5. Recommended Indicators of Estuarine Water Quality in Georgia

The discussion below is focused on measurements of the conditions, response variables, and ancillary data collected to evaluate water quality in estuaries. Most efforts to evaluate water quality concentrate on eutrophication and hypoxia as the main water quality concerns, and the parameters chosen as indicators are intended to describe the magnitude of, or susceptibility to, such problems. We have not reviewed other aspects of the environment that might be included in an assessment of an estuary, such as sediment quality, biotic assemblages, pathogens, priority pollutants, or metals, as we did not have the data to assess these characteristics in Georgia estuaries. Moreover, monitoring of some of these parameters (e.g. fish tissue contamination) falls under the jurisdiction of other State agencies or divisions and so is not part of the GA DNR CRD dataset. Our discussion also does not focus on recommendations for measurements made to evaluate human uses of water bodies (i.e. for recreation, fishing, and shellfishing). Water quality must always be assessed with regard to the designated use of the water body, and these indicators, which are generally mandated by State and federal agencies to protect human health, should augment or supersede the recommendations suggested here where appropriate. However, we do include a discussion of some of the issues associated with potential indicators of human health under "Microbial Indicators". In this section we first describe several ongoing programs that use indicators to assess coastal water quality in the U.S., at both the national and regional scales, and their applicability to Georgia. These studies all use integrated indices to distill the evaluation of many separate indicators of water quality into a single score intended to represent the overall condition of water quality or eutrophication, and we describe the pros and cons of this approach. We then provide our recommendations for a suite of seven water quality indicators to be used in Georgia, along with the criteria that we suggest for rating observations of each of these indicators as "good", "fair", or "poor."

National and Regional Water Quality Indicators

National Estuarine Eutrophication Assessments

The National Oceanic and Atmospheric Administration (NOAA) has taken the lead on two National Estuarine Eutrophication Assessments (NEEA) (Bricker et al. 1999, 2007) designed to evaluate the presence of symptoms of eutrophication in U.S. estuaries as well as their potential future susceptibility. Their approach was to evaluate what they termed primary (high concentrations of chlorophyll *a*, problematic epiphytic growth, and problematic macroalgal growth) and secondary (low dissolved oxygen, loss of submerged aquatic vegetation (SAV, e.g. seagrasses), and nuisance/toxic algal blooms) symptoms of eutrophication. In this case, "primary" and "secondary" refer to symptoms that develop first and second, not to more- and less-important symptoms. Levels of expression of symptoms included information on frequency of occurrence, spatial extent, and duration of problems. They evaluated future susceptibility using projected nitrogen loads and the potential for nutrient retention in the estuary based on flushing and dilution characteristics.

Although the information used to assess eutrophic condition changed slightly between the two reports, both assessments reflect the classic idea of hypoxia formation in which nutrients stimulate phytoplankton blooms that subsequently degrade and cause low dissolved oxygen. They also focused heavily on whether or not waters are supporting healthy stands of SAVs, or whether they had experienced SAV *loss* and/or overgrowth by epiphytes. Although this is a major concern in estuaries that have historically supported SAVs and have lost important nursery habitat for fish and crustaceans, the emphasis on loss of a habitat that generally does not exist in Georgia made it difficult to fit information about Georgia estuaries into the established framework and make meaningful comparisons. In the second assessment (Bricker et al. 2007), allowance was made for omitting inapplicable indicators and recalculating the indices without them where appropriate.

Both assessments found that approximately two-thirds of the U.S. estuaries for which data were available exhibited moderate to high expressions of eutrophic conditions, with most estuaries being highly influenced by human-related activities. Georgia estuaries included in these studies (Savannah River, Ossabaw Sound, St. Catherines/Sapelo Sounds, Altamaha River, St. Andrew/St. Simons Sounds, and St. Marys River/Cumberland Sound) rated low to moderate in eutrophic conditions in both studies but much of the data were missing, partially because of a mismatch between the indicators used (e.g. SAVs) and relevant conditions in Georgia estuaries.

National Coastal Condition Reports

The National Coastal Condition Report is an effort led by the U.S. Environmental Protection Agency (EPA) that has completed three assessments to-date (U.S. EPA 2001a, 2004, 2008, hereafter referred to as NCCR I, NCCR II, and NCCR III). These reports draw on data from a wide variety of programs, but emphasis is placed on a probability-based spatial sampling design to ensure that percentages of estuarine areas in different quality categories can be assessed with known confidence. Site locations may change from one assessment to the next. Samples are typically collected once annually during late summer, the period during which water quality may be limiting the biota and use of sensitive nursery habitat may be high. This approach means that characterization of temporal variation is limited and trends over time are difficult to assess using these data.

Water quality in these reports is evaluated according to a water quality index, which is based on five indicator parameters: dissolved oxygen, chlorophyll *a*, dissolved inorganic nitrogen, dissolved inorganic phosphorus, and water clarity. Classifying water clarity using standardized criteria for the entire country proved to be problematic in NCCR I, as there are natural geographic differences in clarity and in the presence of SAVs (which require high light). As a result, the later reports used separate, more lenient water clarity criteria for estuaries with naturally turbid waters, such as those in Georgia and South Carolina (neither of which have historically supported extensive beds of submerged aquatic vegetation).

The National Coastal Condition Reports have consistently rated water quality in continental U.S. estuaries as “fair” overall, while the southeast as a whole has been rated on the high side of “fair”. Assessments for individual states are not generally provided in the national reports; however, a report by GA DNR CRD (Guadagnoli et al. 2005) using data collected in 2000-2001 for the U.S. EPA National Coastal Assessment program (data not included in our analyses) classified 80% of Georgia estuarine waters as having “fair” water quality, 18% as “poor”, and only 1% as “good” based on the NCCR indicators.

South Carolina Estuarine and Coastal Assessment Program (SCECAP)

The SCECAP program publishes interagency assessments of the South Carolina coastal zone (Van Dolah et al. 2002, 2004, 2006; Bergquist et al. 2009). The sampling design is similar to that used by the U.S. EPA National Coastal Assessment program, with a probability-based, random tessellation, stratified sampling design and sample collection once at each station during mid-June through August, generally within 3 hours of low tide. Measured water quality parameters included dissolved oxygen, salinity, temperature, pH, total and dissolved components of nitrogen (total nitrogen, ammonia, nitrate+nitrite, total Kjeldahl nitrogen (TKN), total dissolved nitrogen (TDN)) and phosphorus (total phosphorus, orthophosphate, total dissolved phosphorus (TDP)), dissolved silica, total alkalinity, total organic carbon (TOC), total suspended solids, turbidity, five-day biochemical oxygen demand (BOD_5), chlorophyll *a*, and fecal coliform.

Georgia estuaries appear to be similar to South Carolina estuaries in many respects, according to the national reports (Bricker et al. 1999, 2007; U.S. EPA 2001a, 2004, 2008); therefore, comparison of Georgia data with those collected in South Carolina is appropriate to establish regional conditions. In the first report, Van Dolah et al. (2002) developed a single water quality index based on dissolved oxygen, BOD_5, fecal coliform, total nitrogen, total phosphorus, and pH. Subsequent reports dropped BOD_5 for lack of guidelines and added chlorophyll *a*. Starting in 2005, there were also changes in the method of computing the indices aimed at improving the ability to detect degraded environmental conditions (Bergquist et al. 2009). Tidal creeks and open water are assessed separately in the SCECAP program to see if there are significant differences between the two types of water bodies. The percentages of each water body type falling into different water quality categories changed little from 1999-2006, and the changes appear to be related more to the changes in indices than to any changes in actual water quality (Table 5-1).

Table 5-1. Percentage of each water body type falling into each water quality category in South Carolina during SCECAP reporting periods (compiled from Van Dolah et al. 2002, 2004, 2006; Bergquist et al. 2009).

	Good	Fair	Poor
Tidal Creeks			
1999-2000	62	33	5
2001-2002	73	22	5
2003-2004	75	22	3
2005-2006	74	20	6
Open Water			
1999-2000	89	11	0
2001-2002	88	12	0
2003-2004	87	13	0
2005-2006	90	8	2

Integrated Indices

All of the above-mentioned studies use integrated indices that combine many separate indicators into a single score intended to represent the overall conditions. Reasons given for using them are that they are more reliable assessment tools (Van Dolah et al. 2002); allow comparisons, ranking, and priority-setting, and facilitate national and regional summaries (Bricker et al. 1999); and provide natural resource managers and the general public with simplified statements about coastal condition (Bergquist et al. 2009). The general procedure for developing integrated indices is to rank individual indicator values (dissolved oxygen, etc.) into ordinal categories such as “good”, “fair”, and “poor” based on specified criteria such as numeric ranges. The categories are often assigned values (e.g. “poor”=1, “fair”=3, “good”=5). Although they denote rank order, the actual numbers (and the spacing between them) are somewhat arbitrary. The individual indicators are then combined into a single score, either by averaging the ordinal category values (e.g. NCCR reports) or by referring to a matrix of all combinations of levels of the indicator categories (e.g. NEEA). In some cases the water quality index is further combined with other integrated indices (sediment quality, biota, etc.) to produce an overall index of water body condition.

There are several drawbacks to the use of these types of integrated indices. Perhaps the most problematic is the fact that integrated indices involve, by necessity, decisions as to the relative weights of the various indicators that are being combined. It is often difficult to decide if some aspects of water quality are more important than others and to assign numerical weights to those decisions, and the default is generally to weight each aspect equally. The assignment of equal weight to each indicator introduces an arbitrary idea of equality that may not reflect the relative importance of the actual processes at work. Bricker et al. (1999) give equal weight to their three primary symptoms of eutrophication but choose the highest score from their three secondary symptoms, and then combine primary and secondary indices into an overall expression of eutrophic condition. While these assigned weights may qualitatively reflect best professional judgment, sometimes they just reflect a lack of more specific knowledge of the relative importance of symptoms.

One example of the difficulties in deciding how best to weight individual factors can be seen in the evolution of the index used by SCECAP. Their original approach was to weight each of six factors equally in the calculation of an integrated index of water quality. Using this approach, three of the measures (TN, TP, and chlorophyll *a*) together accounted for 50% of the water quality score. In 2005, the same three measures were first used to calculate a eutrophication index (equally weighted), which was then combined with the other three measures (dissolved oxygen, fecal coliform, and pH) to produce a water quality index. Thus, the eutrophication parameters collectively accounted for only 25% of the final index. Also in 2005, SCECAP changed their category scoring system to give “poor” ratings more weight by assigning a numerical value of 0 rather than 1; thus, a “poor” score reduced the overall condition score more than it had in earlier reports. The change was made in order to improve their “ability to detect degraded environmental conditions", which implies that the previous scoring system did not concur with independent opinions about which waters are degraded. While these attempts to improve the water quality index are certainly commendable, it appears that integrated indices are being designed to follow, rather than inform, professional judgment.

Another problem with the use of integrated indices is that complications can occur when values are missing or indicators are not applicable to a given water body. Where inapplicable indicators or missing data are omitted, this necessarily changes the weighting of the remaining indicators relative to their weights in estuaries where all indicators are applicable. This problem arose in the calculation of the NEEA eutrophication index (Bricker et al. 2007): the assessment of SAVs was not relevant for Georgia and South Carolina so the NEEA index for these estuaries is based on fewer indicators (giving additional weight to the remaining ones) than that used in the evaluation of other estuaries around the country.

Finally, integrated indices do not communicate all the information about their components and can lead to more questions than answers. If an estuary's water quality is rated "good" then all is probably well, but a "poor" rating will surely lead to questions about the cause, which will lead coastal resource managers and others to examine the individual indicators.

Although it is easy to see why an integrated index is attractive as a way to facilitate comparisons and make easily understandable statements about coastal conditions, we elected not to use integrated indices of water quality in this report for the reasons described above. Instead, we provide information on a suite of individual indicators, each of which has relevance to coastal water quality, along with criteria for assessing whether each observation is "good", "fair" or "poor." If a particular location is classified as poor for more than one indicator, that information can be communicated without the difficulties of assigning weights and numeric values to categorical information. It also makes it possible to simply omit a score in situations where a particular indicator is missing or not applicable.

Recommended Indicators for Georgia Estuaries

We took several factors into account when selecting indicators of water quality for Georgia estuaries. First, we considered the parameters that were measured in the national and regional studies described above. The similarity in the suites of indicators used in these assessments is not coincidental: the long lists of authors, contributors, and reference materials included in those reports underscore the fact that a general consensus is emerging regarding estuarine water quality and how to assess it. Another relevant report came from the Nutrients Workgroup of the National Water Quality Monitoring Council, which issued advice regarding the nutrient parameters that should be monitored as part of the National Water Quality Monitoring Network for U.S. Coastal Waters and their Tributaries (Caffrey et al. 2007). Their recommendations are divided into nutrients, response variables, and ancillary data. For estuaries and nearshore coastal waters, they divide the nutrient parameters into two groups: Tier 1, required parameters, includes total nitrogen, total phosphorus, and the dissolved fractions of ammonium, nitrate+nitrite, orthophosphate, and silica; Tier 2, parameters that would add significant value but may not be essential to all programs, includes total dissolved nitrogen and phosphorus and particulate nitrogen and phosphorus. Response variables include chlorophyll *a*, dissolved oxygen, and salinity or conductivity. Ancillary data include dissolved organic and inorganic carbon, particulate carbon, pH, total suspended sediments, and photosynthetically active radiation.

We also considered the federal requirements currently being developed by the EPA as it moves towards the adoption of numeric nutrient criteria (Grumbles 2007). Although they are still under development, the guidelines set forth in the Nutrient Criteria Technical Guidance Manual for Estuarine and Coastal Marine Waters (U.S. EPA 2001b) suggest, at minimum, the measurement of total nitrogen, total phosphorus, chlorophyll *a*, and transparency. They further suggest that dissolved oxygen be measured as an additional primary response variable in systems that have already experienced hypoxia. Proposed nutrient criteria for Georgia's coastal and marine waters are expected in the second half of 2012, with expected adoption by

2014 (Rose, pers. comm.). While there may be some latitude in the choice of indicators, the use of indicators preferred by the U.S. EPA will probably facilitate regional and national comparisons.

We recommend seven basic indicators of water quality for Georgia estuaries: pH, dissolved oxygen, nitrogen, phosphorus, chlorophyll *a*, transparency, and biochemical oxygen demand (BOD). The first two of these, pH and dissolved oxygen, may be considered "immediate" indicators of poor water quality in that they may indicate that a stressful and potentially lethal condition is already in progress. Furthermore, a single episode of hypoxia/anoxia or pH outside the normal range may do lasting damage to the biotic community. It is also important to measure several "early warning" indicators of potentially poor water quality in order to anticipate problems and make appropriate management decisions. Measuring several indicators that cover the progression of eutrophication, from nutrient over-enrichment to algal overgrowth (if present) to enhanced microbial respiration and hypoxia, will help to ensure that problems will not be missed entirely due to limited sampling frequencies. With this progression in mind, we recommend measuring nitrogen, phosphorus, chlorophyll *a*, transparency, and biochemical oxygen demand (BOD). While most of these tend to lead to problems only if they are chronically outside the desirable range, there could be circumstances where an individual extreme episode could lead to lasting damage, e.g. ammonia high enough to be toxic to fish, or a phytoplankton bloom dense enough to lead quickly to hypoxia.

The information provided by these indicators will help to both classify and understand the causes of water quality degradation in Georgia. Below we discuss the rationale for choosing each of these indicators; the development of criteria for classifying observations as "good", "fair", and "poor"; and our recommendations for evaluation. Where appropriate, we also provide suggestions regarding methodology.

pH

Rationale

pH is a measure of the hydrogen ion concentration of a water sample. It is expressed on a $\log_{10}$ scale and ranges between 1 (for acidic samples) and 14 (for basic samples). The buffering capacity of seawater is often thought to protect estuaries and coastal waters against pH changes large enough to affect organisms, so pH is not always used as an indicator in coastal waters. However, there is mounting evidence that estuaries do experience pH changes that may be stressful to their inhabitants (Knutzen 1981; Ringwood and Keppler 2002). Ocean acidification due to rising atmospheric CO_2 levels is a growing concern. Much of the research on this topic has focused on the ability of calcareous organisms such as corals and shellfish to produce shells, but decreased pH and increased partial pressure of carbon dioxide (pCO_2) can result in other physiological responses in many other taxa. A recent exchange in the literature regarding the resistance of marine biota to predicted levels of ocean acidification (pH declines of 0.3 – 0.5 units by 2100) underscores the ideas that some, but not all, taxonomic groups and life stages are sensitive to that magnitude of pH change and that the threshold of deleterious effects for short-term pH changes is in the range of 0.5 units (Hendriks et al. 2010; Dupont et al. 2010; Hendriks and Duarte, 2010). Decreases of 0.5 units or less appear to be tolerated well by many organisms, although some stress responses (e.g. increased ventilation in sharks) start within this range. A decrease of 1 or more pH units can result in more serious deleterious effects (e.g. metabolic stress due to extended internal pH compensation; the inability of larvae of some species to compensate; and reductions in growth, reproductive potential, and survival (especially when combined with hypoxia)) (Knutzen 1981; Fabry et al. 2008). Both these reviews of the literature noted the urgent need for further studies across a wide range of taxa. The effects of increased pH on marine organisms are even less well studied, because ocean acidification has been the primary concern.

Criteria Development

pH values outside the normal range for a location can be stressful to organisms, but what constitutes "normal" can vary tremendously from one location to another, especially in estuarine environments where pH varies with salinity. If simple constant values of pH are used as criteria then they must necessarily be broad enough to encompass normal conditions across a wide range of salinities, rendering them nearly meaningless as real indicators of stressful conditions at any single location. The U.S. EPA (2009) recommends separate standards for chronic exposure for freshwater (6.5-9.0) and saltwater (6.5-8.5), but these are still broad ranges.

South Carolina's standard (SC DHEC 2008) states that the pH in Class SA and SB tidal saltwaters (designated for recreation, crabbing, and fishing) should always be in the range 6.5-8.5 and should not vary more than 0.5 units above or below that of effluent-free waters in the same geologic area having a similar salinity, alkalinity and temperature. Shellfish harvesting waters are more strictly controlled, allowing for a variation of 0.3 units. In the initial SCECAP reports (Van Dolah et al. 2002, 2004, 2006), criteria were established for polyhaline waters (salinity≥18) using data from sites considered to be pristine. The most recent study (Bergquist et al. 2009) included enough low-salinity data to establish a linear relationship between pH and salinity and then used percentiles below that line to define the "good/fair" (25^{th} percentile) and "fair/poor" (10^{th} percentile) boundaries for a given salinity. These boundaries correspond to 0.22 and 0.35 units below the mean line, well within the South Carolina state standard of 0.5 units.

The GA DNR (2009) water quality control rules for the designated use of "fishing and propagation of fish, shellfish, game, and other aquatic life" (hereafter "fishing") state that the standard is within the range of 6.0-8.5, but these standards do not necessarily apply to Georgia estuaries with naturally low pH. As described in the Correlations section, we separated the GA DNR CRD observations into three different estuary types: blackwater systems (Satilla River, St. Andrew Sound, St. Marys River, and Cumberland Sound), alkaline blackwater (Ogeechee River/Ossabaw Sound) and alluvial and tidewater (all other sites) (Figure 4-4). In contrast to the situation in South Carolina, where a linear relationship could be applied to pH and salinity observations, the relationships in Georgia estuaries were non-linear, at least for the blackwater and alkaline blackwater systems. We used the equations that we developed to relate pH and salinity for each estuary type (Figure 4-5) as the basis for our recommended pH criteria, as described below. We were unable to conclusively define reference streams for each type because a variety of permitted wastewater and industrial discharges exist in the watersheds of many systems, including the Satilla and Ogeechee Rivers, which are the source of most or all of the data for the blackwater and alkaline-blackwater pH types, respectively. However, approximately 20% of the observations are from shellfishing locations, which by their nature are relatively pristine, and we have no reason to think that the pH relationships defined here are heavily influenced by effluents.

Recommendations

Based on the pH/salinity relationships described in the Correlations section, we defined "normal" pH conditions for each estuary type as the values represented by the regression lines in Figure 4-5. While the calculated "normal" pH for freshwater (salinity = 0) varies widely across the estuarine types (pH = 5.2 for blackwater systems, 6.7 for alkaline blackwater systems, and 7.2 for alluvial and tidewater systems), the "normal" pH values for seawater (salinity = 35) in all types converge to 7.7-7.9 (based on equations in Table 5-2). Given the literature reviews of effects of decreasing pH on marine organisms, summarized above, we recommend that the "good/fair" boundary for pH be defined as a deviation of 0.5 units from these lines, and the "fair/poor" boundary be defined as a deviation of 1 unit. This means that a given observation of pH can be classified by comparing it to the predicted normal pH calculated using the salinity observed at the time of sampling and the appropriate equation in Table 5-2 for the estuary type. The boundaries for these criteria are depicted in Figure 5-1, for comparison with the CRD observations for each system type.

Table 5-2. Recommended pH criteria for Georgia estuarine and coastal waters depending on site salinity (S) and system type. Separate equations define the pH criteria that are either greater or less than "normal".

Type	**Alluvial & Tidewater**	**Blackwater**	**Alkaline Blackwater**
Systems	All others	Satilla/St. Andrew St. Marys/Cumberland	Ogeechee/Ossabaw
Poor	$>8.243 + 0.390*\log_{10}(S+1)$	$>6.250 + 1.688*\log_{10}(S+1)$	$>7.722 + 0.657*\log_{10}(S+1)$
Fair	$>7.743 + 0.390*\log_{10}(S+1)$	$>5.750 + 1.688*\log_{10}(S+1)$	$>7.222 + 0.657*\log_{10}(S+1)$
Normal	$7.243 + 0.390*\log_{10}(S+1)$	$5.250 + 1.688*\log_{10}(S+1)$	$6.722 + 0.657*\log_{10}(S+1)$
Fair	$<6.743 + 0.390*\log_{10}(S+1)$	$<4.750 + 1.688*\log_{10}(S+1)$	$<6.222 + 0.657*\log_{10}(S+1)$
Poor	$<6.243 + 0.390*\log_{10}(S+1)$	$<4.250 + 1.688*\log_{10}(S+1)$	$<5.722 + 0.657*\log_{10}(S+1)$

pH is an immediate indicator of poor water quality, and a single episode of unusually low pH for a given location may be sufficient to cause ecosystem dysfunction for an extended period of time. Therefore, we recommend using both the annual minimum and the annual median values to assess both acute episodic and chronic conditions. We further recommend that refinements of these criteria should be considered in the future as new information becomes available.

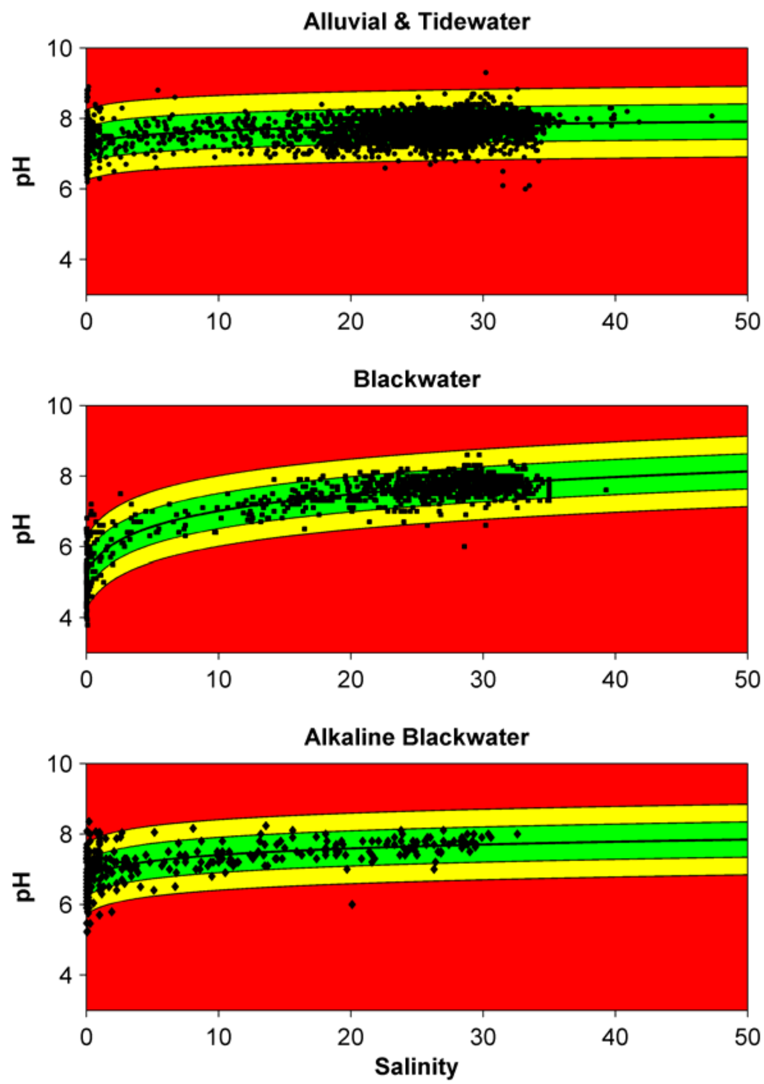

Figure 5-1. Recommended pH criteria for Georgia estuarine and coastal waters depending on site salinity and system type. Green denotes "good" water quality values, yellow denotes "fair", and red denotes "poor". Black dots are observations from the GA DNR CRD dataset.

Dissolved Oxygen

Rationale

As described earlier, hypoxia (low dissolved oxygen (DO) concentration) is often a response to eutrophication. DO affects organisms directly: very low concentrations can kill benthic organisms and widespread hypoxia can lead to fisheries collapses. Sublethal responses to reduced oxygen concentrations include reduced growth and reproduction, physiologic stress, forced migration, reduction of suitable habitat, increased vulnerability to predation, and disruption of life cycles (reviewed in Vaquer-Sunyer and Duarte 2008).

DO is used as an indicator in most water quality studies and is listed in the EPA guidance manual (U.S. EPA 2001b) as an additional primary response variable in systems that have already experienced hypoxia. Considering that 69% of sites sampled by GA DNR CRD experienced DO<= 3 mg L^{-1} at least once during the study period examined here, the need to include DO as an indicator of water quality is clear.

Criteria Development

Many previous studies have developed criteria for evaluating DO. Both the NEEA and NCCR reports used 2 mg L^{-1} to define the "fair/poor" boundary and 5 mg L^{-1} for the "good/fair" boundary (Bricker et al. 1999, 2007; U.S. EPA 2001a, 2004, 2008). The SCECAP reports used 3 mg L^{-1} for the "fair/poor" boundary and 4 mg L^{-1} for the "good/fair" boundary (Van Dolah et al. 2002, 2004, 2006; Bergquist et al. 2009). The U.S. EPA (2000) derived a value of 2.3 mg L^{-1} as the limit of survival of juvenile and adult fish, crustaceans, and bivalves in coastal waters of the Virginian province, and 4.8 mg L^{-1} as the chronic protective value for growth. The GA DNR (2009) water quality control rules for the designated use of "coastal fishing" state that the standard is a daily average of 5.0 mg L^{-1} and no less than 4.0 mg L^{-1} at all times. The rules describe only two classifications, either supporting or not supporting the designated use.

Although there appears to be some agreement on 2 mg L^{-1} to define the "fair/poor" boundary for DO, we question the protectiveness of this value because there is confusion in the literature over the units used to describe oxygen concentrations. Earlier studies that defined hypoxia as oxygen concentrations below 2 mL O_2 L^{-1} (e.g. Diaz and Rosenberg 1995) have sometimes been cited incorrectly as using 2 mg O_2 L^{-1} (e.g. U.S. EPA 2004, 2008). The conversion factor from mL (at standard temperature and pressure) to mg O_2 is 1.4276, so 2 mL O_2 L^{-1} is equivalent to approximately 2.85 mg O_2 L^{-1}. This difference may help explain the results of a recent review of empirical observations by Vaquer-Sunyer and Duarte (2008), which showed that 2 mg L^{-1} is barely protective against fisheries collapse and is inadequate for many benthic organisms. Their survey of published studies of hypoxia effects on benthic organisms shows that a criterion of 3 mg DO L^{-1} would be greater than the median lethal concentration (and therefore protective) for most gastropods, bivalves, and fishes but only about 75% of crustaceans. They further showed that a criterion of 5.5 mg DO L^{-1} would be greater than the median sublethal threshold for most or all cnidarians, echinoderms, polychaetes, mollusks, and crustaceans, but only about 75% of fishes.

Another consideration for setting oxygen criteria is the fact that there is both spatial and temporal variation in oxygen concentration. Dissolved oxygen criteria apply best to bottom waters where hypoxia is most likely to develop and most likely to affect organisms with limited or no mobility. In the GCE-LTER domain (Altamaha, Sapelo, and Doboy Sounds), it is not uncommon to find DO changing by 0.5 mg L^{-1} or more vertically throughout the water column (D. Di Iorio, unpubl.). Furthermore, dissolved oxygen generally shows a diel cycle, with levels increasing during the day due to photosynthesis in excess of respiration and decreasing at night due to respiration. Ideally, criteria should be developed for samples that are taken at places and times when transient hypoxic conditions are most likely to occur.

Recommendations

We recommend using criteria of 3 mg L^{-1} to define the "fair/poor" boundary and 5.5 mg L^{-1} for the "good/fair" boundary for dissolved oxygen concentrations. Although these are slightly higher than the boundaries that have been used in previous studies, the observations considered here represent oxygen concentrations in daytime surface samples. Since they were not collected at either the places or the times when dissolved oxygen would be expected to be lowest, the criteria used to evaluate these observations should be conservative. Moreover, the information compiled by Vaquer-Sunyer and Duarte (2008) suggests that these levels will be protective for most organisms. Although crustaceans are more sensitive to low DO than most fishes, studies of the most commercially important crustacean species in Georgia waters (white shrimp, brown shrimp, and blue crabs) suggest that they tend to avoid waters with < 2 mL DO L^{-1} (= 2.85 mg L^{-1}) (Diaz and Rosenberg 1995), so a criterion of 3 mg L^{-1} should protect against mass migration of these species.

The GA DNR rule (2009) allows for variances for waters with naturally low DO. However, we do not recommend separate criteria for blackwater systems. While the organisms in these areas may be accustomed to low DO, they may be living close to the limits of acceptable physiological conditions. The high capacity for microbial respiration in blackwater systems, combined with inputs of water often already low in DO, can make them particularly vulnerable to any additional inputs of nutrients or organic matter (Meyer 1992). We recommend allowing these systems to be classified as naturally “fair” or “poor” in recognition of this vulnerability and as a reminder that further reductions in DO may not be tolerated.

Dissolved oxygen is an immediate indicator of poor water quality, and a single episode of hypoxia may be sufficient to cause ecosystem dysfunction for an extended period of time. Therefore, we recommend using both the annual minimum and the annual median values to assess both acute episodic and chronic conditions.

Nitrogen and Phosphorus

Rationale

Nutrient input to estuaries can cause eutrophication through the classic sequence wherein increases in inorganic nutrients stimulate excessive algal blooms and potential hypoxia (Howarth and Marino 2006). However, as described in the Introduction, studies in Georgia have shown that hypoxia can also occur via direct stimulation of microbial heterotrophs by organic nutrients (Verity et al. 2006). The question of which nutrient(s) may trigger a eutrophication response is related to the idea of a limiting nutrient, or that nutrient which is in shortest supply relative to amounts needed for growth of phytoplankton or microbes. Primary production by phytoplankton in estuaries is usually limited by nitrogen inputs, although phosphorus can also be important (Howarth and Marino 2006). Phosphorus has also been shown to stimulate microbial production in North Carolina blackwater streams (Mallin et al. 2004) and in South Carolina salt marshes (Sundareshwar et al. 2003).

Virtually all estuarine water quality studies recommend measuring at least some fractions of both the nitrogen and phosphorus pools because the combination of information on nutrients and the other indicators may reveal how eutrophication is occurring. As discussed in the Correlations section, nutrient concentrations in estuaries can be dynamic, as they can be altered by both biological and chemical processes after they have entered the estuary. Ultimately, it is the input of nutrients *to* the system that must be controlled in order to prevent eutrophication. The NEEA indicators suite acknowledged this by using nitrogen load to the system as one component of an “influencing factors” score (Bricker et al. 2007). However, measuring concentrations of nutrients within estuaries is a critical step in understanding eutrophication processes and identifying potential problems.

Criteria Development

Recommendations regarding which fractions (inorganic vs. organic, dissolved vs. particulate, total) of nutrients are the best indicators of water quality are extremely variable. Total nitrogen (TN) and total phosphorus (TP) are recommended by both the U.S. EPA (2001b) and the National Water Quality Monitoring Council (Caffrey et al. 2007) as Tier 1 parameters. TN and TP are also used as indicators by the SCECAP program (Van Dolah et al. 2002, 2004, 2006; Bergquist et al. 2009). However, two panels of experts that have been convened to recommend nutrient indicators, especially regarding eutrophication, have concluded that total dissolved nitrogen (TDN) and total dissolved phosphorus (TDP) are the most appropriate parameters to measure if financial resources for analysis are limited (Bricker et al. 1999, 2007; DiDonato, in press). The National Water Quality Monitoring Council also recommended TDN, TDP, and the particulate fractions PN and PP as Tier 2 parameters. Finally, some would argue that the dissolved inorganic fractions (nitrate + nitrite + ammonium = dissolved inorganic nitrogen (DIN) and orthophosphate = dissolved inorganic phosphorus (DIP)) are the most labile and therefore important to measure (U.S. EPA 2001a, 2004, 2008; Caffrey et al. 2007).

Table 5-3. Nutrient criteria used by national and regional water quality studies to differentiate between “good”, “fair”, and “poor” conditions. N=nitrogen, P=phosphorus. All units are mg (N or P) L^{-1}.

	N Good/Fair			N Fair/Poor		
Study	**TN**	**TDN**	**DIN**	**TN**	**TDN**	**DIN**
NEEA 1999		0.1			1.0	
NCCR II, III			0.1			0.5
SCECAP 1999-2004	0.95			1.29		
SCECAP 2005-2006	0.81			1.05		

	P Good/Fair			P Fair/Poor		
Study	**TP**	**TDP**	**DIP**	**TP**	**TDP**	**DIP**
NEEA 1999		0.01			0.1	
NCCR II, III			0.01			0.05
SCECAP 1999-2004	0.09			0.17		
SCECAP 2005-2006	0.1			0.12		

Regardless of which fractions are measured, establishing appropriate criteria for nutrients is problematic. It would be best to link the concentrations of nutrients to the expected subsequent values of other indicators such as chlorophyll or DO. This has been done in other systems, such as Chesapeake Bay and the Gulf of Mexico, where springtime nutrient inputs are related to chlorophyll and subsequent summer hypoxia (Scavia et al. 2003; Hagy et al. 2004; Scavia et al. 2006). Chlorophyll is not currently measured by the CRD monitoring programs, but we explored the relationships between N and P annual peak concentrations and subsequent minimum DO values (see Correlations section). If the data could be fit with a simple predictive model, our goal was to choose nutrient criteria such that, if observations remained below the criteria, DO would remain above its criteria with 95% confidence. There appear to be higher than usual N and P concentrations in late 2002 followed by lower than usual DO in 2003 (Figures 3-23, 3-27, 3-10). However, the dataset is too limited to be able to tell if this pattern generally occurs in high flow years, especially following a drought. Over the 5 years, there is a weak but significant relationship between annual median TDP and minimum DO the following year (R^2=0.1, $p<0.05$), but no significant relationship between annual median nitrate or DIN and minimum DO. We were unable to establish workable criteria due to the large amount of scatter (unexplained variation) in the data: the confidence intervals for prediction were extremely broad and thus the criteria would have been extremely low. Nevertheless, developing relationships among indicators that reflect the mechanisms that are believed to lead to poor water quality is a worthwhile goal for the future.

None of the national or regional studies discussed above has established mechanistic relationships between their recommended nutrient criteria and their criteria for chlorophyll *a* or dissolved oxygen. The first NEEA report (Bricker et al. 1999) convened panels of experts, and we presume that their recommendations were based on best professional judgment of nutrient levels that may lead to problems in most systems. It used order-of-magnitude differences to distinguish between “good”, “fair”, and “poor” conditions for TDN and TDP (Table 5-3). The NCCR reports evaluate different fractions of the nutrient pools (DIN and DIP), which represent only part of the TDN and TDP pools. However, in NCCR II and III they misquoted the NEEA report by ascribing the threshold values that NEEA provides for TDN and TDP to DIN and DIP, respectively (DiDonato, in press). The “good/fair” TDN and TDP thresholds from the NEEA report were then used as thresholds for DIN and DIP whereas the “fair/poor” thresholds from NEEA were reduced by half, although the reason for this is not explained (Table 5-3). Taken at face value, this suggests that the assumption in the NCCR report is that TDN and TDP are 50-100% inorganic.

Our review of the literature suggests that this fractionation does not hold (at least for TDN) for Georgia coastal waters, where TDN is about 25% DIN[1] and TDP is about 85% DIP[2].

It is more difficult to evaluate the TN and TP criteria from the SCECAP reports because these values include particulate nitrogen (PN) and particulate phosphorus (PP), both of which can be highly variable[3]. However, a larger problem is that the SCECAP reports used the 75th and 95th percentiles of TN and TP data collected in earlier studies as the boundaries between "good/fair/poor". These percentiles changed between the first three reports (Van Dolah et al. 2002, 2004, 2006) and the last (Bergquist et al. 2009) (Table 5-3), which highlights a potential problem with using earlier studies as reference data rather than an objective determination of nutrient concentrations that lead to eutrophication. We do not recommend using percentiles as criteria, which is the approach SCECAP takes throughout their reports, because it makes assumptions about measurements of water quality that may not be true. In this case it is equivalent to deciding *a priori* that 20% of the values are "fair" and 5% are "poor". Although a comparison of site measurements to percentiles from the dataset as a whole may be useful in a relative sense, especially when firm criteria are lacking, it would be better to use comparative terms such as "higher" or "lower".

There are two additional points that must be considered when evaluating nutrient concentrations. First, the timing of sampling with respect to the process of eutrophication can affect which symptoms are observed at a given time. Low nutrient concentrations may be observed in the wake of an event wherein excess nutrients have been taken up by the phytoplankton or the microbial community. In this case the water quality may be degraded due to excessive algal production or respiration, but high concentrations of dissolved nutrients are no longer present. This means that infrequent sampling, especially once per year, is unlikely to coincide with the timing of the levels of nutrients that eventually cause problems. Van Dolah et al. (2004) did not find strong relationships between N and P fractions and chlorophyll *a* and concluded that U.S. EPA criteria for DIN and DIP do not appear to be effective indicators of high phytoplankton concentrations, but this is likely a result of their synoptic sampling design. Ideally it would be best to measure frequently enough to capture both the nutrient inputs and the subsequent response by the algae or microbes in order to be in a position to predict these responses.

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[1] Estimates of the fraction of TDN that is organic (DON) are variable but there is a general consensus that in southeastern coastal waters it is more than half. In a study of Georgia coastal systems, Haines (1979) found that DON was >90% of TDN in coastal shelf waters and more variable in a marsh creek but >50% most of the time. In a study of DON transport to coastal ecosystems by southeastern rivers from 1974-1993, DON averaged 50 to 90% of TDN at the most downstream stations in five Georgia rivers (Savannah, Ogeechee, Altamaha, Satilla, and St. Marys) with an overall mean of 75% (Alberts and Takács 1999). The lower figure of 50% for the Altamaha River is in general agreement with an estimate of 60% during 2000-2002 (Weston el al. 2003). In the tidal Skidaway River during 1986-1996, DON was about 90% of TDN (Verity 2002a). Among SCECAP sites overall, DON was about 81% of TDN (Van Dolah et al. 2004). As a rough estimate, we assume that DON is about 75% of TDN, and DIN is about 25%, in Georgia coastal waters.

[2] The fraction of TDP that is organic (DOP) is also variable. Among SCECAP sites overall, DOP was about 16% of TDP (Van Dolah et al. 2004). In Apalachicola Bay, DOP was about 60% of TDP (Mortazavi et al. 2000). There is no consensus in these estimates, but given the similarity between Georgia and South Carolina estuaries, we use the SCECAP proportions as rough estimates of the TDP pool and assume that about 15% of TDP is DOP, and 85% is DIP, in Georgia coastal waters.

[3] Estimates of the particulate fractions of the TN and TP pools vary widely: in a study of the waters offshore from the mouths of Sapelo, Doboy, and Altamaha Sounds and a Sapelo Island marsh creek, the particulate fraction of TN was about 30% in coastal shelf waters but highly variable (19 to 66%) in the marsh creek (Haines 1979). Estimating the particulate fraction of the TP pool is even more problematic because of adsorption and desorption of phosphorus onto clay particles (Pomeroy et al. 1965). In the GCE-LTER domain on the Georgia coast, particulate phosphorus (PP) is generally about half to two-thirds of TP but highly variable (K. Hunter, pers. comm.). In Apalachicola Bay, PP was about 59% of TP (Mortazavi et al. 2000).

A second consideration is the fact that the amount of nutrient load that can lead to degraded water quality depends on the capacity of the system for dilution as well as the transit time (the time the nutrients spend between entrance and exit). Estuaries with short transit times can transport excess nutrients and phytoplankton out of the system before water quality degrades (although this may just move a problem downstream), whereas estuaries with longer transit times may be more vulnerable to the progression of eutrophication symptoms. The transit times of dissolved substances generally depend on the rate of throughput of freshwater through the estuarine system as well as the rate of exchange of seawater through tidal actions (Sheldon and Alber 2006). The transit times of particulate substances may be different and even more complex because of the interactions of particle settling velocities and water currents (Jay et al. 2000). The transit times of an individual estuary may also change seasonally with freshwater inflow, making it even more difficult to establish the levels of nutrient loads and concentrations that may lead to eutrophication. The effects of transit time may also interact with other factors that could affect the rate of uptake of nutrients, such as temperature and light availability for photosynthesis. Grazing pressure on phytoplankton may also affect the formation of blooms. Therefore, any nutrient criteria that are developed to be applied to a wide range of estuarine systems must necessarily be very general guidelines.

Recommendations

Our primary recommendation is for Georgia to measure both total dissolved (TDN and TDP) and particulate (PN and PP) material, and then use this information to calculate total nitrogen (TN = TDN + PN) and phosphorus (TP = TDP + PP). This approach accomplishes several things. First, it is dissolved nutrients that get taken up by phytoplankton and microbes, so information on TDN and TDP is probably the best indication of the potential drivers of eutrophication. As mentioned above, TDN and TDP have been recommended by several panels of experts as the most appropriate parameters to measure if financial resources are limited. Second, it allows for the estimation of TN and TP. These are listed as core parameters in the EPA guidance documents, and monitoring of these parameters is likely to be required in the near future. Collecting information on both the dissolved and particulate fractions will put Georgia in position to comply with these requirements, and will also facilitate comparison with national efforts.

Although there are a number of ways to measure TN and TP, a third advantage to measuring them via their components is that it includes the direct estimation of PN and PP. The high turbidity of coastal Georgia waters (and the variable nature of that turbidity, which changes over the course of a tidal cycle), may lead to high and variable estimates of particulate nitrogen (PN) and particulate phosphorus (PP), which will in turn cause high and variable measurements of TN and TP. As described above, both NEEA and NCCR have modified their water clarity criteria for southeastern U.S. estuaries in recognition of the fact that a single national standard cannot be applied fairly to all regions. If TN and TP are likely to be the nutrient indicators used for comparison at the national level then we anticipate similar problems with regional differences that may be due primarily to the particulate fraction. We suggest being proactive and collecting information on both the dissolved and particulate nutrient fractions so that southeastern regional conditions can be represented accurately from the outset.

While the recommendations above will satisfy many program requirements, we also recommend measuring DIN (nitrate+nitrite, ammonium) and DIP (orthophosphate) at selected sites. This would provide direct information on the nutrients available to phytoplankton, and also allow the organic fractions to be determined by subtraction (DON = TDN – DIN, DOP = TDP – DIP). Given the two potential pathways to eutrophication in coastal Georgia waters, decaying algal blooms and direct stimulation of microbial heterotrophy, information on the relative importance of organic and inorganic nutrients would provide the greatest understanding of eutrophication and its potential causes. We suggest doing these additional measurements at least quarterly at sentinel sites (see the Recommendations section). Another approach would be to collect and store extra filtered water for each sample, and then determine the inorganic and organic fractions only if TDN and/or TDP exceed the “good” level criteria.

Until more localized criteria can be developed, we recommend using the NEEA (Bricker et al. 1999) criteria for TDN and TDP in Georgia coastal waters.

Nitrogen and phosphorus nutrient compounds in estuarine waters are required at low levels to maintain normal ecosystem functions and are generally not toxic at levels routinely encountered. A single high pulse of nutrients may lead to a problematic bloom, depending on other factors as outlined above, but chronically high nutrient loads are generally the larger concern. Therefore, we recommend monitoring the annual median values as general indicators of water quality.

Chlorophyll **a**

Rationale

Excess algal biomass (a "bloom") is the most obvious symptom of classic eutrophication, and it frequently leads to other problems. While a bloom is in progress, it may attenuate light penetration to the point of harming any rooted vegetation (SAV) that may be present. If grazing pressure is insufficient to consume the phytoplankton, then sinking and decay of the bloom may lead to hypoxia or even anoxia in deeper waters. Harmful algal blooms (HABs) of species that release toxic substances are of special concern: they may cause discomfort and illness to humans, even those nearby on land; kill fish and other aquatic organisms; and make others unpalatable or toxic to eat, thus causing problems throughout the food web as well as economic harm due to fishery losses (Paerl 1988). Any suite of indicators to detect eutrophication should include chlorophyll *a* (or some other measure of algal biomass) as a required parameter (U.S. EPA 2001b ; Caffrey et al. 2007).

Criteria Development

While a full analysis of phytoplankton pigments can be very informative about the types of phytoplankton that are present (Schlüter et al. 2000), chlorophyll *a* alone is thought to be a good general indicator of algal biomass (but see Kruskopf and Flynn 2006).

The chlorophyll *a* criteria that are used by the national water quality studies mentioned above were developed for the first NEEA report (Bricker et al. 1999) and then used in the second NEEA (Bricker et al. 2007) as well as the NCCRs (U.S. EPA 2004, 2008). The criteria are 5 $\mu g\ L^{-1}$ as the "good/fair" threshold and 20 $\mu g\ L^{-1}$ as the "fair/poor" threshold. The first SCECAP report (Van Dolah et al. 2002) also compared South Carolina data to these criteria. The second and third SCECAP reports (Van Dolah et al. 2004, 2006) used 20 $\mu g\ L^{-1}$ as the "fair/poor" threshold but increased the "good/fair" threshold to 12 $\mu g\ L^{-1}$ to match the 75th percentile of the data. The fourth report re-evaluated the 75th and 90th percentiles and set the criteria at 11.5 and 16.4 $\mu g\ L^{-1}$, respectively (Bergquist et al. 2009).

Recommendations

We strongly recommend adding chlorophyll *a* to the Georgia CRD monitoring programs. It is a critical response variable that can be used to evaluate whether algal biomass increases in response to nutrient inputs. Chlorophyll *a* was used in every national and regional survey of water quality that we examined, and it is on the EPA list of core parameters so monitoring may be required in the future. Additionally, if a harmful algal bloom is suspected, sampling and analysis should be undertaken to identify the causal organism.

We have no chlorophyll data to examine in the current GA DNR CRD dataset, and we do not recommend the South Carolina percentile method for the reasons outlined above, so we recommend using the NEEA criteria to evaluate chlorophyll data until a more detailed analysis of Georgia data can be undertaken.

Phytoplankton blooms are episodic by nature, and it is possible that a single bloom could cause eutrophication symptoms severe enough to cause lasting damage. Therefore we recommend monitoring the annual maximum value as an indicator of seasonally acute conditions, as well as the annual median as an indicator of chronic problems.

Transparency

Rationale

It is usually desirable to measure some aspect of water clarity as an indicator of light availability for photosynthesis by phytoplankton. Clear water is also valued in other states for maintaining SAV beds as important habitat for other organisms. However, light limitation can also change the balance between nutrient uptake by autotrophs and heterotrophs, with consequences for the potential pathway of eutrophication and the severity of symptoms. Algal biomass can itself affect water clarity, but so can other factors such as suspended sediments and humic substances; therefore chlorophyll *a* is not a reliable measure of clarity and a separate measurement should be taken.

Criteria Development

Several different methods have been used to measure water clarity. Perhaps the simplest approach is to use a Secchi disk, which provides an integrated measure of transparency throughout the water column. There is a great deal of historic data on Secchi depth (the depth at which the Secchi disk is no longer visible) because it is a simple and inexpensive measurement. More recently, light attenuation (percent light transmission at a reference depth, where the light measured is photosynthetically active radiation (PAR)) has been measured using light meters equipped with quantum sensors. This method differs from Secchi depth in the integration depth (usually 1 m for light attenuation vs. variable Secchi depth), the wavelengths used (PAR vs. visible) and the sensor accuracy (calibrated light sensor vs. human eye, with associated differences among operators). Although percent light transmission is more accurate, light meters are much more expensive than Secchi disks. Another method for measuring water clarity is to measure turbidity using a nephelometer. Unlike Secchi depth and percent light transmission, which account for light attenuation due to both absorption and scattering, turbidity is a measure of light scattering only and is insensitive to light absorption.

National studies have tended to focus on measures of transparency as opposed to turbidity. Bricker et al. (1999) used criteria of 3 m and 1 m Secchi depth as thresholds to distinguish between “good/fair” and “fair/poor”, respectively. The first NCCR report (U.S. EPA 2001a) used a single criterion of 10% light transmission at 1 m to differentiate between “good” and “poor” for all regions, and they equated this to a Secchi depth of 0.5 m. Later NCCR reports (U.S. EPA 2004, 2008) have acknowledged the naturally high turbidity of southeastern U.S. estuaries and established separate, more lenient criteria for states that do not expect to support SAV (including South Carolina and Georgia). Those criteria are 10% and 5% light transmission at 1 m as the thresholds for “good/fair” and “fair/poor”, respectively.

It is possible to roughly correlate Secchi depth and light attenuation measurements across waters of similar turbidity. Smith et al. (2006) established correlations for three water clarity classes of Gulf of Mexico estuaries. (Disk size and the optical properties of the waters of interest may affect these relationships and, ideally, they should be derived specifically for each system.) Using their Table I and Eq. 2, the 10 and 5% light transmissions at 1 m recommended by NCCR II for the thresholds in southeastern waters would roughly correspond to Secchi depths (20 cm diameter black and white disk) of 0.5 and 0.3 m, respectively. This is in agreement with the relationship used in NCCR I (U.S. EPA 2001a). In contrast, the 3 m and 1 m Secchi depth criteria suggested by Bricker et al. (1999) in the NEEA report are comparable to 57% and 18% light transmission at 1 m. These higher values are more in keeping with the 40% and 20% transmission criteria suggested by Smith et al. (2006) for low-turbidity waters supporting SAV beds, and are probably not applicable to Georgia coastal waters.

Both GA DNR CRD and SCECAP measure turbidity in their monitoring programs. South Carolina established saltwater criteria for turbidity using their 75^{th} and 90^{th} percentile method, and the resulting "good/fair" and "fair/poor" thresholds are 15 and 25 NTU, respectively (Van Dolah et al. 2002, 2004, 2006). There are no general relationships between turbidity and measurements of transparency, so it is hard to compare these turbidity criteria with those established in the national reports. Although correlations between Secchi depth and NTU have been developed for specific watersheds or regions, these are site-specific[4]. Ott et al. (2006) found that color was better than turbidity at predicting Secchi depth in six of eight Florida estuaries. We therefore cannot compare the SCECAP thresholds with the transparency criteria used in national studies.

Recommendations

The difficulty of comparing the GA DNR CRD turbidity data with other studies leads us to recommend that CRD switch to a measure of transparency. Another reason for this recommendation is that the Nutrient Criteria Technical Guidance Manual for Estuarine and Coastal Marine Waters (U.S. EPA 2001b) specifically mentions water clarity as a required parameter, so it may well be required in the future. If CRD does make this switch, we recommend that they continue to measure nephelometric turbidity along with the new method at all sites for at least several months in order to establish correlations that are relevant for Georgia coastal waters. It would then be possible to relate the turbidity data already in hand to clarity criteria established for other methods.

Although either Secchi depth or light transmission would measure transparency, we recommend the use of a PAR meter if possible. Secchi depth measurements are simple and the equipment cost is minimal, but measurements are prone to differences among operators and may be impossible in shallow water. The EPA guidance manual acknowledges that Secchi depth is a widely used method, but they recommend a switch to light meters. If the cost of light meters is prohibitive, then we suggest borrowing one for a period of time for comparison with Secchi depths and establishment of site-specific relationships between the two.

As for criteria to be used for transparency data collected in the future, we recommend the light (PAR) attenuation criteria established by U.S. EPA (2004, 2008) for turbid southeastern estuaries: 10% and 5% transmission at 1 m depth as the thresholds for "good/fair" and "fair/poor", respectively. If Secchi disks are used, then we recommend the corresponding values derived above, 0.5 m and 0.3 m, until site-specific relationships can be developed.

A single episode of poor water transparency, as after a storm that temporarily suspends a large amount of sediment, is not necessarily a concern unless the duration is long. Chronic conditions are the greater concern. Therefore, we recommend monitoring the annual median values as general indicators of water quality.

Biochemical Oxygen Demand (BOD)

Rationale

One suggested cause of hypoxia in coastal Georgia waters is the direct stimulation of microbial respiration by organic and inorganic nutrients (Verity et al. 2006). Biochemical oxygen demand (BOD) provides a way to measure the potential for this to occur. In a BOD assay, water is incubated at a standard temperature and length of time and the resulting decrease in oxygen is recorded. This provides information on how quickly the microbial populations that are present can break down the substrates in the sample (including decaying algal blooms).

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[4] A study of New York lakes and rivers reaffirmed the site-specific nature of the relationship between Secchi depth and turbidity but at the same time provided some useful bounds on the relationship: Secchi depth of 0.5 m corresponded to about 15-20 NTU, 1m corresponded to about 5-10 NTU, and 3 m corresponded to about 2-4 NTU (Effler 1988). Steel and Neuhauser (2002) compared four methods for measuring water clarity or turbidity in the Skagit River (WA) and found that log (vertical Secchi disk readings) correlated fairly well with log (turbidity measured with an electronic nephelometer) (r=-0.86).

Although 5-day BOD incubations are traditional (BOD_5), varying the incubation time provides information on the relative lability of the substrates present in the sample. In a sense, this mimics differing flushing times: during shorter incubation times (faster flushing) there may be time for only the most labile substances to be broken down and consumed, whereas with longer incubations (slow flushing), the more refractory substances may be used as well. This may be addressed by comparing BOD_5 with a 20-day analysis (BOD_{20}). The longer incubation times can reveal how much of the more refractory substances may eventually be utilizable by bacteria and may be more appropriate for slowly flushed systems. Mallin et al. (2006) found that in North Carolina rivers, lakes, and streams, BOD_5 was often more strongly correlated with chlorophyll *a*, while BOD_{20} was often more strongly correlated with turbidity, total suspended solids, total phosphorus, and total nitrogen, but there was also a great deal of overlap where both BOD parameters were equally well correlated with a third parameter.

Criteria Development

There is well-developed guidance for BOD in wastewater management applications, but these criteria are geared towards regulating the exogenous BOD *load to* receiving waters. We have been unable to find any clear guidelines or standards for BOD *concentration in* estuarine waters. The SCECAP program (Van Dolah et al. 2002) established criteria based on 75th and 90th percentiles of historical data, which were 1.8 and 2.6 mg BOD_5 L^{-1}, respectively. As noted above, these do not necessarily indicate thresholds for "good/fair" and "fair/poor" water quality, but they do give us regional values for comparison. The SCECAP program dropped BOD from its list of indicators after 2000 because of the lack of documented criteria (Van Dolah et al. 2004).

Recommendations

We recommend that BOD_5 be added as an indicator of the hypoxic potential of Georgia coastal waters. This will provide information on the potential for the microbial pathway of eutrophication. BOD_5 is listed in the U.S. EPA's (2001b) suggested methods and will be more useful than BOD_{20} for comparison with other programs, but we also encourage a focused study comparing BOD_5 with BOD_{20} to ascertain the relative importance of labile and refractory components of the BOD in Georgia estuaries.

Since GA DNR CRD does not currently measure BOD, we cannot establish criteria values at this time and instead suggest that a future analysis be undertaken to relate measured BOD_5 to subsequent DO minima to use as guidance for establishing criteria.

A single episode of elevated BOD is not necessarily a concern unless other conditions (temperature, slow flushing) also favor the development of hypoxia. Chronic conditions are the greater concern. Therefore, we anticipate monitoring the annual median values as general indicators of water quality.

Ancillary Data: Salinity, Specific Conductance, and Temperature

Measurements of salinity, specific conductance, and temperature are required for interpreting the indicator parameters described above. As general environmental parameters, they help to describe seasonal and interannual changes and climatic trends, giving context to the water quality information supplied by the indicator parameters. In addition, knowledge of these parameters is necessary for correctly processing the data for many of the indicators.

Salinity is important as a general characteristic of estuarine habitats: the normal range in a given location often determines in large part the community of organisms that reside there. As ancillary information, it is a reflection of freshwater input to the site, whether by streamflow, direct rainfall, or overland runoff. Correlation of other parameters with salinity can often be informative. For example, when high nutrient concentrations are correlated with low salinity it implies that the nutrients were associated with freshwater input. Salinity can also interfere with some analyses (e.g. ammonium by the Koroleff (1983) method) and therefore must be measured as part of a correction factor.

Specific conductance is generally redundant with salinity as environmental data, but it should be included as part of the data collection because it is part of the instrument readout (no additional effort), and the relationship between salinity and specific conductance can be a good check on instrument calibration and operational procedures.

Temperature is likewise an important habitat characteristic, an index of seasonality, and an important moderator of the rates of estuarine processes. It is also part of the normal instrument readout and a necessary factor, along with conductivity, in the calculation of salinity. There is a Georgia state criterion for the temperature of coastal fishing waters, 90°F, that is useful in regulating warm effluents but it is difficult to imagine how the temperature of natural waters could be regulated.

These parameters are not generally evaluated as being “good” or “poor” unless they are well outside their normal ranges of variability, and they are not generally regulated (or even able to be controlled) except in the case of effluents that would be substantially different from their receiving waters.

Summary

Table 5-4. Proposed indicators, criteria, metrics, and ancillary data for assessing the general quality of Georgia coastal and estuarine waters. TBD = to be determined.

Indicator	Units	Good	Fair	Poor	Metric
pH: 3 system types: Alluvial & Tidewater Blackwater Alkaline Blackwater	pH unit deviation from established relationship between pH and salinity for system type	<0.5	0.5 - 1	>1	Annual minimum Annual median
Dissolved oxygen (surface, daytime)	mg L^{-1}	>5.5	3 – 5.5	<3	Annual minimum Annual median
TDN	mg L^{-1}	<0.1	0.1 - 1.0	>1.0	Annual median
TDP	mg L^{-1}	<0.01	0.01 - 0.1	>0.1	Annual median
Chlorophyll *a*	µg L^{-1}	<5	5 - 20	>20	Annual maximum Annual median
Transparency	A: % transmission at 1m B: Secchi depth (m)	A: >10 B: >0.5 or TBD	A: 5 - 10 B: 0.3 - 0.5 or TBD	A: <5 B: <0.3 or TBD	Annual median
BOD_5	mg L^{-1}	TBD	TBD	TBD	Annual median
Ancillary Data					
Salinity	PSU				
Specific Conductance	mS cm^{-1}				
Temperature	°C				

For the Future: Microbial Indicators

Some water quality studies (Van Dolah et al. 2002, 2004, 2006; Bergquist et al. 2009) have included measurements of microbial populations, especially fecal coliforms, as part of their indicator suites. The indicator species themselves are often not the organisms responsible for causing illness, but they should be species that specifically indicate sewage or other fecal contamination and correlate with the presence of disease-causing organisms or the incidence of gastrointestinal and other illness. Microbial indicators can provide information that can be important for human health, particularly in terms of fishing, swimming, or other recreational water contact.

The most commonly measured microbial indicator is fecal coliform abundance. While it has long been assumed that these bacteria would be reliable indicators of fecal contamination, studies have shown that the standard analysis responds to organisms that are not from fecal sources (U.S. EPA 1986; Doyle and Erickson 2006). This may explain the studies that have shown that “fecal coliforms” do not in fact correlate well with incidence of gastrointestinal illness (Wade et al. 2003). The continuing requirement by the Food and Drug Administration (FDA) to use fecal coliform as the indicator organisms for safe shellfish consumption has been cited as an impediment to adopting methods known to be better (Doyle and Erickson 2006), and the GA DNR water quality rules recognize the fact that fecal coliform abundance is not necessarily a reliable indicator of human health (Georgia DNR 2009). Until the FDA adopts a new indicator, however, GA DNR CRD will be required to measure fecal coliforms in shellfish harvesting areas. We do not recommend prolonging the use of this assay and have therefore omitted it from our suite of indicators.

Other bacterial indicators have been proposed (Frischer and Verity 2006). Enterococci have been shown to be a better, if still imperfect, indicator of fecal contamination in marine waters (U.S. EPA 1986), and they are now measured as indicators of marine beach water quality (U.S. EPA 1986). However, issues regarding source tracking and the relative risks from enterococci from wildlife and other animal, as opposed to human, sources have been raised (Frischer and Verity 2006). *Enterococcus* abundance is already in use as a water quality indicator at Georgia coastal beaches for the purpose of protecting human health. The question of whether it would also be useful as a general indicator of water quality, regarding eutrophication, remains unresolved. Its most likely use would be as an indicator of fecal sources of nutrients to estuarine waters, but nutrient data were not collected at the beach sites so we are unable to establish whether such a relationship exists. Our studies of the *Enterococcus* abundance at Georgia Tier 1 coastal beaches (Sheldon 2009a,b,c) showed that it was correlated with dissolved oxygen at only 1 of 17 sites (and then only weakly), and it was correlated weakly with turbidity at 8 sites. Occasional correlations with pH (3 sites) were probably related to freshwater input conditions (rainfall and streamflow), the primary parameters that correlated with *Enterococcus*.

An indicator that is related specifically to human health would be a valuable addition to the suite of indicators proposed here, but we recommend waiting until a better national consensus on microbial indicators can be achieved before adopting one for assessing general water quality.