factors to increase the probability they will catch fish. They are also not standardized samplers; some are more experienced and better at using this information than others. The catch of these individual fishermen is modeled as a function of their fishing avidity, captured by their response to the MRFSS question asking how many times they have gone fishing in the past year (FDAY). The catch rate during the individual fishing trip is also a function of the migratory and seasonal nature of the targeted species. To measure the fluctuating stocks available to fishermen, the catch rate is averaged (catch per hour fished) over all fishing trips and over years for each species in a month (MCR). The other factors potentially affecting catch rate are related to environmental conditions at the time of fishing, such as salinity (SALIN), water temperature (TEMP), chlorophyll *a* concentrations (CHLORA), and dissolved oxygen (DO), which is the variable most linked to eutrophication. The model used to estimate catch is then:

(1) $TC_{i,j,m} = \alpha + \beta1\ MCR_{j,m} + \beta2 HRSF_i + \beta3 FDAY_i + \beta4 SALIN_m + \beta5 CHLORA_m + \beta6 TEMP_m + \beta7 DO_m + \beta8 (DO_m)^2 + \beta9 (DO_m * TEMP_m)$

where $TC_{i,j,m}$ is the expected catch of recreational angler i, fishing for species j (striped bass, bluefish, or summer flounder) in month m, and $HRSF_i$ is the number of hours fished on the fishing trip by angler i. Parameters to be estimated in the statistical model are represented by α and β1–β9.

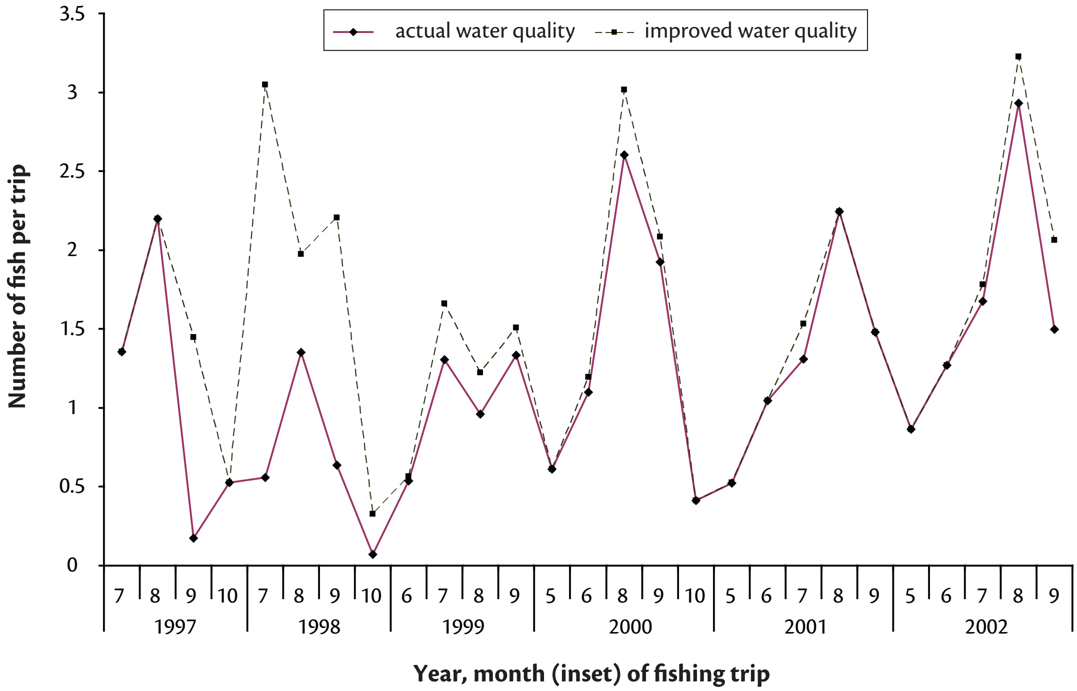
Recreational fishing data collected in the MRFSS in New Jersey was obtained for 1993–2002. Only fishermen intercept survey data, not the telephone interviews, were used for this analysis. Intercept data records catch rates for individual fishermen on a specific fishing trip. For this analysis, only fishermen that indicated striped bass, winter flounder, or bluefish were their primary or secondary target species were selected. Many fishermen indicate that they are not targeting a specific species, and these are excluded from our study. To determine whether a New Jersey fisherman was fishing in Barnegat Bay, a geographic information system analysis was used to select only the intercept sites that fell within the Barnegat Bay boundary.

Salinity, temperature, and dissolved oxygen data for Barnegat Bay sampling stations were averaged by month and year and then matched to the month and year of the fishing trip from the MRFSS data. Because Barnegat Bay is considered a relatively shallow and well-mixed estuary, data measurements at different depths were averaged.

Table 6.2. Parameter estimates from Poisson regression of striped bass, bluefish, and summer flounder recreational fishing trips in Barnegat Bay, NJ. An "*" indicates significance at 90% Confidence Interval.

Variable	Bluefish	Striped Bass	Summer Flounder
Intercept	0.0583	-10.2257	-32.3057*
Hours fished	0.1452*	0.2516*	0.2056*
Days fished in 12-month pd.	-0.0023*	0.007*	0.0046*
Mean catch rate	3.5274*	10.8873*	1.9921*
Mean DO	0.7391	1.6092	7.481*
Mean DO^2	-0.0956	-0.0966	-0.4423*
Mean salinity	-0.0105	-0.0129	-0.0746*
Mean temp.	-0.0818	0.0385	0.5802
Mean DO x mean temp.	-0.0015	-0.0093	-0.0622
Mean Chl *a*	0.0208*	0.0774*	-0.1356*
Number of observations	446	939	458

Figure 6.6. Barnegat Bay monthly average summer flounder actual catch per recreational fishing trip (solid line), and predicted catch rates under improved water quality (WQ) conditions (dashed line).



Results

Equation 1 was estimated for each of the target species using a Poisson regression due to the fact that there are a large number of fishing trips for which catch was zero. Estimation results are given in Table 6.2. The two water quality measurements related to eutrophication that are included in the model are dissolved oxygen and chlorophyll *a*. Chlorophyll *a* levels had a significant and positive impact on bluefish and striped bass recreational catches and a significant, but negative impact on summer flounder catches.

Dissolved oxygen is incorporated into the model in a quadratic form allowing for diminishing marginal improvements in catches as dissolved oxygen levels increase. An interactive term between dissolved oxygen and water temperature is also included in the model. None of the terms containing dissolved oxygen had a significant impact on either striped bass or bluefish recreational catches. For summer flounder, both the DO and DO^2 parameter estimates were significant at the 90% confidence level, but the dissolved oxygen-temperature cross-product term was not significant.

Based on the results in Table 6.2, it appears that neither striped bass nor bluefish are good indicators of human use impairment due to eutrophication in Barnegat Bay. This does not mean that impairment is not occurring, just that the impact is difficult to detect with current data. For example, more spatially explicit analysis might reveal an impact not apparent from the aggregated catch and water quality data. Lipton and Hicks (1999) found this to be the case for striped bass in the Chesapeake Bay where catches linked to specific water quality stations were shown to be negatively impacted by low dissolved oxygen.

For Barnegat Bay, summer flounder, the most sought after species, is a good indicator of the human use impacts of eutrophication. The solid line in Figure 6.6 shows the average actual catch of summer flounder in a month for the period from 1997-2002. The statistical model was then used to predict summer flounder catches under different water quality conditions. Specifically, an upper limit on chlorophyll *a* concentrations was set so that they could not exceed the sample averages of 7.12 μg L^{-1}, and a lower limit on dissolved oxygen of 6.51 mg L^{-1}. The dashed line in Figure 6.6 represents the predicted summer flounder catches under these improved water quality conditions, and the distance between the two lines is the impairment due to eutrophication. In some months, the limits are rarely exceeded and there is no difference in expected catches. Overall, the average catch of summer flounder is reduced from the predicted average of 1.25 fish per trip to 0.92 fish per trip, a 26% reduction.

To illustrate the economic magnitude of the reduction in recreational fish catch due to eutrophication, some of the estimates made for mid-Atlantic fisheries by McConnell and Strand (1994) were examined. Using a Poisson regression and random utility model, they estimated that increasing catch rates of mid-Atlantic fishermen by 0.5 fish per trip increased the net value of the trip to the fishermen by $7.51-$8.13, depending on the month. The average catch rate for Barnegat Bay is increased by one-third or 67% of the McConnell and Strand rate. However, to adjust for diminishing marginal utility, 75% of the mid-point of the McConnell and Strand value is taken and adjusted to current (2005) dollars to yield an estimate of increased value per trip of $10.26. Given that 42% of New Jersey fishing trips target summer flounder and there were 5.9 million inland fishing trips in New Jersey (MRFSS data), we roughly estimate that eutrophication costs these fishermen an average of $25.4 million per year in net benefits.

This example demonstrates a method to determine the impact of eutrophication on an economic basis. Additionally, this indicator is generally transferable and the intent is to develop it for use as a nationally applicable indicator. However, before a full application can be made, similar analyses must be performed to determine the appropriate fish species to use in different systems around the country.

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