***IV. Eagle Directors' Manuscript:*** to be submitted to Environmental Health Perspectives**.**

**Title**
**Rationale for a New Generation of Ecological Indicators for Coastal Waters**


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**Running title:** New generation of coastal indicators

**Keywords (5-10 required):** coastal, ecological, estuarine, health, indicators, marine, nutrients, responses, stressors

**Acknowledgments** Although the research described in this article has been funded wholly or in part by the United States Environmental Protection Agency through cooperative agreements (R-82867501, R82867701, R82867701, R-82867601, R-82945801) to the University of Minnesota, University of North Carolina, Pennsylvania State University, University of California, and the University of Southern Mississippi, respectively), it has not been subjected to the Agency's required peer and policy review and therefore does not necessarily reflect the views of the Agency and no official endorsement should be inferred.

**List of abbreviations used in the manuscript with definitions:**

COHH - Centers for Oceans and Human Health

EaGLe - US EPAs Estuarine and Great Lakes coastal research program funded through its

STAR - Science to Achieve Results

EMAP - Environmental Monitoring and Assessment Program

HABs -harmful algal blooms

HPLC High performance liquid chromatography

HSI - habitat suitability indices

IBI - Indices of Biotic Integrity

LTER- Long Term Ecological Research

NIEHS - National Institute of Environmental Health Sciences

NRC - National Research Council

NRE - Neuse River Estuary

NSF - National Science Foundation

PDAS - photodiode array spectrophotometry

PS - Pamlico Sound

TMDL - Total Maximum Daily Load Program

US EPA - United States Environmental Protection Agency

**Outline of Section Headers**

**Abstract:** The population in the coastal zone has been burgeoning over the past few decades. Half of the worlds population lives within 100 km of the coast and that number is expected to increase by 25% in the next two decades. As a result of this population explosion, coastal ecosystems are at serious risk. Increased nutrient loading leads to hypoxia, algal blooms, and damage to coastal fisheries. Sea level rise leads to loss of wetlands and salt water intrusion into coastal aquifers. Increasing development, recreation and industry lead to more pollution and ecological disruption. Monitoring coastal resources has traditionally been done on a stressor by stressor basis. Typical indicators of ecosystem health were nutrient loading or dissolved oxygen but the linkages from pollutant (stressor) to effect were difficult to make. To fully understand and measure the complexities of the coastal systems, a new set of ecological indicators that span the realm of biological organization from genetic markers to entire ecosystems and are broadly applicable across geographic regions need to be developed. Along with these indicators appropriate reference conditions from which to benchmark change need to be developed. The use of photopigments as indicators of the interactive effects of nutrients and hydrology is an example of this new breed of indicators. Additionally an approach that integrates a wide variety of taxa can communicate information on ecosystem structure and function such as habitat change, hydrological modifications, and water quality.

## Introduction

More than half the world's population resides within 100 km of the coastline (Vitousek et al. 1997, Culliton 1998); a percentage expected to increase over the next two decades (Stegeman and Solow 2002). The coastal zone represents at least half the value of global ecological services (Costanza et al. 1997), and in economic terms, is the single most important source of fisheries as well as recreational and residential income worldwide (Jackson et al. 2001a; Ray and McCormick-Ray 2004).

Human development of coastal watersheds has greatly accelerated environmental pressure on downstream estuarine and coastal ecosystems. The symptoms include deterioration of water quality; loss of habitat and biodiversity; beach closings; nutrient, sediment, and toxic inputs; hydrological and habitat alteration; increased fishing pressure; and an overall decline in the livability of the coastal zone (Nixon 1995; Rabalais et al. 1996; Richardson 1997; Hobbie 2000; NRC 2000; Boesch et al., 2001). While the pressure on coastal systems is enormous, they also pose special challenges when assessing their responses. Coastal systems are hydrologically complex and are among the most susceptible to global climate change because of direct effects on water levels and intensified pressure gradients along coastal regions (Jackson et al. 2001b). Sea levels have increased in the past century and greater changes are predicted over the next 50 years. These changes affect coastlines and will dramatically increase saltwater intrusion, worsening saltwater effects in freshwater coastal aquifers, and displacing agriculture in coastal watersheds (Jackson et al. 2001a).

Non-point source nutrient pollution from coastal watersheds is also a major problem and has contributed to moderately to severely degraded pollution in more than 60% of coastal rivers and bays (Howarth et al. 2000). Increases in nitrogen and phosphorus loading (discharge) to coastal ecosystems have led to disruptions of basic ecological functions, including rising frequencies and proliferation of harmful algal blooms (HABs), an increase in oxygen depletion (hypoxia) events, leading to major damage to coastal fisheries and biodiversity (NRC 2000; Jackson et al. 2001; Sundareshwar et al. 2003). In addition, nonpoint sources of toxic substances including agricultural chemicals and urban runoff have impaired the human use of numerous coastal watersheds as well as the habitat quality for aquatic life (Kuivila and Foe 1998; Detenbeck et al. 1999).

It is increasingly evident that ecosystem and human health are intricately linked (Stegeman and Solow 2002). For example, HABs also cause many diseases in humans, which range from acute neurotoxic disorders to chronic and persistent diseases such as amnesic shellfish poisoning and chronic liver disease caused by the cyanobacterial toxins (ref.). These diseases may result from consumption of contaminated seafood or inhalation of toxins entrapped in sea spray. Moreover, the distribution and frequency of HAB events have increased along US coastlines over the last 30 years (ref.). People are also exposed to water-borne diseases by recreational contact and their incidence is increasing worldwide. The principal agents of diseases are bacteria, viruses and protists. Bacteria include native marine organisms such as *Vibrio vulnificus* and *Vibrio parahemolyticus*, but most are human or animal-derived pathogens from sewage and runoff (ref.). To improve our knowledge of the connections between coastal conditions and human health, the NIEHS and NSF have initiated establishment of Centers for Oceans and Human Health (COHH).

Given the importance of coastal systems and their response to increasing pressure, quantitative measures of ecological conditions in coastal regions are absolutely essential for comparative assessments and detection of change. Because it is resource-intensive to obtain a comprehensive assessment of condition, the most expedient approach has been to use broadly applicable indicators such as water clarity, nutrient or contaminant loads and levels, and various biodiversity measures such as species richness (NRC 2000; Whittier et al. 2002). The development of indicators has evolved substantially over the past 100 years since crude measurements such as the canary in the mines or the saprobian index (Kokwitz and Marsson 1908). Today, measurements of environmental condition are both more sophisticated and applicable across a variety of scales which include three major dimensions: space, time, and biological organization (Cottingham 2003). Examples of such indicators include diagnostic photopigments of algal functional groups to assess eutrophication (Cottingham and Cooper 1998), biochemical and genetic indicators of toxicant exposure and stress (McCarthy and Shugart 1990; Huggett et al. 1992; Anderson et al. 1994), indices of biological integrity or other biological community responses (Karr 1981; Simon 2003), ecosystem and population modeling approaches (Gentile 2001), and landscape metrics (DeAngelis 1998; Poiani et al. 2000, Whittier et al. 2002). While these indicators represent impressive advancements in both science and technology, limitations to their widespread and integrated use exist (NRC 2000).

Here, we focus on defining these limitations and illustrate new approaches from our own efforts at improving indicator development and deployment. We define the coastal environment to include both marine and freshwater coastal ecosystems (Great Lakes) and their watersheds that are major sources for many stressors affecting these ecosystems. However, the concepts and approaches included here have applications worldwide. The impetus for this effort is US EPAs Estuarine and Great Lakes (EaGLe) coastal research program funded through its Science to Achieve Results (STAR) program. Contributors to this paper include investigators from US EPA and the four major coasts of the US (Atlantic, Pacific, Gulf of Mexico, and the Great Lakes).

This paper is not a comprehensive review of indicators, nor is it the final solution to developing integrative indicators of coastal environments. The needs and solutions are not unique to coastal regions. There are formidable needs for the development of indicators that are capable of detecting and diagnosing the signals of environmental condition over space and time at cellular, organismal, habitat, ecosystem, and regional levels. The explosion of technical and conceptual advancements in disciplines ranging from molecular biology to ecosystems ecology to remote sensing and geographic information systems provides a plethora of tools to nurture this new generation of indicators.

**Limitations of Current Coastal Indicators**

Most indicators were previously designed to provide specific information on local conditions such as water clarity, eutrophication, or broad-based snapshots of regional-scale water quality and habitat condition as exemplified by the Total Maximum Daily Load Program (TMDL) or the Environmental Monitoring and Assessment Program (EMAP) (NRC 1994; US EPA 1999). From these studies we know that large areas of the US coastal zones are impaired (Bricker et al. 1999; US EPA 2001; Environment Canada and US EPA 2001) and the TMDL program have identified many of the specific stressors within coastal watersheds that contribute to impairments (US EPA 1999). Because new techniques are available and the range of issues that indicators must address continues to grow, the development of these new techniques for improvement in environmental management is critical. We focus on the limitations of current indicators that are imperative to overcome for future advancements in their development (Fig. 1).

1) **Stress with Response.** Most current indicators of coastal condition are not linked with specific stressors and, hence, it is unclear what causes are reflected in a change in the indicator or what management solutions should be implemented to improve coastal conditions.
2) **Multiple Stressors.** Stressors in coastal ecosystems are diverse and originate from both anthropogenic and natural perturbations. Most current indicators are incapable of providing diagnostic information that separates the relative contribution of changes in coastal condition from diverse sources of stress.
3) **Space and Time.** Sources of stress to coastal environments operate over a range of spatial scales (e.g., square meters to entire landscapes) and time (seconds to decades). Current indicators are not always explicit in how they relate condition with stressors over these different scales.
4) **Reference Conditions.** The interpretation of the condition or change of an indicator is based on a comparison to a reference condition or benchmark. Frequently, these reference conditions are not defined, so judgment of condition or change in indicators is

limited and subject to considerable interpretation and debate.

**Linkages of Stress with Response**

Among the greatest limitations of many indicators of coastal condition is the lack of linkage with the cause or causes for change (Suter et al. 2002). The development and designation of indicators must first distinguish between measurements of disturbance or stress and the measurement of ecosystem response. Terminology in the literature has varied considerably, but recent reviews (NRC 2000; US EPA 2003a) have clarified this terminology considerably.

The most frequently employed approach to discern stress response relationships is the use of natural experiments or surveys in which varying levels of disturbance are identified over a gradient from relatively pristine to highly disturbed areas (Karr and Chu 1999). In these situations, it is often difficult to control the stress or independent variable because multiple factors often vary across any environmental gradient. As a result, inherent limitations in the scope of inference must be explicitly recognized. The quality of the response variable is also dependent on the strength of conceptual models used to describe the factors structuring the ecosystem and the extent to which anthropogenic disturbances influences that ecosystem. Detection of a response to a stressor is best accomplished by an experimental approach in which the stressor can be manipulated in frequency, intensity, duration, or extent. These experiments are often limited to laboratory situations, field microcosms, or mesocosms. A combination of a gradient design with field and laboratory experiments is a powerful approach for the initial phases of indicators development but as the application of indicators occurs at larger spatial scales and in uncharacterized sites, additional approaches are needed.

Fortunately, a variety of additional approaches are available to couple stress with response. For example, evaluation of the effects of toxic substances on ecosystems can involve multiple approaches: 1) comparison of a toxic response or specific dose level of a contaminant to an action level that has been linked to biological effects, 2) assessment of environmental effects of multiple toxic substances associated with well characterized contaminant exposures, or 3) the coupling of physiologic and genetic indicators with environmental chemistry and ecological responses at multiple spatial and temporal scales. The first approach is limited when multiple contaminants and multiple stressor types are present. The second approach is primarily limited because often no direct integration of the toxic response and exposure occurs. The result is data correlation, which is limited to the spatial and temporal scope of the immediate investigation. The third approach considers multiple stressors and permits direct integration and scaling, but significant challenges remain to fully develop indicators for a range of habitats and model organisms. A simple index of sediment contamination combines the first and second approaches. The Sediment Quality Triad, which provides a framework for analyzing benthic community data, analytical chemistry, and toxicity test data to assess whether a site is impacted by toxicants, is widely used throughout the nation (Long and Chapman 1985; MacDonald and Ingersoll 2002). Yet, the action levels derived for specific contaminants are often unknown and the interpretation of benthic data are highly variable and lack specific reference conditions.

Clearly, there is a need for controlled experiments in laboratory settings for those stressors (e.g. toxicants, nutrients, hypoxia, and turbidity) amenable to manipulation. For larger scale stressors (e.g., exotic species introductions or habitat change and fragmentation) not amenable to manipulation, better experimental designs that test responses over gradients of stressor levels are among the options for linking stress with response. Coupling of these two methodological approaches will be essential for future advances in developments for measuring appropriate indicators of response.

**Multiple Stressors of Environmental Condition**

Stress on coastal ecosystems is usually a combined effect of natural and anthropogenic disturbances. Natural disturbances in the US coastal zones primarily include water level fluctuations resulting from droughts and floods, wind such as hurricanes, natural soil/sediment deposition, insect infestations, and forest fires. Each of these natural disturbances has varying intensity of effects both spatially and temporally. Major anthropogenic disturbances to coastal ecosystems include permanent land cover conversions of native vegetation to agricultural, residential, and industrial areas, and temporary conversions of land due to forestry. Each of these conversions result in concomitant anthropogenic disturbance to coastal ecosystems such as 1) landscape effects of fragmentation, 2) increased surface water runoff, 3) increased nutrient and sediment input, 4) increased pesticide and other chemical inputs, 5) increased water temperature, and 6) greater human disturbance from recreational use, increased fish and shellfish extraction, and noise. Climate change and resultant change in weather patterns itself is a combination of both natural, stochastic events as well as human induced warming that affects vegetation, water levels, and virtually all types of disturbance. Deciphering the simultaneous effects of these natural and anthropogenic disturbances in coastal ecosystems is a challenging and complex task.

It is imperative to put in context the relative effects of anthropogenic disturbance with the ranges of variation in natural disturbance regimes. Because of the large size of coastal ecosystems, manipulative experiments to untangle the complexities of the varying disturbance regimes are difficult except on a relatively small-scale. Combining specific indicators with modeling efforts clarifies and distinguishes anthropogenic from natural stress in individual ecosystems and regions (DeAngelis 1998; Gentile 2001). In general, there has been a trend from using direct diagnostic measures of stressors to using integrated indicators of ecosystem structure and function (NRC 2001).

Characterizing the effects of multiple stressors on any ecosystem is among the most challenging tasks facing scientists today. Multiple stressors can result in synergistic, additive, or antagonistic types of effects on biological responses. Disentangling the various effects of multiple stressors will likely emanate from a combination of controlled laboratory experiments, large-scale studies over multidimensional gradients of stress, and insightful modeling of ecosystem responses and change.

**Spatial and Temporal Explicitness**

Ecological indicators are constructed or selected to assess the condition of ecosystems and to detect environmental change related to human disturbance. Condition is often assessed by documenting the state or rate of ecological processes such as productivity, respiration, or the structuring of biological communities. Indicators may do this by either measuring those processes directly (such as primary productivity) or inferring process from pattern (such as utilizing Indices of Biotic Integrity (IBI) as descriptors of community structure). Ecological processes operate over a range of spatial and temporal scales and the resulting patterns are expressed over varied scales. Hence, the relevant scale of each indicator must be specified if the appropriate conceptual model relating pattern to process is to be deduced. Levin (1992) stated "the concepts of scale and pattern are ineluctably intertwined. The description of pattern is the description of variation, and the quantification of variation requires the determination of scales."

Many studies have sought to quantify spatio-temporal patterns across a range of scales. Unfortunately, few have determined whether patterns are consistent across scales or related phenomena across scales (Caldow and Racey 2000). In addition, due to technical and logistical reasons, most ecological studies have focused on small systems such as the site or plot level and short periods of time. These limitations have led to a reluctance to develop indicators over large spatial scales (Edwards et al. 1994; Innes 1998). Alternatives exist, however, such as top-down approaches where large-scale processes form the basis of inferring process from pattern and are being applied in regional classification schemes (Hawkins et al. 2000).

The need for scale explicitness is complicated by multiple stressors arising from human and natural disturbances. Most ecological indicators are related to multiple stress and scales. For example, in a study of littoral macroinvertebrate communities (Johnson and Goedkoop 2002), 23% of the variance in taxonomic composition was associated with habitat factors, but greater spatial scales (riparian, catchment, and ecoregion classification) accounted for 24% of the variance. If indicators are to be utilized effectively in management, it is necessary that we know the relevant scale(s), so that the scale of management actions matches the scale of the phenomena being measured (Hobbs 1998). Experimental approaches that allow for the partitioning of the variance among different stress components and over a hierarchy of spatial scales will be necessary in future endeavors at improving our understanding of ecological indicators.

**Reference Conditions**

To interpret any set of indicators, it is necessary to compare results of monitoring to a standard or benchmark. One of the preferred benchmarks is a reference site or condition. The use of reference sites has become increasingly common as ecologists and managers search for reasonable and scientifically-based methods to measure and describe the inherent variability in natural aquatic systems (Hughes et al. 1986; Kentula et al. 1992; Rheinhardt et al. 1999). Since there are likely few places on earth that are unaffected by anthropogenic disturbances, true reference areas remain elusive. For example, even extreme coastal regions of Greenland or Antarctica have been affected by atmospheric chemical inputs and climate change. Alternatively, in coastal regions with long histories of human occupation and, hence, anthropogenic disturbances, reference sites can be specific estuaries, watersheds, or lotic systems entering the coastal zone. These can be represented by the best attainable environmental conditions for a specific geographic setting, a historic representation using paleolimnological data, a simulated reference condition, or a situation where conditions fall within the range of natural variability for the system (NRC 2002).

Determining reference condition of the system is highly dependent on the indicators used and where the samples were gathered. Benthic indicators will provide different results than those from planktonic habitats. Similarly, indicators will be different in large, ephemerally-stratified systems (e.g., Chesapeake Bay, MD-VA; Pamlico Sound, NC; Mobile Bay, AL; San Francisco Bay, CA; or Green Bay, WI) compared with smaller, well-flushed systems. For instance, phytoplankton growth responses to nutrient enrichment will not be as profound as those for benthic microalgae in well flushed systems. Here, benthic microalgae may be more sensitive and meaningful indicators of ecosystem response to nutrient enrichment. Indicators of community structure (i.e., diversity indices, keystone species) may gauge ecosystem conditions quite distinct from indicators of function (e.g., primary and secondary production, respiration, and nutrient cycling). IBI, habitat suitability indices (HSI), and chemical monitoring are specific examples of indicators that in combination can assess structure, physical-chemical quality, and biological measures.

**Examples of New Indicators**

The development and use of environmental indicators have included an enormous number of possible endpoints and most reflect the breadth and diversity in the sciences of biology, chemistry, and physics. For instance, biological indicators span the realm of biological organization from genetic markers to entire ecosystems. Chemical indicators reflect a variety of spatial or temporal scales ranging from oxygen demand for a specific point source to global carbon dioxide distributions in the atmosphere (US EPA 2002; McKenzie et al. 1992; NRC 2000; Noss 1990; O'Neill et al. 1988). Physical indicators include elevational, morphological, transport, circulation, exchange and stratification processes that have huge ramifications for ecosystem structure and function. Because of the massive amounts of information that can be gathered at any one level of physical, chemical, and biological organization and for many spatial or temporal scales (Dixit et al. 1992; Karr 1985), the ability to integrate data among levels of organization in space and time are daunting.

The lack in ability to integrate is exacerbated by funding scenarios that are limited in amount and duration. For instance, most funding sources (federal, state and private) are tens or hundreds of thousands of dollars and studies greater than three years are rare. These monetary and time deficiencies have been recognized by funding sources such as NSF (e.g., Long Term Ecological Research, Biocomplexity, Global Ocean Flux, Biotechnology and other centers) and are increasingly being recognized by US EPA with the development of programs such as ~~EaGLe~~. As such programs mature, increasing advancements will be made to integrate knowledge from a variety of trophic levels as well as spatial and temporal scales. Here we provide two brief examples of potentially new types of indicators; one that links productivity and hydrology and another linking community and landscape patterns. These new types of indicators will be essential to better measure and understand the complexity, response and condition of coastal systems.

**Photopigments indicators of interactive effects of nutrients and hydrology on estuarine and coastal primary producers.**

Nitrogen availability most frequently controls microalgal and higher plant primary production in estuarine and coastal waters (Ryther and Dunstan 1971; Nixon 1995). Loading rates of this nutrient directly reflects human population density and activity in coastal water- and airsheds (Peierls et al. 1991). Excessive N loading is a key causative agent for accelerating primary production or eutrophication (Nixon 1995; Paerl 1997). Symptoms include phytoplankton blooms, which may accumulate as ungrazed organic matter in the sediments, providing the ?fuel? for oxygen consumption and depletion in bottom waters and sediments. This chain of events is particularly problematic in salinity or temperature-stratified waters, where oxygen may not be easily replenished from the atmosphere. Hypoxic conditions alter nutrient cycling and promote fish disease and mortality (Diaz and Rosenberg 1991; Paerl et al. 1998).

Suspended microalgae, or phytoplankton, account for the bulk of estuarine and coastal primary production. Their composition and activity play central roles in determining fertility, eutrophication, and water quality. Water discharge controls transport of phytoplankton through these systems, and plays an interactive role with nutrient supply to control phytoplankton growth, competition, succession, and community composition. For example, high rates of freshwater discharge reduce the salinity and residence time. These conditions favor fast-growing oligohaline phytoplankton, such as chlorophytes (green algae). In contrast, low discharge conditions promote long water residence, high salinity conditions, which favor slower growing, halophyllic taxa, such as dinoflagellates and certain cyanobacteria. Phytoplankton community composition impacts the structure and function of estuarine food webs, nutrient cycling, habitat condition, fisheries resources and overall ecosystem condition (Paerl et al. 2002) (Fig. 2 **note:** not included in this draft).

Chlorophyll *a* has been used for many years as a sensitive indicator of phytoplankton biomass. However, since virtually all phytoplankton contain this pigment, it alone cannot be used to determine community composition. Using additional diagnostic chlorophyll and carotenoid photopigments as indicators of major phytoplankton functional groups (i.e., diatoms, dinoflagellates, chlorophytes, cyanobacteria, cryptomonads), we can examine the interactive effects of nutrient and hydrologically-driven changes of phytoplankton community composition and activity in the Neuse River Estuary (NRE) and Pamlico Sound (PS), NC. High performance liquid chromatography (HPLC), coupled to photodiode array spectrophotometry (PDAS) is used to determine phytoplankton group composition based on the diagnostic photopigments. Photopigment markers include Chl *b* and lutein (chlorophytes), zeaxanthin, myxoxanthophyll, and echinenone (cyanobacteria), fucoxanthin (diatoms), peridinin (dinoflagellates), and alloxanthin (cryptomonads). A statistical procedure, ChemTax (Mackey et al. 1996) partitions chlorophyll *a* (i.e., total microalgal biomass) into the major algal groups, to determine the relative and absolute contributions of each group.

Using data from ongoing studies in the NRE and PS (1994-present), it can be seen that these systems have experienced the combined stresses of anthropogenic nutrient enrichment, droughts (reduced flushing combined with minimal nutrient inputs), and since 1996, elevated hurricane activity (high flushing accompanied by elevated nutrient inputs). Seasonal and hurricane induced variations in river discharge, and the resulting changes in flushing rates and hence, estuarine residence times, have differentially affected phytoplankton taxonomic groups as a function of their contrasting growth characteristics. For example, the relative contribution of chlorophytes, cryptophytes, and diatoms to the total chl *a* pool appeared strongly controlled by periods of elevated river flow in the NRE. These effects are due to the efficient growth rates and enhanced nutrient uptake rates of these groups. Cyanobacteria, on the other hand, showed greater relative biomass when flushing was minimal (i.e., longer residence times) during the summer.

Further evidence that hydrologic changes have altered phytoplankton community structure is provided by the observed historical trends in dinoflagellate and chlorophyte abundance in the NRE. Both decreases in the occurrence of winter-spring dinoflagellate blooms and increases in the abundance of chlorophytes coincided with the increased frequency and magnitude of tropical storms and hurricanes since 1996. The relatively slow growth rates of dinoflagellates may have led to their reduced abundance during these high river discharge events. These phytoplankton community changes have been linked to altered trophodynamics and nutrient cycling with subsequent impacts on fisheries habitat and yields.

Diagnostic photopigment analyses can be adapted to routine monitoring programs (Pinckney et al. 2001). In addition, these analyses can be used to calibrate remotely-sensed phytoplankton distributions on habitat, ecosystem and regional scale.

**Community and Landscale-level Indicators**

A particularly daunting task in the development of indicators has been to scale and aggregate species population responses at the site level to reflect conditions of the biological community for specific taxa or provide assessments of large scale patterns such as IBI (Karr 1981), biological species profiles (Simon et al. 2001), multi-taxa indices (O'Connor et al. 2000), or indices of environmental integrity (Paul 2003). These approaches hold tremendous potential as indicators for assessments of environmental condition and over large landscape or regional areas as well as for detection of change over time. However, there are considerable development needs among these types of indicators such as 1) providing linkages with specific and multiple stressors, 2) exploration of analytical techniques to integrate and synthesize the complex biological signals, 3) communicating the results of these multivariate responses among stressors and over varying spatial scales, and 4) providing explicit spatial or temporal scales in which the indicators can be aggregated to scales of management actions (Wardrop et al. 2003).

Plant and animal community structure and function have been extensively measured to describe the condition of both aquatic and terrestrial systems. The strength of these more integrative approaches is the greater likelihood of identifying biological responses that are sensitive to the wide variety of stressors that exist within any environment. For example, monitoring of fish communities in the Great Lakes over the past 30 years has tracked the spread and effects of the exotic sea lamprey on the native fishery (Ashworth 1986). In an experiment on the pesticide effects of mosquito control agents in wetlands on zooplankton, aquatic insects, and birds, aquatic insects were the only organisms that exhibited a response to treatment (Hershey et al. 1998; Niemi et al. 1999). The response by birds to chemical treatment was likely masked by high predation rates on nesting birds and, hence, birds were a better integrative measure of the regional predatory population. Similarly, O'Connor et al (2000) found a correlation among many taxa (diatoms, benthos, zooplankton, fish and birds) to the gross condition of lakes, but fish provided the best metric of near-shore conditions.

The overall strength of the species, population, and community approach is the ability to sample a wide variety of taxa; each of which has a unique life history capable of being disrupted by stress. All coastal regions are represented by thousands of species including taxa such as bacteria, plankton, macroinvertebrates, fish, vascular and non-vascular plants, amphibians, and birds. Many associations between these taxa and stress exist. For example, diatoms are particularly sensitive to the nutrient status of water quality (Dixit et al. 1992). Benthic invertebrate communities are responsive to sediment contamination (Bailey et al. 1995). Fish communities integrate information on human development (Brazner 1997) and exotic species (Rahel 2000). Wetland vegetation is directly affected by hydrological modifications such as dikes and road building (Herdendorf 1992). Amphibians are sensitive to water quality in wetlands (Kutka and Bachman 1990). Birds are affected by habitat change and fragmentation at a landscape scale (Robinson et al. 1995). Moreover, many of these taxa have well-refined sampling methods available and some have nationwide monitoring programs that are currently in use and have been in effect for over 30 years (Robbins et al. 1989; NSF LTER 19xx).

Probability-based and standardized sampling of specific sites over large landscapes allows for regional-scale assessments of environmental conditions. Examples of these types of activities are illustrated by the identification of imperiled systems (Stein et al. 2000), the extensive development of biological indicators for the Mid-Atlantic Highlands (Herlihy et al. 1998), and US EPA's EMAP (US EPA 2002). Among the important aspects of these large-scale approaches is the development of indicators that can identify areas that have the most severe problems and greatest need of management attention, action, and potentially restoration. Integration of these types of data will be challenging and will require multivariate, integrative approaches of multi-taxa biological communities over large-scale landscapes and regions.

Promising new techniques to achieve integration of measurements at multiple spatial, temporal, and biological scales include development of multimetric indices (e.g. Karr 1981; Paul 2003), integration techniques such as statistical (Jongman et al. 1995), probability network models (Borsuk et al. 2003), path analysis, and neural nets, and mechanisms for communicating results to scientists, managers, and the lay public (tiered aquatic life reference). Moreover, these techniques will require coupling with population and ecosystem-based models for aid in the interpretation of stressor risk and alternative management actions (DeAngelis 1998; Gentile 2001). The overall integration of multiple measurements of ecological responses to multiple stressors over large spatial and temporal scales is only beginning to be developed. Several large-scale programs have been initiated (Fore 2003), and EaGLe reported here. With the exponential increase in computer capabilities, remote sensing technology, modeling and statistical sophistication, and Internet communication speeds, integration of indicator signals and assessments of environmental condition and change will also advance exponentially.

## Conclusions

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## Figures

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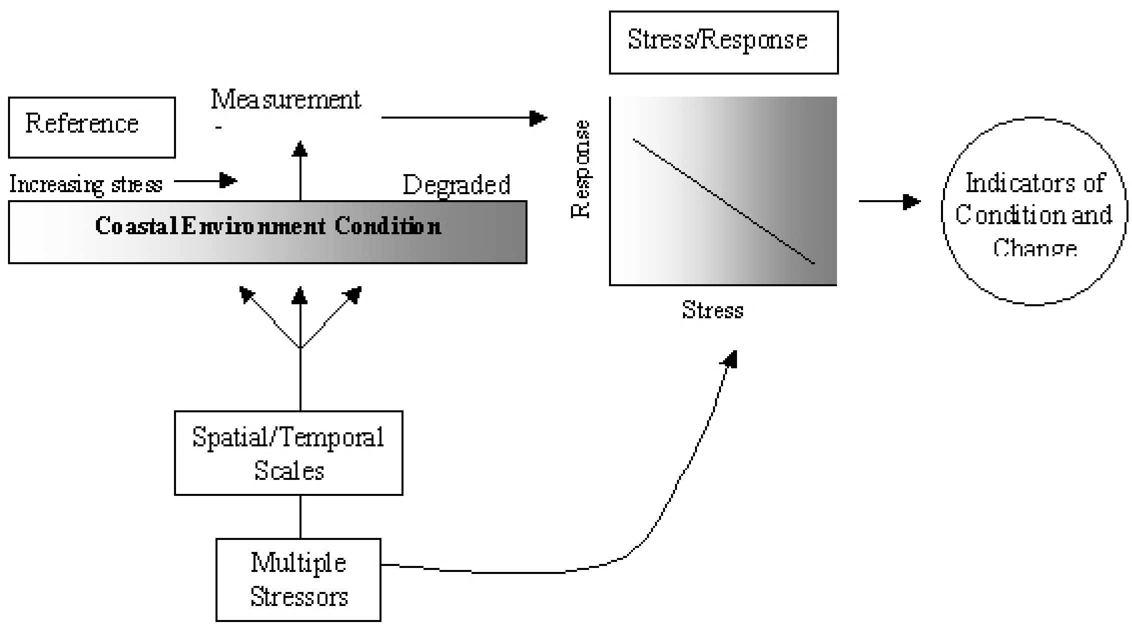


Figure 1. Conceptual diagram of critical elements in indicator development.