An Approach for Developing Numeric Nutrient Criteria for a Gulf Coast Estuary

U.S. Environmental Protection Agency

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Disclaimer

This report provides information to States, Indian tribes (hereafter tribes), and other authorized jurisdictions for the purpose of assisting development of numeric nutrient criteria for adoption into water quality standards under the Clean Water Act (CWA). Under the CWA, States and tribes are to establish water quality criteria to protect designated uses. State and tribal decisionmakers retain the discretion to adopt approaches on a case-by-case basis that differ from the approaches used in this report when appropriate and scientifically defensible. Although this report provides hypothetical numeric nutrient criteria that would serve to protect resource quality and aquatic life, it does not substitute for the CWA or U.S. Environmental Protection Agency's (EPA's) regulations nor is it a regulation itself. Thus, it cannot impose legally binding requirements on EPA, States, tribes, or the regulated community and may not apply to a particular situation or circumstance.

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Executive Summary

Nutrient enrichment is currently a major cause of impaired water quality and degraded ecosystem health in coastal marine ecosystems such as estuaries. The Clean Water Act (CWA) legislation provides much of the legal framework for water quality management. Effective management of nutrient enrichment in estuaries under the CWA presently is inhibited by the lack of numeric criteria for nutrients and nutrient-related water quality indicators. We describe and demonstrate a manageable, science-based approach for developing water quality criteria for nutrients (nitrogen and phosphorus), chlorophyll-a, dissolved oxygen (DO), and water clarity for a single estuary, Pensacola Bay, FL, with which we were very familiar. As a component of the approach, we modified the U.S. Environmental Protection Agency's (EPA's) percentile approach to compare water quality indicators in three salinity zones of Pensacola Bay with comparable values from other estuaries. We also conducted a broad-based quantitative assessment of both nutrient-related conditions and processes in the bay and nutrient inputs to the bay, which contribute to a logical process leading to determination of nutrient criteria.

To apply the percentile approach, it is necessary to identify a class of water bodies, estuaries in this case, that can be compared reasonably. We defined a class of estuaries on the basis of EPA's Level III ecoregions and Florida's Level IV ecoregions. Within these schemes, Pensacola Bay is located within ecoregion 75 and ecoregion 75a, respectively. Estuaries were defined according to the National Oceanic and Atmospheric Administration's coastal assessment framework, which identifies estuarine drainage areas. We assembled median summer surface water quality data for 28 estuaries in Level III ecoregion 75 and 7 estuaries in Level IV ecoregion 75a. Three salinity zones (<5, 5 to 18, and >18) also were defined so that water quality in each salinity zone of Pensacola Bay could be

compared with water quality within corresponding salinity zones in other estuaries. Chlorophyll-a in Pensacola Bay was comparable with the median for ecoregion 75 and was somewhat higher than other systems in ecoregion 75a. Nutrients and water clarity in oligohaline water were also similar to the median. In contrast, water clarity and nutrient concentrations in mesohaline and higher salinity waters were in the best 25% of estuaries (>75th percentile for water clarity and <25th percentile for nutrients).

Our analysis of major nutrient-related water quality concerns in Pensacola Bay found that seasonal hypoxia and seagrass loss were the major issues. Seasonal hypoxia affected an average of 25% of the bottom area in recent studies. Seagrass loss is extensive in the system. About 95% of seagrass habitat was lost between 1950 and 1980. Almost no recovery was observed in surveys in 1992 and 2003. Yet, other indicators of ecosystem function point to lower nutrient-related impacts. Nitrogen and phosphorus loading rates to the system are moderate. Phytoplankton productivity was moderate (320 g carbon m^{-2} $year^{-1}$) as was plankton and benthic metabolism. Nutrient concentrations were generally low, and phytoplankton were nutrient limited. Studies indicate that hypoxia may result significantly from a natural propensity toward oxygen depletion, whereas the extent of seagrass loss may be a legacy of formerly more degraded water quality conditions.

Nutrient yields from Pensacola Bay's largely forested watershed are low (262 kg nitrogen km^{-2} $year^{-1}$ and 21.8 kg phosphorus km^{-2} $year^{-1}$). Concentrations of nitrogen in rainfall are also relatively low (14 μM). However, rapid population growth in the coastal counties of the watershed may cause increased nutrient loading in the future. We conclude that water quality criteria for nutrients and nutrient-related water quality measures could be based reasonably on

currently observed conditions because evidence that more stringent criteria are scientifically defensible, necessary, or even achievable, is lacking. Hypothetical criteria resulting from application of this approach are presented.

The approach to criteria development that we describe could provide a broad template for analyses in other systems. It would be most useful for systems that, like Pensacola Bay, do not have clear evidence of water quality or water quality-related ecological impacts resulting from anthropogenic nutrient enrichment. Conversely, systems with very high nutrient inputs resulting from point sources, landscape changes, or other anthropogenic causes may require substantial nutrient reductions. Although elements of our analytical approach may be useful as part of an approach for determining criteria for these systems, additional elements will be needed to determine reduced levels for nutrient concentrations and nutrient-related water quality measures that are appropriate to the system and protective of designated uses.

1. Introduction

1.1 Background

The U.S. Congress responded to the desire of the American people to aggressively protect U.S. waterways from pollution with passage of the 1972 Federal Water Pollution Control Act and subsequent amendments, which, since the 1977 amendments, have been known collectively as the Clean Water Act (CWA; U.S. House of Representatives 2000). The CWA put into place a water quality standards process, within which ambient water quality criteria are centrally featured. Ambient water quality criteria are concentration limits determined such that the legally identified or "designated" human and aquatic life uses of a water body are not impaired by the target pollutant. Since passage of the CWA, standard approaches have been developed for establishing aquatic life criteria for toxic chemicals, generally utilizing experimentally determined acute and chronic laboratory toxicity data for multiple species (Stephan et al. 1985). Management actions enabled by the CWA have successfully eliminated the most egregious toxic impairments.

Nitrogen and phosphorus are major plant nutrients that occur naturally in varying amounts and are necessary for normal ecological function of aquatic ecosystems. Nutrients become pollutants, however, when concentrations or loadings are increased excessively. Anthropogenic nutrient enrichment often causes eutrophication (Nixon 1995) and a related spectrum of usually adverse environmental impacts. Coastal nutrient enrichment has increased during the past half century and has not been curtailed effectively by water quality management under the CWA (NRC 2000). In the United States, water quality impairment is tracked by the U.S. Environmental Protection Agency (EPA) via a list of impaired waterways known as the Section 303(d) list, referring to the relevant section of CWA legislation. Nutrient enrichment or nutrient-related effects (e.g.,

oxygen depletion, algal growth, harmful algal blooms) combine as the leading cause for inclusion on the Section 303(d) list (U.S. EPA/OWOW, public communication). Global trends in coastal nutrient enrichment are similar in many regards to those in the United States (GESAMP 2001).

In 1998, EPA expressed its view (U.S. EPA 1998) that establishing numeric criteria for nutrients and related water quality indicators (e.g., chlorophyll-a) was an important step toward reducing nutrient impacts. Others have expressed a similar perspective. A survey of water quality managers for 18 U.S. estuaries found that one of the major impediments to effective management of eutrophication was a lack of regulatory authority to require nutrient reductions, including a lack of numeric nutrient criteria (NRC 2000). In the absence of numeric criteria, determination of water quality impairment generally is based on narrative water quality criteria. Commonly referred to as "free-froms," narrative criteria describe in qualitative terms what undesirable environmental conditions are proscribed (e.g., water should be free from excessive algal blooms caused by nutrient enrichment). A problem with narrative standards is that a procedure for interpreting the standard to determine numeric targets must be developed before regulatory actions can be imposed (U.S. EPA 1999a). With the possibility of regulatory action undermined by poorly defined standards, many efforts to manage eutrophication rely primarily on nonregulatory (i.e., voluntary, educational) approaches (NRC 2000). For example, the more than 20-year, high-profile effort to restore the Chesapeake Bay has emphasized voluntary rather than regulatory approaches, with numeric criteria being proposed only recently (U.S. EPA 2003). Adoption of numeric standards for nutrients and nutrient-related water quality indicators could simplify both determination of nutrient-related water

quality impairments and implementation of regulatory actions to improve water quality. Regulatory approaches ultimately may be needed to reverse current trends.

There are several reasons why most water bodies are presently protected under narrative rather than numeric nutrient criteria:

- natural nutrient concentrations vary widely among aquatic ecosystems, requiring that numeric criteria be derived on a regional or even site-specific basis;
- nutrient enrichment effects usually are expressed in relatively complex ways that preclude the use of experimental assays, similar to toxicity tests, to derive nutrient criteria;
- sensitivity to nutrient enrichment is expected to vary significantly among systems; and
- ambient nutrient concentrations are usually not as useful as indicators of biological effects as are ambient concentrations of toxic chemicals (e.g., high nutrient concentrations may reflect a lack of nutrient limitation, whereas low nutrient concentrations may have been caused by an active phytoplankton bloom).

Despite the challenges of deriving ambient water quality criteria for nutrients, EPA adopted a national strategy to develop nutrient criteria in 1998 (U.S. EPA 1998) and has since published technical guidance and recommended nutrient criteria for most U.S. rivers, streams, lakes, and reservoirs (e.g., U.S. EPA 2000a,b,c,d). EPA published technical guidance for establishing nutrient criteria for estuaries and coastal marine waters (U.S. EPA 2001) but stopped short of recommending specific approaches. Determining nutrient criteria for estuaries may be more complicated than for many freshwater systems because of a number of factors that include those below.

- Estuaries are influenced to varying degrees by exchanges of matter and energy (e.g., resulting from tides) at their seaward margins.
- Widely varying physical and chemical conditions strongly influence biotic communities in estuaries (Kennish 1986). Feedback effects contribute to significant water quality differences among systems.

- Estuaries often are subject to multiple stressors influencing nutrient-related water quality. Examples include fishing, dredging, and exotic species invasions.
- Strong gradients in water quality are often present within estuaries, complicating specification and implementation of numeric criteria.

In view of these observations, developing a scientific basis for nutrient criteria could require an understanding of nutrient inputs, transport, fate, and the important biological response mechanisms of the resident biotic communities in each system (NRC 2000). The failure in most cases of simple nutrient dose-ecological response models has been noted as a reason that complex simulation models may be needed to directly model complex ecological behavior related to nutrients in coastal waters (Fitzpatrick and Meyers 2000, but see Boynton and Kemp 2007 for examples of simple models that illustrate predictable ecological responses). Cloern (2001) outlined how the field of coastal eutrophication research has progressed largely from limnological concepts to the point where it is grappling increasingly with the additional complexity of coastal marine systems.

Recognizing the importance of developing numeric nutrient criteria and also the difficulty of doing so, this study was undertaken to demonstrate an approach that could be used to determine numeric nutrient criteria (and criteria for nutrient-related response indicators) for a single coastal system (Pensacola Bay, FL) with which we were very familiar. Like many estuaries, Pensacola Bay has not received intense research and management focus. Thus, this study demonstrates that conclusions can be reached regarding criteria with reasonable but still incomplete scientific information. To address our objectives, we undertook the following:

- examined the data available to characterize current and past water quality conditions in the Bay,
- reviewed the scientific knowledge regarding the environmental conditions and processes influencing water quality in the bay that could inform nutrient criteria setting,

- compared water quality conditions in Pensacola Bay with a group or class of estuaries that we expected to share similar ecological attributes, and
- synthesized the assembled information using a "weight-of-evidence" approach to suggest scientifically justifiable water quality criteria for nutrient and nutrient-related water quality indicators.

To place our demonstration in a larger context, we discuss the potential for applying the demonstrated procedure to other estuaries.

1.2 Study Site

The Pensacola Bay system is a complex of estuarine bays arranged in two major arms that combine to form Pensacola Bay proper. The component bays include Escambia Bay, Blackwater Bay, East Bay, and Pensacola Bay (Figure 1). Santa Rosa Sound is excluded from this analysis even though it adjoins the bay, because in terms of important ecological processes, it is relatively distinct. The combined Pensacola Bay system is medium-sized (370 km^2) and shallow (mean depth = 3 m) and has been characterized as a partially stratified, drowned river valley estuary (Schroeder and Wiseman 1999). Tides are diurnal and have a low amplitude, ranging from 15 to 65 cm. The basin includes three major watersheds that drain via the Escambia, Blackwater, and Yellow rivers. The Escambia River discharges into Escambia Bay, whereas the Blackwater and Yellow rivers discharge into Blackwater Bay and East Bay, the

eastern branch of the system. Both branches join Pensacola Bay, which connects to the Gulf of Mexico through the narrow (800-m-wide) Pensacola Pass (Figure 1).

The condition of the Pensacola Bay system, particularly that of Escambia Bay, became a matter of public concern as early as the late 1960s (Olinger et al. 1975), earlier than in many estuaries. Initial evidence for a pristine condition was based on biological surveys of stream fauna (Academy of Natural Sciences of Philadelphia 1953). By the early 1960s, after the establishment of significant industrial point-source discharges (especially ammonium [NH_4^+] and organic matter), similar biological surveys indicated declining health in the river. Reports of fish kills and declining fisheries outputs ultimately led to actions that, by the mid 1970s, greatly reduced the point sources and, in some
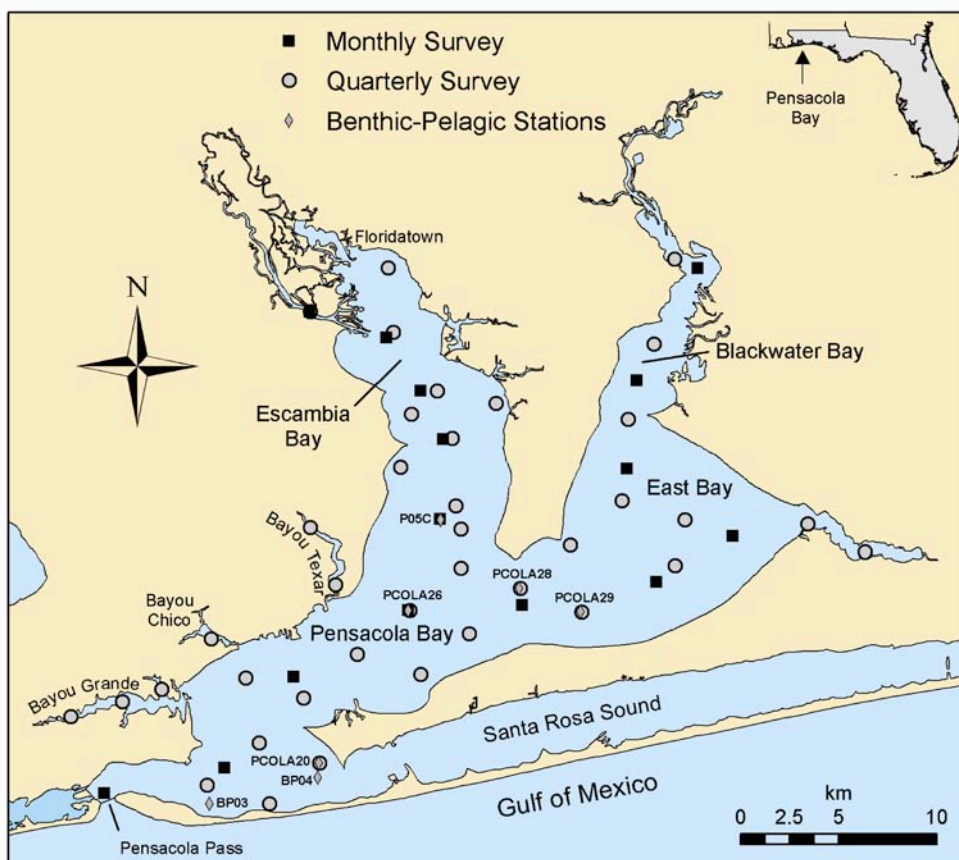
Figure 1. Locations of stations sampled during two major studies of water quality conditions in the Pensacola Bay system (Table 1) and a 2003-to-2004 study of benthic and planktonic metabolism. Monthly survey stations were sampled approximately monthly from March 2002 through November 2004. Quarterly survey stations were sampled quarterly from 1996 to 2000. Station identifications for benthic-pelagic stations correspond to rates reported in Table 4. Santa Rosa Sound is not considered in this study to be part of the Pensacola Bay system.

cases, eliminated them completely. Olinger et al. (1975) provides a remarkable early compilation of ecological conditions in the bay, intended principally to document the recovery of the system following reductions in industrial waste loads. No comparable data were collected after 1975 until bay-wide water quality surveys were begun by EPA beginning in 1996 (U.S. EPA 2005). Whereas differences in survey methodology and data reporting mostly preclude quantitative analysis of ecological changes, comparing early data and the conclusions of the early investigators with more recent studies suggests that neither the ecological conditions nor the nature of the major ecological concerns have changed dramatically in the past 30 years. The major ecological concerns in 1975 were bottom water hypoxia; loss of seagrass habitats; toxic contamination; and degradation of biotic communities, including fisheries. Bottom water hypoxia and loss of seagrass habitats, both of which are likely to be related to nutrient enrichment, continue to be concerns for Pensacola Bay (U.S. EPA 2005). U.S. EPA (1999b)

reported that sediment toxicity in the Pensacola Bay system is confined mainly to the three urban bayous (Bayou Grande, Bayou Chico, and Bayou Texar; Figure 1). However, toxic contamination of seafood, especially by mercury, remains a regional issue in the Gulf of Mexico. Blue crab (*Callinectes sapidus*) in Pensacola Bay have been shown to be contaminated with polychlorinated biphenyls, likely acquired by bioaccumulation
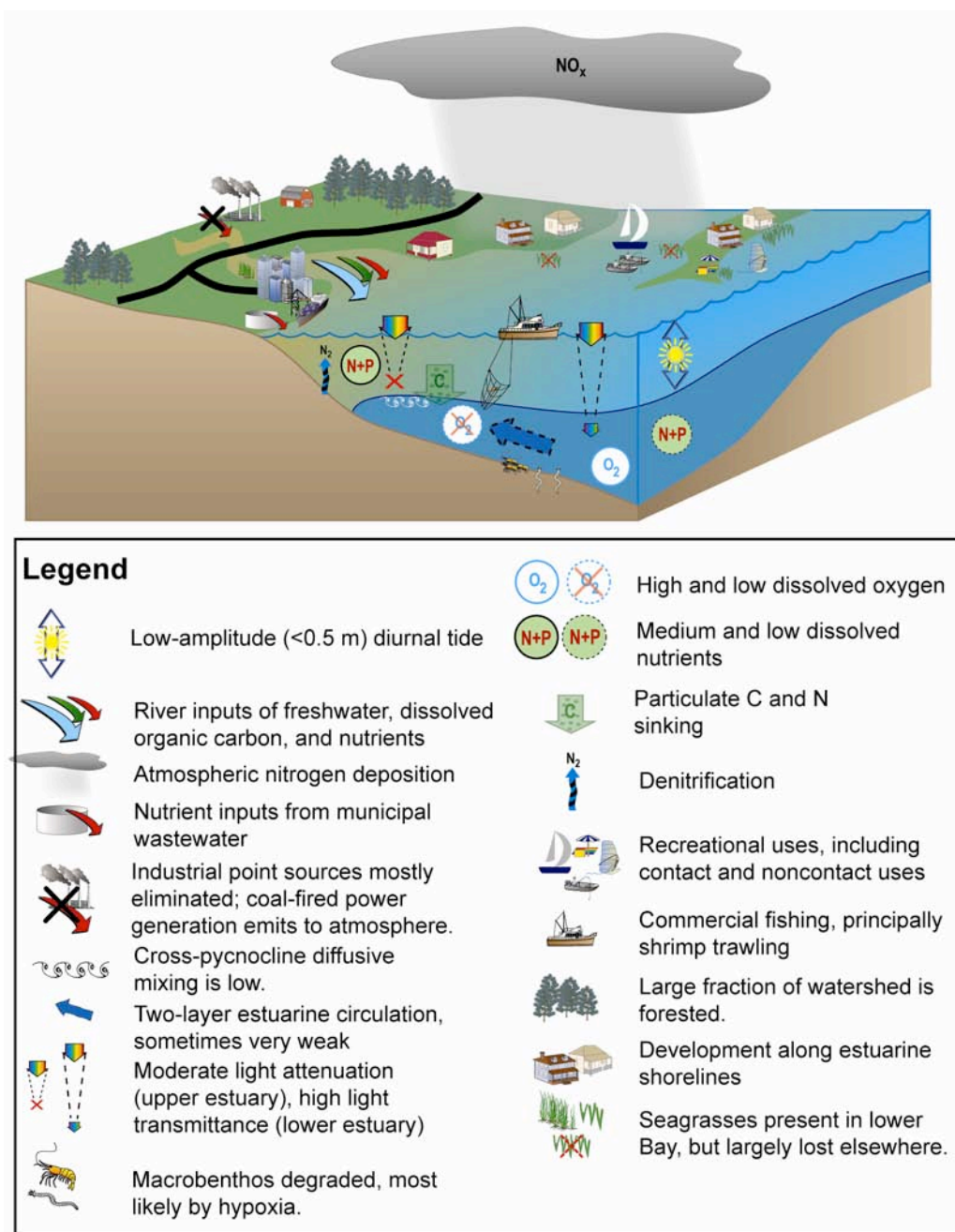
Figure 2. A conceptual model illustrating historic, ecologic, and socioeconomic factors of potential importance for developing and evaluating water quality management options for Pensacola Bay. Iconography from the University of Maryland Center for Environmental Science Integration and Application Network, http://ian.umces.edu.

from residual contamination of sediments (Karouna-Renier et al. 2007).

Many of the important ecological processes and environmental concerns, including the human and aquatic life uses of the bay and the major stressors of the ecosystem can be summarized effectively and communicated in a conceptual model (Figure 2). Developing such a model, or at least acquiring the information necessary to do so, is probably an important step toward determining a scientifically sound procedure for deriving nutrient criteria. By comparing and contrasting conceptual models for different systems, one can gain a greater understanding of the similarities and differences among those systems and how those differences may impact criteria development. We include our conceptual model without extensive discussion here because many aspects of the model are addressed in subsequent sections.

2. Methods

This study is based on "found" data from several types of sources, including national databases, water quality surveys, and aerial remote sensing. The following describes the data collected and either the methods or appropriate references describing the methods.

2.1 Water Quality Data for Pensacola Bay

Two major studies provided the majority of the water quality data for Pensacola Bay used in this study (Table 1). In the first study (hereafter "quarterly survey"), 38 sites selected using a probabilistic survey design (Summers et al. 1995, U.S. EPA 2005) were sampled quarterly (spring, summer, fall, and winter) during surveys beginning in May 1996 and concluding in February 2001. In the second study (hereafter "monthly survey"), 29 water quality surveys were conducted approximately monthly from April 2002 through December 2003 and from May 2004 through November 2004 (e.g., Hagy and Murrell 2007). Monthly surveys included 15 sites oriented along two axial transects of Pensacola Bay (Figure 1). Additional water quality data were obtained from Florida's Inshore Marine Monitoring and Assessment Program (IMAP; McRae et al. 2004), which collected a total of 64 observations on multiple water quality variables in the Pensacola Bay system between 2000 and 2004. Most observations (53 of 64) were made during an intensive survey in 2003.

2.2 Ecoregional Water Quality Characterization

To compare water quality in Pensacola Bay with other estuaries in the region, water quality statistics were summarized for estuaries in EPA's Level III ecoregion 75 (Omernik 1995), encompassing the southeast coastal plain and Florida's Level IV ecoregion 75a (Griffith et al. 1994), encompassing the west Florida panhandle region eastward to the Big Bend region of Florida (Figure 3). Because these ecoregions are defined to encompass regions of land, not water, estuaries were associated with an ecoregion when the land surrounding the estuary was substantially within ecoregion boundaries. Thus, Florida's ecoregion 75a includes all inshore marine waters from Perdido Bay eastward to the Suwannee River (Figure 3). Water quality data for the ecoregional characterization were obtained from IMAP (McRae et al. 2004) and EPA's National Coastal Assessment (NCA; U.S. EPA 2004). Both assessment studies utilized a probabilistic survey

Table 1. Sources of water quality data used in this study. Water quality data include physical data (water temperature, salinity, and DO), water clarity (secchi depth), chlorophyll-a, and nutrients (NO_3^-, NO_2^-, NH_4^+, PO_4^{3-}, total nitrogen, total phosphorus).

Data Set	Source	Survey Design/Scope/Resolution
Pensacola Bay Quarterly Survey (U.S. EPA 2005)	EPA/ORD	Probabilistic survey design, May 1996 to February 2001, Approximately 40 stations sampled quarterly. Surface and bottom.
Pensacola Bay Monthly Survey (Hagy and Murrell 2007)	EPA/ORD	Transect surveys, 2002 to 2004, 15 stations sampled monthly. Surface and bottom, with 25-cm profile of physical variables.
Florida Inshore Marine Monitoring and Assessment Program (McRae et al. 2004)	Florida Marine Research Institute	Florida coastal waters, probabilistic survey design. 2000 to 2004, late summer. 30 stations per year statewide, with rotating schedule of regional focus at 180 stations per year. 64 station visits in Pensacola Bay system, most in 2003.
National Coastal Assessment (U.S. EPA 2004)	EPA/ORD	National scope. 2001 to 2002, late summer. Approximately 375 stations in southeast coastal plain.
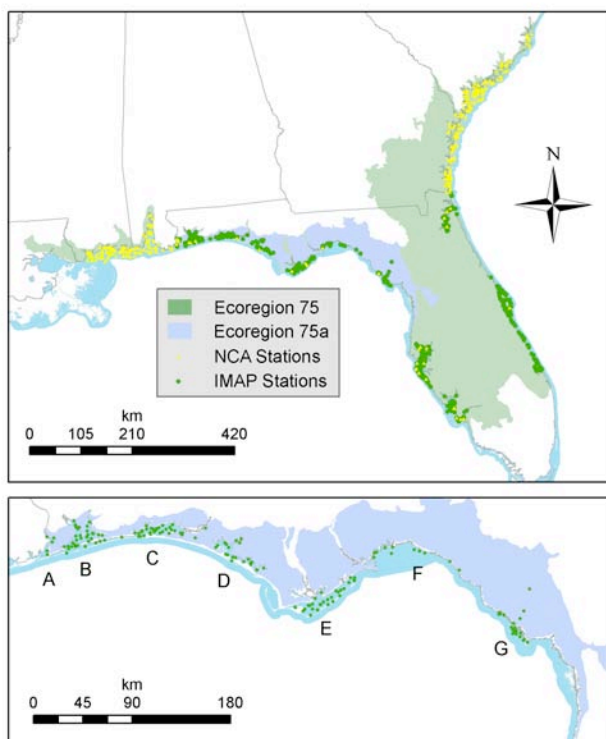
Figure 3. The locations of water quality monitoring stations visited under EMAP during 2001 and 2002 and IMAP during 2000 to 2004. The distribution of sampling stations within the southern coastal plain ecoregion (region 75 of EPA Level III ecoregions) (upper panel) and within the Gulf Coastal Flatwoods region (Florida's Level IV ecoregion 75a) (lower panel). NCA stations were omitted from the ecoregion 75a analysis. Estuarine drainage areas in ecoregion 75a, as defined under NOAA's coastal assessment framework, are identified as follows: A = Perdido Bay, B = Pensacola Bay, C = Choctawhatchee Bay, D = St. Andrew Bay, E = Apalachicola Bay, F = Apalachee Bay, and G = Suwannee River.

design with sampling conducted only during a late summer index period (Table 1, Figure 3).

The procedure for water quality characterization (Figure 4) involved computing, for each variable of interest, the median of all water quality observations in each of three salinity zones in each estuary. We defined three salinity zones based on surface salinity values: (1) oligohaline (salinity < 5), (2) mesohaline (salinity 5 to 18), and (3) polyhaline (salinity > 18). We defined each estuary according to the boundaries of estuarine drainage areas (EDAs) defined by the National Ocean and Atmospheric Administration's Coastal Assessment Framework (NOAA 1999a). Data from IMAP and NCA were referenced to EDAs within the coastal assessment framework using a geographic information

system (ArcGIS 9.2, ESRI, Inc.). Data not falling within EDAs, such as those falling within "coastal drainage areas," were excluded from the analysis.

2.3 Aerial Submerged Aquatic Vegetation Surveys

Data describing the extent of submerged aquatic vegetation (SAV) in Pensacola Bay were obtained from investigations that employed interpretation of aerial photographs taken as early as 1949 and as recently as 2003. Maps of SAV coverage for circa 1950 were obtained from Olinger et al. (1975), who interpreted photographs obtained for highway construction. For this study, maps from Olinger et al. (1975) were scanned to TIFF files, geo-referenced against corresponding modern shorelines, and then manually digitized to a vector coverage at the same resolution as the original map (modern shoreline extracted from 1:100,000 scale digital line graphs by the U.S. Geological Survey (USGS) National Wetlands Research Center, Gulf Breeze, FL, Project Office). Tabulated SAV coverages for 1960, 1980, and 1992, were obtained from Schwenning et al. (2007), which also describes the photo-interpretation and digitization protocols (also see Lores et al. 2000). SAV coverage data for 1992 and tabulated SAV
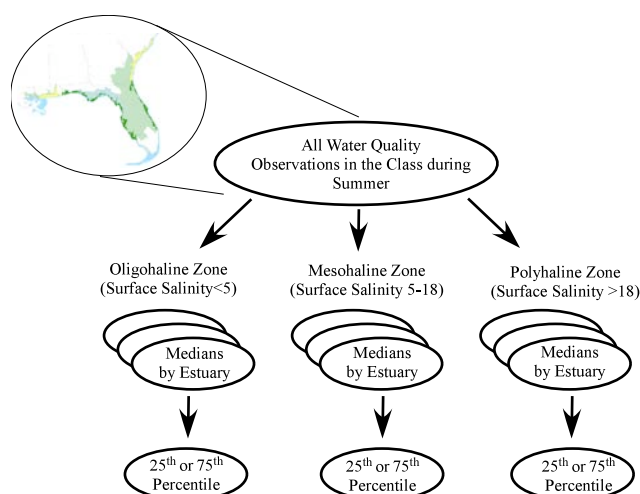
Figure 4. Procedure for computing statistical comparison values for estuaries. The 25th percentile of estuary medians was used for comparison values for water quality indicators where lower values are better (e.g., nutrients, chlorophyll-a). The 75th percentile was used for secchi depth, where higher values indicate higher water clarity.

coverage data for 2003 were obtained from the USGS National Wetlands Research Center, Lafayette, LA.

2.4 Watershed Characteristics and Land Use

Data describing land use in the Pensacola Bay watershed were obtained from the 2001 National Land Cover Database using the Multi-Resolution Land Cover Consortium Viewer (Homer et al. 2004; http://www.mrlc.gov/). The database provides land use and land cover classifications at 30-m resolution. Classifications are derived from Landsat-7 Enhanced Thematic Mapper imagery and additional supporting data. Land cover data for the Pensacola Bay watershed were extracted from the larger data set and tabulated by classification using the ArcGIS 9.2 geographic information system indicated above.

3. Results and Discussion

3.1 Ecoregional Comparisons and the Percentile Approach

Identifying appropriate benchmarks or "reference conditions" against which to compare water quality conditions is a key feature of EPA's recommended approaches for developing nutrient criteria (U.S. EPA 2000a,b). We modified EPA's percentile approach for freshwaters to define benchmark water quality values that have potential applicability to Pensacola Bay. A central feature of the approach, which we applied here, is identification of a suitable class of similar estuaries from which a reference condition can be determined. In the context of nutrients, similarity refers to a similarity in ecological attributes that influence the response of a system to nutrient inputs.

There are many classification schemes for coastal systems, each serving different objectives and emphasizing different ecological attributes (Kurtz et al. 2006). Unfortunately, there is little consensus regarding which classification schemes would be most appropriate in terms of predicting response to nutrient inputs. None have been demonstrated to be useful for that purpose. One concern is that, even though many important attributes of estuaries have been considered (e.g., residence time, mean depth, ratio of watershed area to estuary area; Kurtz et al. 2006), a much broader set of factors (e.g., climate, geology, character of offshore water quality) likely contributes to the ecological character of ecosystems.

As an alternative to the available estuarine classifications, we utilized EPA's Level III ecoregions and, simultaneously, the more finely resolved Level IV ecoregions for Florida to define a class of estuaries for Pensacola Bay. Pensacola Bay lies within the "southern coastal plain" ecoregion 75 in the Level III scheme (Figure 3). The shores of Pensacola Bay include several of Florida's Level IV ecoregions, but the "Gulf coastal flatwoods region" (ecoregion 75a)

encompasses much of the coast surrounding Pensacola Bay and a significant span of coastline mostly to the east of the bay (Figure 3). Although the definition of these ecoregions is based on climatological, geological, and biological attributes (Omernik 1987) and pertains to the watersheds of estuaries rather than to the estuaries themselves, we expect that the ecological character of inshore coastal waters must reflect to some degree the ecological attributes of the surrounding land. Moreover, because the ecoregions define geographically contiguous regions of the coast, we expect similarity in many attributes to arise simply from proximity. Important differences can occur, however, particularly when the ecoregion spans oceanographically distinct regions. For example, tides vary dramatically within ecoregion 75; minimal diurnal tides occur on the northern Gulf coast, whereas large semidiurnal tides occur along the Atlantic coast of Georgia and South Carolina. These differences alone contribute to other large ecological differences, such as the presence of extensive intertidal salt marshes in Georgia (Dame et al. 2000), which practically define the Georgia coast, versus the more limited extent of salt marshes in west Florida. The smaller spatial extent of Florida's Level IV ecoregions eliminates the most glaring within-class differences, but at the cost of a class that includes fewer systems. Even though it is likely that important differences remain within even these smaller classes, further division leads toward the perspective that every estuary is completely unique, precluding any role for comparative ecological analysis in development of nutrient criteria.

U.S. EPA (2000a,b) define a variety of options for determining reference conditions for classes of freshwater systems. The preferred approach is to derive water quality criteria on the basis of conditions in relatively pristine water bodies. When a set of sufficiently pristine water

bodies can be identified, criteria have been based on the 75th percentile of water body median values for such pristine sites (U.S. EPA 2000a,b). Given the pervasive human presence in many coastal marine areas, we assumed that few, if any, estuaries were sufficiently pristine to adapt this approach for estuaries. Another approach is a historical reference condition in which water quality conditions observed prior to significant anthropogenic impacts provide a basis for comparison. For most estuaries, limited availability of historical data precludes this approach. The record of historical water quality data for Pensacola Bay is unusually extensive, but, as is common, early monitoring was undertaken in response to obvious pollution impacts (Olinger et al. 1975) and does not quantify the unimpaired or pristine condition. Thus, historical data cannot be used to characterize pristine water quality conditions in Pensacola Bay. Lacking either pristine sites or adequate historical data, we derived levels we call "comparison values" from the statistical distribution of water quality conditions in all the estuaries in the class (Figure 4), adapting the percentile approach of U.S. EPA (2000a,b). Because the class includes sites subject to a range of nutrient enrichment, we utilized the 25th percentile of estuary median water quality values to derive comparison values. EPA (2000a,b) cite empirical evidence from lakes to support this choice when reference quality sites are not available. We did not assume that there is any relationship between the 25th percentile of estuary median water quality and water quality in a hypothetical set of pristine estuaries. We simply chose to examine the 25th percentile as a potentially useful reference point in the distribution of estuary medians. For water clarity, the same ideas apply, but higher values indicate "better" water quality. Therefore, we used the 75th percentile as a comparison value. We contrasted values based on two classes: (1) all estuaries in ecoregion 75 (Appendix A) and (2) the subset of those estuaries within ecoregion 75a (Appendix B).

Our application of the percentile approach differs from that of U.S. EPA (2000a,b) in several important ways. First, because the available regional-scale data sets from NCA and IMAP include only late summer data (Table 1), our analysis is limited to summer (June through September). We do not see this as a major limitation because most water quality problems related to nutrient enrichment, especially hypoxia, are expressed primarily during summer. Thus, we assumed that, if water quality supports the designated uses during summer, uses outside of summer are also likely to be met. The second difference is that we divided observations into salinity zones, based on the principle that water quality parameters in estuaries commonly vary along the salinity gradient. We selected three zones (salinity < 5, 5-18, and > 18) with the objective of resolving differences associated with the salinity gradient, while ensuring that a sufficient quantity of data was available in each zone in most estuaries. Our objectives in defining the zones likely could have been met with different definitions (i.e., breakpoints); however, an attempt to define values for a greater number of salinity zones likely would have resulted in excessive parsing of the available data. Finally, we consider the values obtained via this approach to be guidelines or comparison values, not values that can be adopted defensibly as criteria without further scientific support.

The computed median values show that summer water quality in a significant portion of estuaries in ecoregions 75 and 75a was characterized by low nutrient and moderate chlorophyll-a concentrations, with concentrations generally lower in ecoregion 75a than in ecoregion 75 as a whole (Figure 5, Table 2). Concentrations of chlorophyll-a, nitrogen, and phosphorus are commonly higher in Mid-Atlantic estuaries (NOAA 1997). Nutrient concentrations are commonly higher in Pacific coast estuaries influenced by coastal upwelling (NOAA 1998). Summer median nutrient concentrations and water clarity in Pensacola Bay were comparable to or lower than the 25th percentile (higher than the 75th percentile for water clarity) for the ecoregion, with a few exceptions.
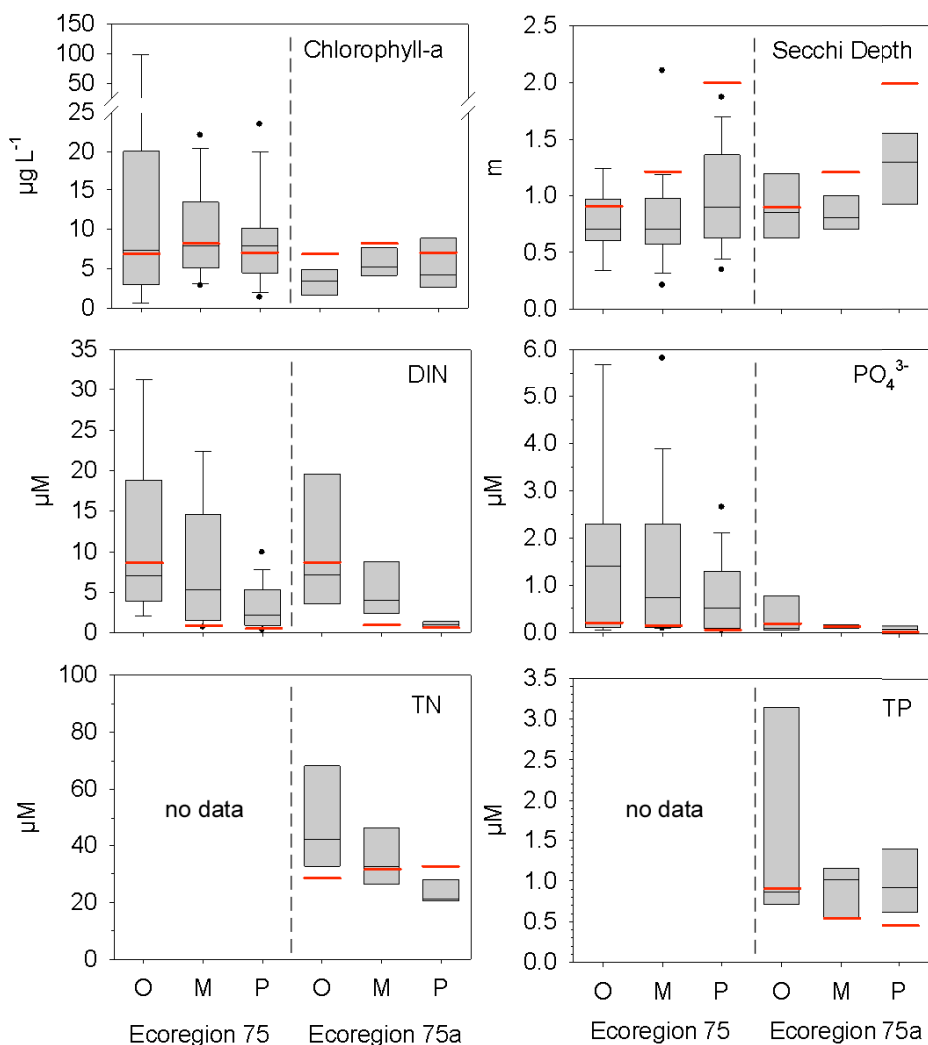
Figure 5. Distribution of summer estuary median surface water quality values by salinity zones for estuaries in Level III ecoregion 75 (left three plots) and Level IV ecoregion 75a (right three plots). Salinity zones are oligohaline (O; salinity < 5), mesohaline (M; salinity 5-18)], polyhaline (P; salinity > 18). Red lines indicate summer medians by salinity zone for Pensacola Bay. Identical values are shown in both the left and right halves of each plot. DIN = $NO_2^- + NO_3^- + NH_4^+$; TN = total nitrogen, TP = total phosphorus.

Bay than the 25th percentile values in all salinity zones for both ecoregion 75 and ecoregion 75a (Table 2). Instead, median chlorophyll-a in Pensacola Bay was similar to the median for ecoregion 75 and closer to the upper quartile for ecoregion 75a (Figure 5).

The statistical approach described here is a repeatable, quantitative scheme for computing water quality measures that can serve as comparison values for both causal variables (e.g., nutrients) and nutrient-related response variables such as water clarity (secchi depth) and chlorophyll-a. In the absence of water quality thresholds clearly tied to loss of ecological integrity, these comparison values are useful for placing values for any one estuary in perspective. There are several factors that limit the range of interpretations for these comparison values. One

Inorganic nitrogen and phosphorus concentrations in oligohaline waters were higher than the 25th percentile, but concentrations in higher salinity water were comparable or lower than the corresponding 25th percentile in those salinity zones. Total phosphorus (TP) concentrations were comparable to the 25th percentile values across the salinity zones, whereas total nitrogen (TN) concentrations were comparatively lower in oligohaline waters and higher in polyhaline waters (Figure 5, Table 2). Median chlorophyll-a was higher in Pensacola

question is whether the 25th percentile (or 75th percentile for water clarity) of estuary medians is an appropriate comparison value, and whether it approximates a minimum condition protective of designated uses as suggested by EPA (2000a,b). A broader comparison with the distribution of values in the ecoregion (i.e., Figure 5) gives a more complete comparative perspective but still leaves open the question of whether a particular water quality value supports use attainment. Water quality values at the 25th percentile of the ecoregional distribution could be either

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Table 2. Ecoregional water quality conditions for chlorophyll-a (Chl-a), secchi depth, dissolved inorganic nitrogen (DIN = NO_2^- + NO_3^- + NH_4^+), phosphate (PO_4^{3-}), total nitrogen (TN), and total phosphorus (TP) computed for EPA's Level III ecoregion 75 and Florida's Level IV ecoregion 75a. Values are the 25th percentile of summer estuary medians, except for secchi depth, where the value is the 75th percentile of estuary medians. Summer medians for Pensacola Bay are shown for comparison. Data are for surface water. Values in parentheses for ecoregion 75 and 75a are the number of estuaries included for each variable. Values in parentheses for Pensacola Bay are the number of observations included. TN and TP values were not computed for ecoregion 75 because data were only available for a portion of the ecoregion. Pensacola Bay medians were computed from a combined data set that includes EPA's EMAP quarterly surveys (1996 to 2001, EPA's monthly surveys (2002 to 2004), and Florida's IMAP surveys. IMAP sampled Pensacola Bay intensively in 2003 and also visited two stations in 2004.

Group	Chl-a ($\mu g\ l^{-1}$)	Secchi Depth (m)	DIN (μM)	PO_4^{3-} (μM)	TN (μM)	TP (μM)
Oligohaline (Salinity < 5)						
Ecoregion 75	3.8 (16)	0.93 (12)	2.9 (15)	0.11 (15)	nd	nd
Ecoregion 75a	2.1 (6)	1.6 (4)	2.6 (6)	0.023 (6)	23.4 (6)	0.67 (6)
Pensacola Bay	5.6 (103)	0.9 (93)	8.7 (102)	0.18 (86)	29.2 (49)	0.88 (32)
Mesohaline (Salinity 5-18)						
Ecoregion 75	5.1 (21)	1.0 (21)	1.3 (20)	0.13 (21)	nd	nd
Ecoregion 75a	4.1 (7)	1.0 (7)	0.81 (7)	0.065 (7)	26.5 (7)	0.56 (7)
Pensacola Bay	7.3 (233)	1.2 (198)	0.97 (210)	0.11 (122)	31.7 (59)	0.55 (47)
Polyhaline (Salinity > 18)						
Ecoregion 75	4.3 (28)	1.3 (28)	0.59 (27)	0.11 (28)	nd	nd
Ecoregion 75a	3.5 (6)	1.4 (5)	0.47 (6)	0.052 (6)	20.5 (6)	0.73 (6)
Pensacola Bay	5.8 (107)	2.0 (100)	0.44 (102)	0.048 (82)	32.7 (57)	0.43 (56)

nd = No data

nonprotective of uses or excessively restrictive for a particular estuary. A second concern is the assumption that the estuaries in either ecoregion 75 or ecoregion 75a are an appropriate "class," and that Pensacola Bay is a member of that class. We suggest that the percentile approach is likely to produce values in a reasonable range for estuaries, but that, because the approach is inherently independent from effects-based considerations and is not certain to generate appropriate values, the values should be evaluated in the context of other information available about the estuary in question before they could be used to determine criteria. In the following sections, we discuss ecological conditions and processes in the Pensacola Bay system to provide a thorough evaluation of the science that may be pertinent to selection of numeric nutrient criteria.

3.2 Algal Biomass, Productivity, and Community Composition

Chlorophyll-a is an easily monitored and conceptually appealing indicator of trophic status and eutrophication in aquatic systems, obvious reasons why it already is used widely in water quality management related to nutrients. Other simple descriptors of the phytoplankton community, such as productivity; community composition, including presence of harmful algae; and growth dynamics, also can provide useful insights. We assembled basic information about the phytoplankton community in Pensacola Bay to explore its implications, if any, for managing nutrients in the bay.

Seasonal and salinity zone chlorophyll-a medians varied between 1.6 and 7.4 $\mu g\ L^{-1}$, with the highest values during summer (5.6 to 7.3 $\mu g\ L^{-1}$; Table 3). Both summer and non-summer values were "low" to "medium" according to standard ranges described by NOAA (1997). "High" concentrations (>20 $\mu g\ L^{-1}$) occurred very infrequently (Table 3, Figure 6). Of 1,390 chlorophyll-a measurements in the Pensacola Bay system between 1996 and 2004, only 36 (2.5%) were greater than 20 $\mu g\ L^{-1}$. Murrell et al. (2007) measured phytoplankton production rates in Escambia Bay using carbon

Table 3. Seasonal median chlorophyll-a in Pensacola Bay surface waters and its interquartile range (25th to 75th percentile). Salinity zones were determined from surface water salinity.

	Median Chlorophyll-a ($\mu g \, L^{-1}$)	Interquartile Range ($\mu g \, L^{-1}$)
Oligohaline (Salinity < 5)		
Winter	2.2	(1.2 - 3.7)
Spring	4.0	(2.6 - 5.4)
Summer	5.6	(3.8 - 8.7)
Fall	1.6	(1.2 - 3.7)
Mesohaline (Salinity 5-18)		
Winter	3.5	(2.6 - 5.2)
Spring	5.8	(3.7 - 8.4)
Summer	7.3	(5.1 - 10.7)
Fall	5.0	(3.1 - 7.2)
Polyhaline (Salinity > 18)		
Winter	3.4	(2.6 - 4.7)
Spring	3.8	(2.1 - 4.8)
Summer	5.8	(3.8 - 8.1)
Fall	3.4	(2.5 - 6.0)

14 uptake methods and estimated that annual phytoplankton productivity was 320 g carbon m^{-2} $year^{-1}$ (0.88 g carbon m^{-2} day^{-1}). Among estuaries, where annual production commonly varies between 100 and 500 g carbon m^{-2} $year^{-1}$ (Boynton et al. 1982, Boynton and Kemp 2007), the annual production rate for Pensacola Bay is "mesotrophic" or moderate. The summer phytoplankton assemblage is dominated by cyanobacteria (Murrell and Lores 2004, Murrell and Caffrey 2005), whose low chlorophyll-a content relative to carbon (phycoerythrin is the dominant light-gathering pigment in most cyanobacteria; MacIntyre et al. 2002) undoubtedly contributes to the high ratio of annual productivity to chlorophyll-a. The phytoplankton community in Pensacola Bay is usually nutrient limited and can be limited by either phosphorus or nitrogen (Murrell et al. 2002, Juhl and Murrell 2005). Relief of nutrient limitation associated with periods of increased freshwater and nutrient inputs manifests as increased chlorophyll-a concentration, increased phytoplankton production (Murrell et al. 2007), and a greater relative abundance of eukaryotic phytoplankton (Juhl and Murrell 2005) in Pensacola Bay.

The available information points to several key conclusions regarding the phytoplankton community in Pensacola Bay. First, there is no clear evidence that the phytoplankton community is stimulated strongly by excess nutrients, or that it is unbalanced or otherwise causing adverse ecological effects. If chlorophyll-a concentrations were frequently in NOAA's (1997) high (20 to 60 $\mu g \, L^{-1}$) or "hypereutrophic" (>60 $\mu g \, L^{-1}$) range, or, if annual phytoplankton productivity was among the highest found in estuaries, one might have a priori cause for concern regarding excess phytoplankton. Median summer chlorophyll-a concentrations were generally higher than the 25th percentile of systems within the ecoregion, especially for ecoregion 75a (Table 2). However, the small differences are probably not ecologically significant. We do not know of any study directly implicating chlorophyll-a concentrations in the low range observed in Pensacola Bay with failure to support any human or aquatic life uses. Experimental evidence (Murrell et al. 2002, Juhl and Murrell 2005) shows that phytoplankton biomass and production in Pensacola Bay are limited by nutrients. This provides a mechanistic expectation that increased nutrient inputs will
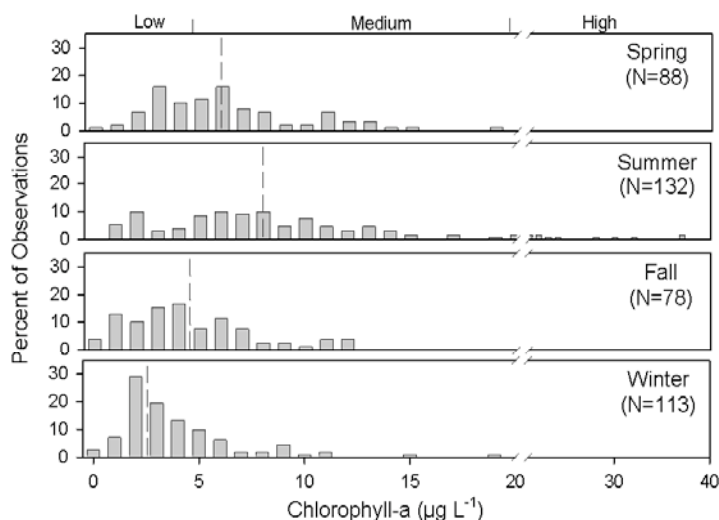

Figure 6. Frequency histogram of surface chlorophyll-a concentrations in mesohaline (surface salinity = 5-18) waters of the Pensacola Bay system from 1996 to 2001. "Low," "Medium," and "High" designations refer to descriptors used by NOAA (1997). Dotted lines indicate the median concentration. Data are from the EPA/ORD quarterly surveys of Pensacola Bay.
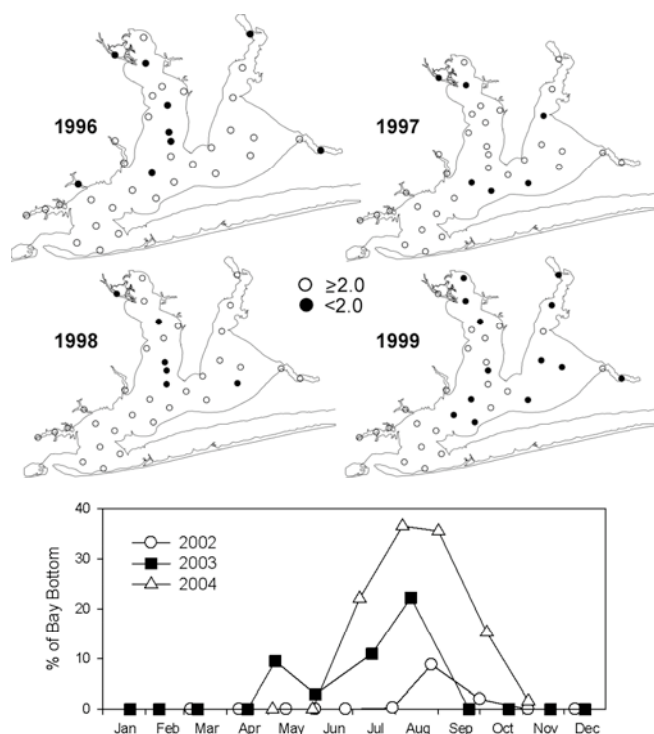
Figure 7. Observations of bottom dissolved oxygen in the Pensacola Bay system during the summers of 1996 through 1999 indicating the locations and frequency of observations less than 2.0 mg L^{-1}, a level commonly used as an operational definition of hypoxia. Data are from EPA/ORD quarterly surveys (upper panels). The computed extent of hypoxia in Pensacola Bay by month from 2002 to 2004, based on EPA/ORD monthly transect surveys (lower panel).

increase phytoplankton production and possibly phytoplankton biomass. Unfortunately, simple linear dose-response relationships fail to capture the relationship between nutrient loads or concentrations and chlorophyll-a in Pensacola Bay (Murrell et al. 2007). Relationships that we expect to observe on the basis of known causal mechanisms often are hidden by the natural complexity that characterizes estuarine ecosystems (e.g., flushing, food web interactions) and, therefore, are not easily seen in field observations (Cloern 2001). Some empirical relationships between nutrients and chlorophyll-a have been found among systems and, through time, within systems (Boynton et al. 1982); however, such relationships usually are only adequate to provide broad direction for management (i.e., increased nutrients are associated with higher chlorophyll-a). Nonetheless, it is important and useful to recognize that, on the basis of experimental evidence documenting nutrient limitation of

phytoplankton, we expect that increased nutrient inputs would increase phytoplankton production and biomass in Pensacola Bay.

3.3 Preventing Hypoxia

Hypoxia, which we define here as DO < 2 mg L^{-1}, is a common phenomenon in bottom waters of Pensacola Bay (Hagy and Murrell 2007). Where it occurs, hypoxia should be a concern because of its direct impact on marine life (Diaz and Rosenberg 1995). Globally, hypoxia has increased in coastal waters in some relation to nutrient overenrichment, commonly causing negative effects on the health and productivity of biological communities (Diaz 2001). Thus, hypoxia is a significant threat to both human (e.g., fisheries) and aquatic life use attainment in coastal systems.

Observations of hypoxia in the Pensacola Bay system date back to Hopkins (1969), who found oxygen concentrations as low as 0.48 mg L^{-1} in bottom waters of Escambia Bay during a late-summer study. Olinger et al. (1975) reported bottom water oxygen less than 1 mg L^{-1} in Escambia Bay but concluded that conditions had improved since 1969. The recent monthly survey data (Table 1) show that hypoxic bottom waters occurred between April and November, with the maximum extent in late summer (Figure 7). Hypoxia currently affects an average of 25% of the bay bottom during summer and can, at times, affect a much larger fraction (Figure 7).

Olinger et al. (1975) recognized long ago that the bay often was strongly stratified, and that weak circulation probably contributed to the risk of hypoxia. Hagy and Murrell (2007) quantified the stratification and circulation to better characterize conditions that create a high susceptibility to hypoxia. They found that hypoxia developed in Pensacola Bay despite moderate oxygen consumption rates in the lower water column and sediments. Total (i.e., combined) oxygen consumption for the lower water column and sediments was estimated to be 0 to 1.5 g oxygen m^{-2} day^{-1} using a box model (Hagy and Murrell 2007). The model estimates were in reasonable agreement with average integrated oxygen consumption rates (0.94 g oxygen m^{-2} day^{-1}) measured during summer at

several stations throughout the Pensacola Bay system (Table 4). Oxygen consumption in Pensacola Bay is low compared with other estuarine and coastal marine systems (e.g., Hopkinson 1985, Cowan et al. 1996) and is much lower than rates from a very eutrophic system such as the Chesapeake Bay, where integrated metabolism below the pycnocline can exceed 10 g oxygen m^{-2} day^{-1} (Kemp et al. 1997).

Table 4. Plankton community respiration rate (R_p) integrated from the pycnocline to sediments (i.e., lower water column) and sediment oxygen demand (SOD) at stations in the Pensacola Bay system (Figure 1) with varying levels of bottom water (BW) oxygen (mg L^{-1}). SOD was measured using 5-hour incubations of diver-collected cores. Plankton respiration rates were measured in 24-hour incubations in BOD bottles. All rates have units of g oxygen m^{-2} day^{-1}. Unpublished data provided by M. Murrell.

Station	BW O$_2$	SOD	R_p	Total
PCOLA28	5.2	0.32	0.34	0.66
PCOLA20	2.5	0.27	0.07	0.34
PCOLA29	0.7	0.04	0.25	0.29
BP04	4.2	0.37	0.86	1.23
PCOLA26	1.1	0.12	0.88	1.00
BP03	5.0	0.38	1.48	1.86
P05C (6/28/04)	1.6	0.48	0.58	1.06
P05C (7/27/04)	1.4	0.50	0.59	1.09
Average	—	0.31	0.63	0.94

Coastal systems can be naturally susceptible to hypoxia. A well-known example on the Gulf coast is the occurrence of "jubilee" events, in which marine animals (especially crabs) climb onto the beach to avoid low-oxygen waters. Jubilees have occurred in Mobile Bay, AL, for more than a century (May 1973, Schroeder and Wiseman 1988). There is presently no evidence that hypoxia did not occur in Pensacola Bay prior to significant human influence, nor are there adequate data to document any trend in dissolved oxygen (DO) during the past several decades. Nevertheless, because relatively extensive hypoxia occurs presently in Pensacola Bay, hypoxia should be a focus for water quality management. We can infer that hypoxia presently limits human and aquatic life uses of Pensacola Bay because the direct effects of hypoxia on many species of marine life are known (Diaz and Rosenburg 1995, Campbell and Goodman 2004). Although current science for Pensacola Bay cannot predict the magnitude of increase in extent or severity of hypoxia expected from a particular increase in nutrient inputs or the extent to which the severity of hypoxia might increase, both conceptual and quantitative models indicate that increased nutrients most likely would increase the extent and severity of hypoxia, resulting in loss of existing aquatic life uses.

Defining a broadly applicable approach for determining DO criteria in estuaries where hypoxia may occur naturally is a difficult proposition. The ecoregional approach that we applied for nutrients and chlorophyll-a may not be adequate for DO because EPA's Level III ecoregions do not reflect the physical oceanographic characteristics of estuaries that influence the development of hypoxia. In fact, none of the classification schemes that have been devised for estuaries have been shown to be effective for grouping systems according to their intrinsic vulnerability to hypoxia. Moreover, the data sets and statistical indicators (i.e., medians) that we used are not suitable to characterize the extent of hypoxia in most instances; the estuary median generally provides little information about the extent of bottom DO. For Pensacola Bay, where an average of 25% of the bottom is hypoxic during summer, the median concentration is, by definition, higher than our defined threshold (i.e., >2.0 mg L^{-1}). Relatively elaborate DO criteria have been developed for Chesapeake Bay based on a tiered aquatic life use approach and the oxygen levels known to be minimally supportive of those uses (U.S. EPA 2003). The complex approach utilized for Chesapeake Bay, however, may not be feasible for many of the nation's estuaries because the information required to delineate tiered-use areas is not available. For Pensacola Bay, it may be reasonable to define only two areas: (1) the area that is susceptible to hypoxia, which would have a criteria defined in terms of areal extent; and (2) an area that is much less susceptible to hypoxia (e.g., surface waters), for which criteria could be defined in terms of the requirements of resident

biota. Current areal extent of bottom water hypoxia is well defined and could provide the basis for a criterion defined as a multiyear average extent (e.g., 3 years). A reasonable implementation strategy must include an averaging period of several years because the extent of hypoxia varies strongly on an interannual basis (Figure 7). Freshwater inflow level, which varies from year to year, is an important driver of such changes (Hagy and Murrell 2007). For above-pycnocline waters and nonstratified waters, DO concentrations should be greater than 4.0 mg L^{-1}, a level sufficient to prevent most impacts on marine life in Pensacola Bay (assuming summer temperature ≈30 °C) and that is also consistent with current Florida criteria for Class III marine waters (FAC 2005). In the monthly survey data, DO concentrations in the bay exceeded this value almost all the time. The Florida statute also requires a minimum of 5.0 mg L^{-1} as a 24-hour average. This is most likely met much of the time as well, but, because most of the available data in the estuary are daytime point observations, good estimates of the 24-hour average are not available.

Design of compliance monitoring is an important consideration for DO because sampling must adequately quantify spatial extent for hypoxia in bottom waters and minimum concentration for other waters. The data sets for Pensacola Bay illustrate the potential for problems. Florida's IMAP measured bottom water oxygen at 29 stations in Pensacola Bay during an intensive probabilistic survey in 2003, observing oxygen < 2 mg L^{-1} at 6 stations (21%), which can be interpreted to mean that 21% of the bottom was hypoxic. The transect-based surveys obtained similar results. In 2004, however, IMAP visited just two sites and observed no hypoxia, even though hypoxia was much more extensive in Pensacola Bay in 2004 than in 2003 (Figure 7). Neither the transect surveys nor the probabilistic IMAP surveys sampled at night, when minimum concentrations may have occurred.

3.4 Protecting Seagrass Habitats

One of the most widespread and well-recognized consequences of nutrient enrichment in coastal systems is the loss of submerged rooted vascular plants (commonly referred to as submerged aquatic vegetation [SAV]), which are well adapted to low-nutrient, high-light environments (Short and Wyllie-Echevarria 1996). Excess nutrients have been cited specifically as a major cause of SAV loss in the northern Gulf of Mexico, including every major estuary in northwest Florida (USGS 2004). Other causes of SAV loss, such as prop scarring, disease, and even food web shifts also have been identified (Dawes et al. 2004).

Historically, the SAV bed extended along a large fraction of the shoreline of Pensacola Bay (Figure 8). Olinger et al. (1975) delineated the extent of SAV beds from aerial photographs taken as early as 1949. Aerial photography was conducted for the purpose of highway construction and was repeated at sporadic intervals, according to highway construction
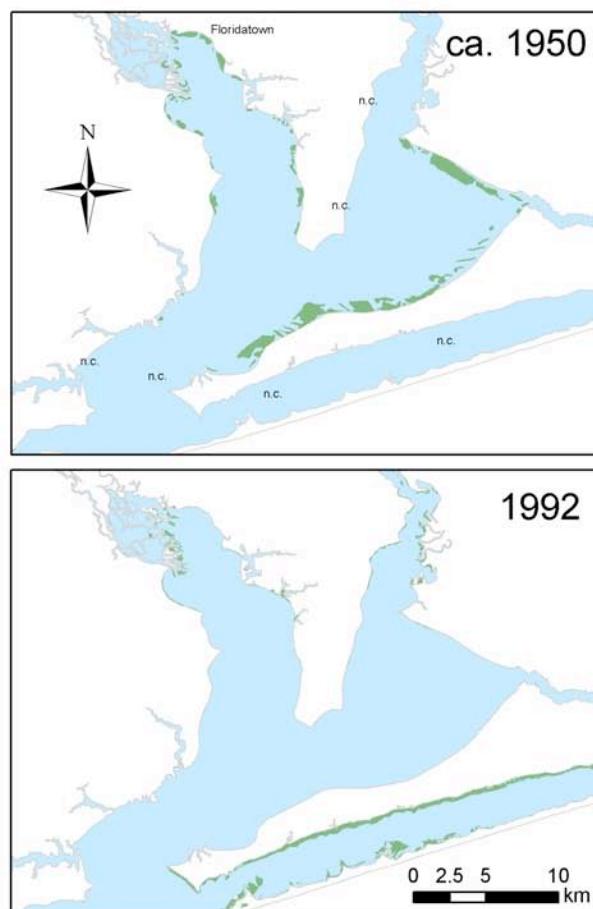

Figure 8. Coverage of submersed aquatic vegetation in Pensacola Bay circa 1950 and in 1992. Data for circa 1950 were digitized from coverage maps reported by Olinger et al. (1975). Areas for which no coverage data was available are indicated by "nd." SAV coverage for 1992 is from the USGS National Wetlands Research Center, Lafayette, LA.

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needs. The result is an irregular series of SAV coverage maps (Olinger et al. 1975, also reproduced by Lores et al. 2000). Although several areas of the estuary never were surveyed during this time period, SAV beds delineated within the available imagery covered 17.7 km^2 from 1949 to 1951 (Table 5). By overlaying the circa 1950 SAV coverage map on the bathymetry of the bay (based on 1993 surveys [Divins and Metzger, public communication]), one can estimate that SAV attained a maximum colonization depth of as much as 6 m in Pensacola Bay, 2 to 3 m in East Bay, and 1 to 2 m in the presently more turbid areas of northern Escambia Bay. Based on average light requirements for seagrasses (Duarte 1991), average secchi depth may have been nearly 5 m in portions of Pensacola Bay. Water clarity is currently much less than this (Figure 5), although several observations approaching this value were made during the monthly surveys.

Table 5. Area (km^2) of submerged aquatic vegetation (SAV) beds in the Pensacola Bay system. Data for 1949 through 1951 were computed from maps reported by Olinger et al. (1975). The computed area is a lower limit because some areas were not covered by the aerial photography, which was conducted for the purpose of highway construction. Omitted areas include Blackwater Bay, the western shore of East Bay, western Pensacola Bay, and Santa Rosa Sound. Data for 1960, 1980, 1992, and 2003 are from the USGS National Wetlands Research Center, Lafayette, LA.

Basin	1949-1951	1960	1980	1992	2003
Escambia Bay	4.82	1.05	0.24	1.78	0.45
East Bay	8.21	4.76	0.20	0.69	0.11
Pensacola Bay	4.70	3.71	0.55	1.14	1.51
TOTAL	17.73	9.52	1.00	3.62	2.07
Santa Rosa Sound	nd	25.00	14.47	11.17	12.27

nd = No data

SAV surveys in Escambia Bay show that significant SAV loss occurred within a few years of the earliest surveys (Olinger et al. 1975), and that SAV virtually was eliminated in much of the Pensacola Bay system by 1980 (Schwenning et al. 2007). Particularly rapid and complete SAV loss occurred in the Floridatown area of northeast Escambia Bay (Figure 8) following initiation of industrial point source discharges in the area. Approximately 50% of SAV was lost bay-wide by 1960, and all but a few percent by 1980 (Table 5). SAV coverage in nearby Santa Rosa Sound also declined about 50% between 1960 and 1980 and remained relatively constant thereafter (Table 5). The freshwater SAV species *Vallisneria americana* accounted for >80% of the 1992 coverage in Escambia Bay, mostly in the Escambia River delta, with limited beds of *Ruppia maritima* also present (Lores et al. 2000). These beds suffered massive mortality because of high salinity in 2000 (Lores and Sprecht 2001) and recovered only partially by 2003. Most of the SAV habitats remaining in the Pensacola Bay system are in Santa Rosa Sound. These beds are dominated by *Thalassia testudinum,* with *Halodule wrightii* also present. Although still significant in coverage, we have observed that the vegetation appears stunted and sparse compared to apparently healthier beds in northwest Florida (e.g., St. Joseph's Bay).

The decline in SAV coverage between 1950 and 1980 has been attributed to poor water quality resulting from industrial pollution and, to a very limited extent, to dredging for harbor construction (Olinger et al. 1975). The causal relationships between current water quality and the present distribution of SAV are not clear. The underwater light environment has been cited as the most important predictor of SAV survival. *Thalassia* requires 20% to 25% of surface irradiance at its maximum colonization depth (Duarte 1991, Dawes et al. 2004). Assuming a conservative 25%, the light field in Pensacola Bay is currently adequate to support *Thalassia* to a depth of approximately 2 m in the polyhaline reaches (where salinity is appropriate for *Thalassia*), consistent with its present maximum colonization depth.

SAV is absent from many places in Pensacola Bay where light appears to be adequate, an observation for which there are many possible explanations (Koch 2001). Metabolic stress associated with high sulfide in sediment pore waters has been implicated as a possible stressor in Pensacola Bay, where sulfide

concentrations as high as 5 mM have been measured (Devereux, unpublished data). Sulfide is especially harmful to plants when water temperature and salinity are high (Koch and Erskine 2001), as is common during summer in Gulf coast estuaries. Absence of *Thalassia* also may result from poor propagation. Flowering and seed production by *Thalassia* is reduced or absent in northwest Florida, which is at the northern limit of the more tropical range for that species (Dawes et al. 2004). In the absence of seed production, revegetation occurs slowly by rhizome extension, which has been observed in Pensacola Bay (Lores et al. 2000). *H. wrightii* appears to be within both temperature and salinity tolerances; the reasons for its absence are not clear.

Based on the global and regional trends relating nutrient enrichment and SAV loss (Short and Wyllie-Echevarria 1996), one may infer that SAV loss in Pensacola Bay is also a consequence of nutrient enrichment. Because SAV beds create important habitat for a variety of estuarine biota (Dawes et al. 2004), their protection and eventual restoration is critical to supporting aquatic life uses of Pensacola Bay. Thus, the water quality requirements for SAV growth and survival should be a significant consideration for determining nutrient criteria for Pensacola Bay. Although much is known about seagrass ecology, effective decision-support tools based on that knowledge are not readily available for many applications, including Pensacola Bay. SAV growth models (e.g., Eldridge et al. 2004) have the ability to integrate the effects of many factors to predict SAV growth. Such models have been implemented for many SAV species and have been integrated in some cases with fully coupled hydrodynamic and water quality models (Cerco and Moore 2001). Although promising, the results of these models cannot be transferred across systems without adaptation to local conditions, calibration, and validation. EPA is presently adapting a SAV model developed for *Thalassia* in Laguna Madre, TX (Eldridge et al. 2004), to growth conditions in Pensacola Bay, a promising first step. However, approaches to extrapolating from local SAV growth conditions to appropriate water column nutrient

concentrations for the bay have not been developed, and quantitative relationships between nutrients and SAV are unclear.

As in the case of hypoxia, some policy alternatives can be evaluated on the basis of our knowledge of SAV ecology in Pensacola Bay. First, consistent with Gulf-wide trends, SAV habitats in Pensacola Bay are degraded presently relative to their past condition. Second, the best available science links losses of SAV to nutrient enrichment both globally and in the Gulf of Mexico region. In the case of Pensacola Bay, it remains unclear to what extent, if any, current nutrient concentrations or water clarity are an impediment to SAV growth, coverage, or restoration success. The weight of evidence suggests that an increase in nutrients likely would pose a risk of further degradation of these important habitats and should be prevented through appropriate numeric criteria. Meanwhile, continued monitoring of SAV habitats should be pursued at a regular interval, along with advancements in predictive models relating water quality conditions to growth and survival of SAV species. An important need is relating SAV requirements in the littoral habitats where they occur to nutrient conditions at the larger scale to which nutrient criteria likely would apply (e.g., Cerco and Moore 2001).

3.5 A Watershed Perspective

An increase in nitrogen and phosphorus export from watersheds accompanies conversion of primarily forested lands to cropland and developed land uses (Reckhow et al. 1980, Jordan et al. 1997). Therefore, information on the impacts of human activity in a watershed provides important insight into the extent to which nutrient inputs to coastal waters may have increased and the potential for management actions to reduce nutrient enrichment and its adverse impacts.

As of 2001, the Pensacola Bay watershed included approximately 7% developed land and 6% cropland. The remaining land was forest (including silviculture), shrub land, or pasture and grassland (Figure 9). Nearly all of the developed land is in the immediate vicinity of Pensacola Bay, whereas the cropland is concentrated in the lower Escambia River

watershed (Figure 9). The relatively small extent of development in the fluvial watershed corresponds to the low human population density, which has remained nearly constant at about 10 km^{-2} from 1970 through 2000 (NOAA, public communication). Population for the watershed as a whole was 29 km^{-2} in 2000 (NOAA, public communication), very similar to the 2000 median for Gulf of Mexico estuaries (31 km^{-2}), which, overall, are populated much less densely than those in the Mid-Atlantic region (182 km^{-2}). As in many coastal areas, the population in the Pensacola Bay watershed is growing rapidly; population in the watershed increased 42% between 1970 and 2000 (NOAA, public communication). Developed land in Santa Rosa County, FL, one of two coastal counties in the watershed is projected to increase 350% by 2060 (Zwick and Carr, 2006).

Nutrient loading from the watershed remains relatively low, most likely because only a small fraction of the watershed has been developed or put into row-crop production. Cherry and Hagy (2006) estimated the input from rivers of dissolved inorganic nitrogen (DIN) and phosphate (PO_4^{3-}) into Pensacola Bay. Based on the ratio of average TN to average DIN for fresh water (TN/DIN = 2), the estimated average input of TN was 1.2×10^4 kg nitrogen day^{-1}. When scaled by the fluvial watershed area (17,069 km^2), the average TN yield delivered from the watershed to the bay was 262 kg nitrogen km^{-2} year^{-1}. Using the same approach for TP, the estimated average TP load and average watershed TP yield was 1.0×10^3 kg phosphorus day^{-1} and 21.8 kg phosphorus km^{-2} year^{-1}, respectively. Both nitrogen and phosphorus yields are consistent with reported values for entirely forested watersheds (Reckhow et al. 1980), indicating that current nutrient loadings from upland areas of the Pensacola Bay watershed are low. The SPARROW model (Smith et al. 1997), which predicts nutrient yields of watersheds at the national scale, provides a point of comparison for our estimates, with a few caveats. The SPARROW model estimate of average nitrogen and phosphorus yield for the hydrologic units comprising the Pensacola Bay watershed are 548 kg nitrogen km^{-2} year^{-1} and 53 kg phosphorus km^{-2} year^{-1}. Relative to the overall range of values from SPARROW for the conterminous United States, these values are very similar to our estimates for Pensacola Bay, even though they are approximately twofold higher for both nitrogen and
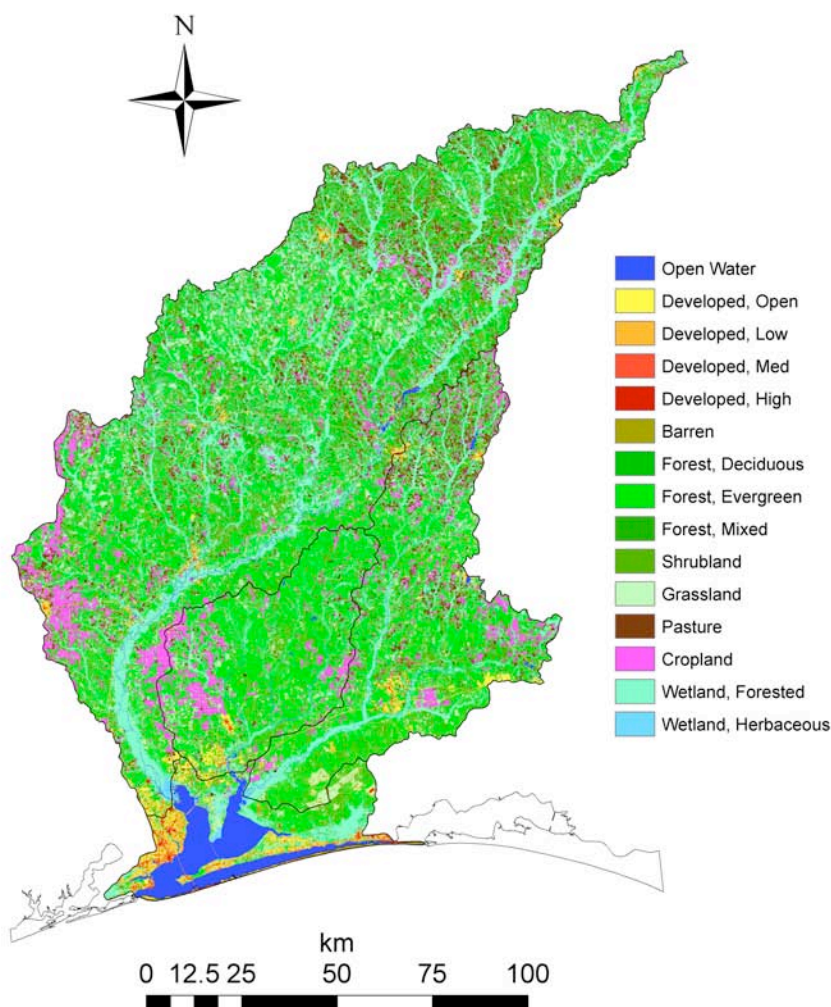

Figure 9. Land cover in the Pensacola Bay watershed (data from the 2001 National Land Cover Database).

phosphorus. As the developers of the SPARROW model note, the estimates are most reliable for comparing regional attributes of watersheds but are less reliable at smaller scales (Smith et al. 1997).

When nutrient export from the Pensacola Bay watershed is expressed per unit of estuary area, rather than per watershed area, the loadings appear somewhat higher. Scaled in this way, the TN and TP loading rates from the watershed to Pensacola Bay are 12 g nitrogen m^{-2} $year^{-1}$ and 1 g phosphorus m^{-2} $year^{-1}$, respectively, well within a moderate range of nutrient loading rates for estuaries (Boynton et al. 1995). The contrast between the relatively low yield rates and the moderate loadings per unit of estuarine area reflects the fact that the Pensacola Bay watershed is nearly 50 times the size of the estuary, approximately 2 times the median ratio for estuaries (NOAA 1999b).

Atmospheric nitrogen deposition is a possible source of anthropogenic nitrogen enrichment that could impact Pensacola Bay independent of land use change. Using local measurements, Cherry and Hagy (2006) estimated that wet nitrogen deposition in the vicinity of the bay was 0.3 g nitrogen m^{-2} $year^{-1}$. The 1999 to 2004 average for the watershed, computed from national maps (NADP 2005; data obtained electronically from source identified therein), was comparable at 0.36 g nitrogen m^{-2} $year^{-1}$. Assuming that dry nitrogen deposition is 55% of wet deposition (ratio based on local data; Hagy and Cherry 2006), atmospheric nitrogen deposition is approximately 0.5 g nitrogen m^{-2} $year^{-1}$ (= 500 kg nitrogen km^{-2} $year^{-1}$). As a direct input to the bay, this amounts to a small (4%) fraction of TN inputs. In contrast, it is a significant input to the watershed, about twofold greater than nitrogen export via rivers. Assuming that nitrogen exports via stream flow are 20% to 40% of the total of all nitrogen inputs to the landscape (Boyer et al. 2002), atmospheric deposition may account for two-thirds of that input. Given its likely quantitative significance, it would be helpful to know if atmospheric nitrogen deposition in the Pensacola Bay watershed has increased substantially over time. Several facts suggest that any increase was probably fairly

small. First, nitrogen concentrations in rainfall remain among the lowest for U.S. coastal watersheds (14 µM; NADP 2005). Only U.S. Pacific coast estuaries have significantly lower nitrogen concentrations in rainfall (NADP 2005). Because U.S. nitrogen oxide emissions in 1990 were 2.3-fold higher than in 1900 (U.S. EPA 1995), and the largest increases probably occurred where deposition presently is elevated, the proportional increase in the Pensacola Bay watershed was likely less.

The above observations regarding the Pensacola Bay watershed suggest that significant reductions in nutrient loading from the fluvial portion of the watershed may not be possible, even if slight improvements could be achieved by implementing best management practices on existing agricultural land, managed forests, developed land, and point sources. Because the landscape in the immediate vicinity of the bay is much more intensely developed and populated, loadings from that segment of the watershed may be higher and, therefore, potentially could be reduced through improved nutrient management. For the largest portion of the watershed, however, the risk of a significant increase in nutrient loading because of conversion of forest to urban and suburban development (Wickham et al. 2002) is much greater than is the potential for nutrient reductions.

3.6 Nutrient Criteria

Below we examine the information that we used and the logic that we followed to define an approach for deriving nutrient criteria for Pensacola Bay. We then apply the approach to derive hypothetical criteria. We also examine how the scientific information about the bay influenced the approach that was followed.

In our evaluation of Pensacola Bay water quality, we examined the environmental history of the bay, current attributes of the phytoplankton community, the prevalence of low DO (i.e., hypoxia), the status of seagrass (SAV) habitats, and the status of land use and nutrient exports from the watershed. We related these elements via a conceptual model of key ecosystem processes related to nutrients (Figure 2). We also evaluated water quality in Pensacola Bay in a comparative context by applying a modified

percentile approach in an ecoregional analysis of water quality (Figure 5, Table 2).

We evaluated these data by posing a series of questions for which the answers, as well as the uncertainty associated with them, influenced the logical progression. It can be informative to consider the questions (below) separately, but they also may be considered part of a logic path (Figure 10).
• Has water quality declined relative to a documented historical condition?
• Is the estuary degraded in comparison to similar estuaries (i.e., estuaries in the same class?)
• Are there environmental measures or indicators of conditions widely associated with nutrient overenrichment?
• Are nutrient loads significantly enriched because of anthropogenic causes?

At the end of the process, we determined that a nutrient criteria for Pensacola Bay reasonably could be defined by the current water quality conditions (Figure 10). Below, we explain the process, as outlined in Figure 10, in greater detail.

Our analysis began with the historical data. We concluded that there were no data to indicate clearly that water quality in Pensacola Bay had declined because of nutrients (box A in Figure 10). This effectively eliminated the possibility of deriving criteria based, in large part, on a historical reference condition approach (box F). Had the available historical data shown a substantial decline in water quality, we may have considered a greater role for historical data in deriving nutrient criteria (box F). Even if this had been the case, additional supporting information still would be useful to further evaluate and support the values.

Comparison of water quality in Pensacola Bay with other estuaries (box B in Figure 10) in its ecoregion-based class (Table 2, Figure 5) showed that water quality was often among the best in the class, and that it was not degraded relative to the class. Because current values for many water quality indicators were better than the comparison values obtained from the percentile approach, it did not seem reasonable to use these values as criteria. If, on the other hand, the comparison values determined via the percentile approach were better than current values, one might reasonably begin with the comparison values as criteria, then provide additional supporting evidence as possible (box G).

Despite finding comparatively good water quality in terms of nutrient concentrations, water clarity, and chlorophyll-a, Pensacola Bay has both significant bottom water hypoxia and extensive loss of seagrass habitat, conditions widely associated with nutrient enrichment in estuaries (box C in Figure 10). Because hypoxia and seagrass loss have well-known negative impacts on estuarine systems, we expect that
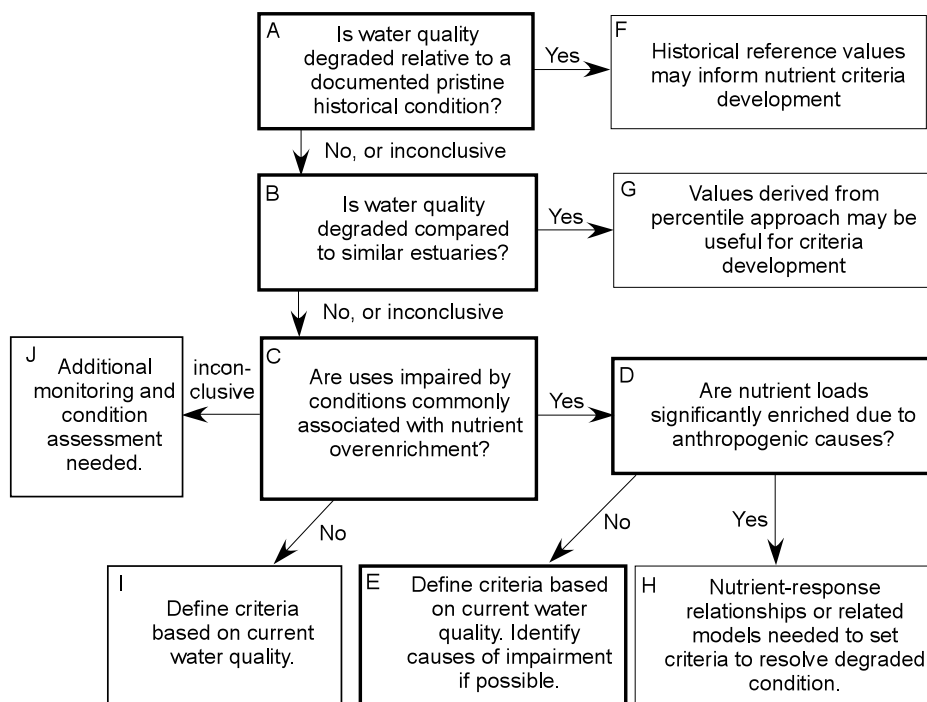

Figure 10. Flow diagram illustrating the logical path followed to determine an appropriate approach for determining nutrient criteria for Pensacola Bay (boxes A through E) and a subset of possible alternative paths (boxes F through J) that may have been followed had the scientific findings been different.

human and aquatic life uses are limited by these conditions. Moreover, seagrass loss and hypoxia often increase with anthropogenic nutrient inputs. The juxtaposition of seagrass loss and hypoxia with otherwise good water quality presents a relatively complex picture for nutrient criteria in Pensacola Bay. It appears that much of the loss occurred during the 1950s and 1960s in association with industrial nutrient and organic inputs that were eliminated by the early 1970s. Thus, these causes are not present today. The factors that impede recovery, as well as the prospects and best approach for successful restoration could be understood better through additional research. The significant extent of hypoxia in the bay appears to reflect a natural susceptibility of the bay to hypoxia. It is not clear that realistic reductions in nutrients could eliminate hypoxia. Scientific uncertainty surrounding this conclusion also could be reduced through additional research, particularly improved modeling of the coupled physical-chemical dynamics of the ecosystem. Both SAV loss and hypoxia commonly are associated in estuarine and coastal waters with significant phytoplankton blooms, sometimes including harmful algal species. However, phytoplankton production and biomass in Pensacola Bay are relatively modest compared to other systems. High-biomass phytoplankton blooms occur infrequently; harmful algal blooms are not known to develop in Pensacola Bay but, on occasion, have been transported into the bay from the Gulf of Mexico (Tester and Steidinger, 1997). It appears that Pensacola Bay is prone to adverse impacts resulting from nutrient enrichment and already has been impacted in the past by nutrients, but it may not be demonstrating clear symptoms of nutrient enrichment at present.

To better evaluate this hypothesis, we examined nutrient inputs to the bay (box D in Figure 10). EPA did not include evaluating nutrient sources as part of a recommended criteria development process for freshwater systems (U.S. EPA 2000a,b). Rather, this step usually is undertaken as part of the total daily maximum load process. However, this analysis was a useful diagnostic step for Pensacola Bay and could prove to be important for criteria

development for other estuaries. Although there were no historical baseline data with which current loading data could be compared, our analysis of current nutrient inputs from rivers, land use in the watershed, and atmospheric nitrogen deposition suggested that it was unlikely that nutrient inputs have increased dramatically. Evaluating water quality in the major rivers entering the bay showed that nutrient export from the watershed was similar to pristine or nearly pristine watersheds. This observation was supported by the absence in the watershed of major causes of increased nutrient export (e.g., low population, minimal row-crop agriculture, low atmospheric nitrogen deposition). One could conclude that reasonable management actions probably could not effect significant reductions in nutrient inputs to the bay via the major tributary rivers. Further research could reduce the scientific uncertainty regarding nutrient loads from the more developed portions of the watershed immediately surrounding the bay, which remain poorly quantified.

Ultimately, our approach to developing criteria (box E in Figure 10) was the same as if no impaired conditions were identified (box I). In each situation, we suggest that it is scientifically justified to define water quality criteria as the current water quality conditions because there is little justification for alternative, more stringent criteria. Table 6 outlines hypothetical criteria that are based on summer medians, by salinity zone, for the 1996 to 2004 timeframe. We computed criteria for chlorophyll-a, secchi depth, DIN, PO_4^{3-}, TN, TP, and DO. For concentration-based water quality indicators (e.g., chlorophyll-a), the criteria include a 10% buffer and a suitable averaging period (e.g., 3 years), such that small climate-driven variations surrounding current water quality would not trigger a determination of impairment. Because we concluded that a portion of the bay may be subject to hypoxia even in the absence of anthropogenic nutrient enrichment, DO criteria for bottom water must accommodate this feature of the ecosystem in some way. Our evaluation of oxygen dynamics in the bay indicates that a sensible approach would define hypoxia and limit the acceptable extent in

Table 6. Suggested nutrient and nutrient-related water quality criteria for Pensacola Bay resulting from the weigh-of-evidence approach described herein. All values are 3-year moving averages of summer medians. The quantities are based on observed values (Table 2) that have been adjusted to provide 10% allowance (i.e., 10% higher for concentrations, 10% lower for secchi depth).

Parameter	Criteria Recommendation
Chlorophyll-a	Summer median concentration <8 µg L^{-1} (all salinity zones)
Secchi Depth	Summer median >0.8 m (salinity 0-5), >1.1 m (salinity 5-18), >1.8 m (salinity >18)
DO	Bottom waters below a pycnocline must have oxygen >2.0 mg L^{-1} except for in an area that may average up to 25% of the mean low water surface area of the bay. Surface waters or unstratified waters: oxygen >4.0 mg L^{-1} at all times. 24-hour average oxygen >5.0 mg L^{-1}
DIN	Summer median <10 µM (salinity <5), <1.0 µM (salinity 5-18), <0.5 µM (salinity >18)
PO_4^{3-}	Summer median <0.2 µM (salinity <5), <0.12 µM (salinity 5-18), <0.05 µM (salinity >18)
TN	Summer median <35 µM (all salinity zones)
TP	Summer median <1 µM (salinity 0-5), <0.6 µM (salinity 5-18), <0.5 µM (salinity >18)

relative to some limited historical data, that values also are degraded relative to ecoregional reference sites, that commonly observed nutrient-related problems are present, and that nutrient inputs are elevated because of obvious anthropogenic causes. Here, a coherent picture emerges and would be supported by several lines of evidence, each of which would provide support and insight and final criteria determination. In an estuary where reasonable criteria would stipulate reduced nutrients and improved water quality, some additional elements may be helpful or even necessary (box H). For example, models describing relationships among water quality variables may be helpful to ensure that criteria for causal variables (i.e., nutrients) are reasonably consistent with expected values for response variables (DO and chlorophyll-a). Criteria other than those proposed here, including biocriteria, could be useful where the indicators that we used fail to capture the ecological impacts resulting from nutrient overenrichment. For example, a harmful proliferation of macroalgae occurs in some estuaries subject to nutrient enrichment and is an important measure of the overall response to nutrients. Development of additional case studies would be a useful way to demonstrate approaches for deriving criteria in situations that were not encountered in the case of Pensacola Bay. Such case studies, like this one, could provide a template that could be adapted as desired as States develop criteria for their estuaries.

near-bottom waters (e.g., 0.5 m above the bottom). An approach that defines either a $DIN = NO_2^- + NO_3^- + NH_4^+$ concentration minimum for all bottom waters of the bay or even a lower limit for the median concentration in the bay provides less precise regulation of the extent of hypoxia. Scientifically defensible definitions of hypoxia for Pensacola Bay could arguably vary from 2 to as high as 4 mg L^{-1}, provided the limits of extent are defined accordingly. In our example, we defined hypoxia as <2.0 mg L^{-1}, the level at which relatively hypoxia-tolerant species begin to exhibit significant mortality (Diaz and Rosenberg 1995). The corresponding limit for average extent of hypoxia (over several years) is approximately 25% of the surface area of the bay, or 90 km^2 (Table 6). Concerns related to natural extent of hypoxia do not apply for surface waters, where Florida's existing statute is both consistent with existing water quality conditions and with expectations of aquatic life uses (Table 6).

Given the complex ecology and ecological historical of Pensacola Bay, it would not be surprising if other estuaries were similarly complex. However, we expect that criteria development for many estuaries could follow a more typical model. For example, one might find that water quality in an estuary is degraded

4. Summary and Conclusions

We developed and applied an approach for developing numerical water quality criteria for nutrients and nutrient-related water quality indicators in an estuary. The approach is best defined as a weight-of-evidence approach whose important elements include those below.

- Developing a conceptual model to organize and present the important ecological science and other information related to nutrients in the system.
- Comparing water quality and other ecosystem attributes in the system with values from other estuarine ecosystems in a reasonable comparison group.
- Evaluating the incidence of actual or potential nutrient-related water quality impacts in the ecosystem. The most common examples include hypoxia, SAV loss, and high-biomass or harmful phytoplankton or macroalgal blooms.
- Evaluating the extent to which impacts have been caused by anthropogenic nutrient enrichment using historical observations, analysis of ecosystem processes, ecological models, or other approaches.
- Evaluating the extent to which nutrient inputs to the system are likely to have been increased by human activities.

An additional desirable element of an analysis supporting criteria development is an assessment of the risks for future environmental changes, including the principal drivers of such changes. Such an assessment would help inform future programmatic reassessments of water quality criteria. Our analysis of the Pensacola Bay ecosystem indicates that increased human development associated with current population trends poses a significant risk for increased nutrient inputs. The most sensitive and easily monitored indicators of enrichment are likely to be nutrient inputs themselves (especially in freshwaters), phytoplankton biomass, and extent of hypoxia. SAV extent is presently too degraded; is too sensitive to other impacts, particularly salinity; and appears too slow to respond to improved conditions to serve as a reliable indicator of enrichment for most of Pensacola Bay.

We believe that the weight-of-evidence approach we outlined here and applied to Pensacola Bay would be applicable for many estuarine systems. The approach is most applicable where enrichment impacts are relatively modest and where a suitable comparison group provides an adequate context that might be used to justify criteria requiring some reduction in nutrients. The approach may be a useful point of departure for systems that have been impacted seriously by nutrients. However, in these cases, additional tools likely will be needed to support criteria development. Better methods would be needed to determine what level of aquatic life uses are achievable (including what uses were supported historically) and to identify the water quality requirements for those uses and, particularly, the relationships among different water quality indicators (e.g., nutrients, chlorophyll-a, hypoxia).

5. References

Academy of Natural Sciences of Philadelphia. 1953. Escambia River, Florida, vicinity of the Chemstrand Corporation Plant, Fall 1952 - Spring 1953. Stream survey report for the E. I. du Pont de Nemours and Company. Department of Limnology, Academy of Natural Sciences of Philadelphia, Philadelphia, PA. December 1953. 165 pp.

Boyer, E. W., C. L. Goodale, N. A. Jaworski, and R. W. Howarth. 2002. Anthropogenic nitrogen sources and relationships to riverine export in the northeastern U.S.A. *Biogeochemistry* **57/58**:137-169.

Boynton, W. R. and W. M. Kemp. 2007. Estuaries. *In* Capone, D. G., D. A. Bronk, M. R. Mulholland, and E. J. Carpenter (eds.). Nitrogen in the Marine Environment. Second Edition. Academic Press, New York. In press.

Boynton, W. R., W. M. Kemp, and C. W. Keefe. 1982. A comparative analysis of nutrients and other factors influencing estuarine phytoplankton production. pp. 69-90. *In* V. S. Kennedy, editor. Estuarine Comparisons. Academic Press, New York.

Boynton, W. R., J. H. Garber, R. Summers, and W. M. Kemp. 1995. Inputs, transformations, and transport of nitrogen and phosphorus in Chesapeake Bay and selected tributaries. *Estuaries* **18**(1B):285-314.

Campbell, J. G. and L. R. Goodman. 2004. Acute sensitivity of juvenile shortnose sturgeon to low dissolved oxygen concentrations. *Transactions of the American Fisheries Society* **133**:772-776.

Cerco, C. F. and K. Moore. 2001. System-wide submerged aquatic vegetation model for Chesapeake Bay. *Estuaries* **24**(4):522-534.

Cherry, J. S. II and J. D. Hagy III. 2006. Pensacola Bay, Florida USA. Land-Ocean Interactions in the Coastal Zone (LOICZ) Biogeochemical Budgets. Electronically published by the Land-Ocean Interactions in the Coastal Zone (LOICZ) programme. (http://www.loicz.org).

Cloern, J. E. 2001. Our evolving conceptual model of the coastal eutrophication problem. *Marine Ecology Progress Series* **210**:223-253.

Cowan, J. L., J. R. Pennock, and W. R. Boynton. 1996. Seasonal interannual patterns of sediment-water nutrient and oxygen fluxes in Mobile Bay, Alabama (USA): regulating factors and ecological significant. *Marine Ecology Progress Series* **141**:229-245.

Dame, R., M. Alber, D. Allen, M. Mallin, C. Montague, A. Lewitus, A. Chalmers, R. Gardner, C. Gilman, B. Kjerfve, J. Pinckney, and N. Smith. 2000. Estuaries of the south Atlantic coast of North America: Their geographical signatures. *Estuaries* **23**(6):793-819.

Dawes, C. J., R. C. Phillips, and G. Morrison. 2004. Seagrass Communities of the Gulf Coast of Florida: Status and Ecology. Florida Fish and Wildlife Conservation Commission, Fish and Wildlife Research Institute and the Tampa Bay Estuary Program. St. Petersburg, FL. 74 pp.

Devereax, R., personal communication. U.S. Environmental Protection Agency, National Health and Environmental Effects Research Laboratory, Gulf Breeze, FL. December 2006.

Diaz, R. J. 2001. Overview of hypoxia around the world. *Journal of Environmental Quality* **30**(2):275-281.

Diaz, R. J. and R. Rosenberg. 1995. Marine benthic hypoxia: A review of its ecological effects and the behavioral responses of benthic macrofauna. *Oceanography and Marine Biology Annual Review* **33**:245-303.

Divins, D. L. and D. Metzger, public communication. NGDC Coastal Relief Model. National Geophysical Data Center, National Oceanic and Atmospheric Administration, Boulder, CO. Accessed November 2006 (http://www.ngdc.noaa.gov/mgg/coastal/coastal.html).

Duarte, C. M. 1991. Seagrass depth limits. *Aquatic Botany* **40**:363-377.

Eldridge, P. M., J. E. Kaldy, and A. B. Burd. 2004. Stress response model for the tropical seagrass Thalassia testudinum: The interactions of light, temperature, sedimentation and geochemistry. *Estuaries* **27**(6):923-937.

FAC 2005. Section 62-302.530(31), Florida Administrative Code. Table: Criteria for Surface Water Quality Classifications. Effective July 27, 2005. Accessed at U.S. EPA Internet site (www.epa.gov/waterscience/standards/wqslibrary).

Fitzpatrick, J. J. and M. B. Meyers. 2000. A critical look at criteria. A review of methodologies that determine nutrient criteria for coastal and estuarine systems. *Water Environment and Technology* **12**(2):34-39.

GESAMP (IMO/FAO/UNESCO-IOC/WMO/WHO/IAEA/UN/UNEP Joint Group of Experts on the Scientific Aspects of Marine Environmental Protection) and Advisory Committee on Protection of the Sea. 2001. Protecting the oceans from land-based activities - Land-based sources and activities affecting the quality and uses of the marine, coastal and associated freshwater environment. GESAMP Reports and Studies Number 71. 162 pp.

Griffith, G. E., J. M. Omernik, C. W. Rohm, and S.M. Pierson. 1994. Florida regionalization project. U.S. Environmental Protection Agency, National Health and Environmental Effects Research Laboratory, Western Ecology Division, Corvallis, OR. EPA/600/Q-95/002. 83 pp.

Hagy, J. D., III and M. C. Murrell. 2007. Susceptibility of a northern Gulf of Mexico estuary to hypoxia: an analysis using box models. *Estuarine, Coastal and Shelf Science* **74**:239-253.

Homer, C., C. Huang, L. Yang, B. Wylie, and M. Coan. 2004. Development of a 2001 national land cover database for the United States. *Photogrammetric Engineering and Remote Sensing* **70**(7):829-840.

Hopkins, T. S. 1969. The Escambia River and Escambia Bay during summer 1969. University of West Florida, Pensacola, FL (unpublished manuscript). 54 pp.

Hopkinson, C. S., Jr. 1985. Shallow-water benthic and pelagic metabolism: evidence of heterotrophy in the nearshore Georgia Bight. *Marine Biology* **87**:19-32.

Jordan, T. E., D. L. Correll, and D. E. Weller. 1997. Relating nutrient discharges from watersheds to land use and streamflow variability. *Water Resources Research* **33**(11):2579-2590.

Juhl, A. R. and M. C. Murrell. 2005. Interactions between nutrients, phytoplankton growth, and microzooplankton grazing in a Gulf of Mexico estuary. *Aquatic Microbial Ecology* **38**:147-156

Karouna-Renier, N. K., R. A. Snyder, J. G. Allison, M. G. Wagner, and K. R. Rao. 2007. Accumulation of organic and inorganic contaminants in shellfish collected in estuarine waters near Pensacola, Florida: Contamination profiles and risks to human consumers. *Environmental Pollution* **145**:474-488.

Kemp, W. M., E. M. Smith, M. Marvin-DiPasquale, and W.R. Boynton. 1997. Organic carbon balance and net ecosystem metabolism in Chesapeake Bay. *Marine Ecology Progress Series* **150**:229-248.

Kennish, M. J. 1986. Ecology of Estuaries, Vol. I: Physical and Chemical Aspects. CRC Press, Boca Raton, FL. 254 pp.

Koch, E. W. 2001. Beyond light: physical, geological, and geochemical parameters as possible submersed aquatic vegetation habitat requirements. *Estuaries* **24**(1):1-17.

Koch, M. S and J. M. Erskine. 2001. Sulfide as a phytotoxin to the tropical seagrass Thalassia testudinum: interactions with light, salinity and temperature. *Journal of Experimental Marine Biology and Ecology* **266**:81-95.

Kurtz, J. C., N. D. Detenbeck, V. D. Engle, K. Ho, L. M. Smith, S. J. Jordan, and D. Campbell. 2006. Classifying coastal waters: Current necessity and historical perspective. *Estuaries and Coasts* **29**(1):107-123.

Lores, E. M. and D. T. Sprecht. 2001. Drought-induced decline of submerged aquatic vegetation in Escambia Bay, Florida. *Gulf of Mexico Science* **19**(2):161-164.

Lores, E. M., E. Pasko, J. Patrick, R. L. Quarles, J. Campbell, and J. Macauley. 2000. Mapping and monitoring of submerged aquatic vegetation in Escambia - Pensacola Bay System, Florida. *Gulf of Mexico Science* **18**:1-14.

MacIntyre, H. L., T. M. Kana, T. Anning, and R. J. Geider. 2002. Photoacclimation of photosynthesis irradiance response curves and photosynthetic pigments in microalgae and cyanobacteria. *Journal of Phycology* **38**:17-38.

May, E. B. 1973. Extensive oxygen depletion in Mobile Bay, Alabama. *Limnology and Oceanography* **18**(3):353-366.

McRae, G., M. Julian, and H. Brown. 2004. Florida's Inshore Marine Monitoring and Assessment Program (IMAP). Annual Report, Year Five. Florida Fish and Wildlife Conservation Commission, Fish and Wildlife Research Institute, St. Petersburg, FL. 97 pp.

Murrell, M., personal communication. U.S. Environmental Protection Agency, NHEERL/Gulf Ecology Division. 1 Sabine Island Drive, Gulf Breeze, FL 32563 (data provided November 2006).

Murrell, M. C. and J. M. Caffrey. 2005. High cyanobacterial abundance in three northeastern Gulf of Mexico estuaries. *Gulf and Caribbean Research* **17**:95-106.

Murrell, M.C. and E. M. Lores. 2004. Phytoplankton and zooplankton seasonal dynamics in a subtropical estuary: importance of cyanobacteria. *Journal of Plankton Research* **26**(3):371-382.

Murrell, M. C., R. S. Stanley, E. M. Lores, G. T. DiDonato, L. M. Smith, and D. A. Flemer. 2002. Evidence that phosphorus limits phytoplankton growth in a Gulf of Mexico estuary: Pensacola Bay, Florida, USA. *Bulletin of Marine Science* **70**(1):155-167.

Murrell, M. C., E. M. Lores, J. D. Hagy III, R. M. Greene. 2007. Phytoplankton production in relation to nutrient distributions in a sub-tropical estuary: importance of freshwater flow. *Estuaries and Coasts* **30**(3):390-402.

NADP. 2005. National Atmospheric Deposition Program. 2004. Annual Summary. NADP Data Report, 2005-01. Illinois State Water Survey, Champaign, IL. 16 pp.

Nixon, S. W. 1995. Coastal marine eutrophication: A definition, social causes, and future concerns. *Ophelia* **41**:199-219.

NOAA. 1997. NOAA's Estuarine Eutrophication Survey, Volume 4: Gulf of Mexico Region. National Oceanic and Atmospheric Administration, Office of Ocean Resources Conservation and Assessment, Silver Spring, MD. 77 pp.

NOAA. 1998. NOAA's Estuarine Eutrophication Survey, Volume 5: Pacific Coast Region. National Oceanic and Atmospheric Administration, Office of Ocean Resources Conservation and Assessment, Silver Spring, MD. 75 pp.

NOAA 1999a. Coastal Assessment Framework. Special Projects Office, National Ocean Service, National Oceanic and Atmospheric Administration, Silver Spring, MD.

NOAA 1999b. Physical and Hydrologic Characteristics of Coastal Watersheds. Coastal Assessment and Data Synthesis System, 1999. National Coastal Assessments Branch, Special Projects Office, National Ocean Service, National Oceanic and Atmospheric Administration, Silver Spring, MD. 6 pp.

NOAA, public communication. Spatial Trends in Coastal Socioeconomics, Coastal Ocean and Resource Economics Program, National Ocean Service, National Oceanic and Atmospheric Administration, Silver Spring, MD. Accessed February 2007.

NRC. 2000. Clean Coastal Waters: Understanding and Reducing the Effects of Nutrient Pollution. Ocean Studies Board and Water Science and Technology Board, Commission on Geosciences, Environment, and Resources, National Research Council, Washington, DC.

Olinger, L. W., R. G. Rogers, P. L. Fore, R. L. Todd, B. L. Mullins, F. T. Bisterfield, and L. A. Wise. 1975. Environmental and recovery studies of Escambia Bay and the Pensacola Bay system, Florida. U.S. Environmental Protection Agency, Atlanta, GA. 904/9-76-016. 468 pp.

Omernik, J. M. 1987. Ecoregions of the conterminous United States. Map (scale 1:7,500,000). *Annals of the Association of American Geographers* **77**(1):118-125.

Omernik, J. M. 1995. Ecoregions: A spatial framework for environmental management. pp. 49-62. *In* Davis, W. S. and T. P. Simon (eds.). Biological Assessment and Criteria: Tools for Water Resource Planning and Decision Making. Lewis Publishers, Boca Raton, FL.

Reckhow, K. H., M. N. Beaulac, and J. T. Simpson. 1980. Modeling phosphorus loading and lake response under uncertainty: a manual and compilation of export coefficients. U.S. Environmental Protection Agency, Washington, DC. EPA/440/5-80/011.

Schroeder, W. W. and W. J. Wiseman, Jr. 1988. The Mobile Bay estuary: Stratification, oxygen depletion and jubilees. *In* Kjerfve, B. (ed.) Hydrodynamics of Estuaries, Volume II, Estuarine Case Studies. CRC Press, Boca Raton, FL. pp. 41-529.

Schroeder, W.W. and W. J. Wiseman, Jr. 1999. Geology and hydrodynamics of Gulf of Mexico estuaries. pp. 3-28 *In* Bianchi, T. S., J. R. Pennock,, and R. R. Twilley (eds.) Biogeochemistry of Gulf of Mexico Estuaries. John Wiley and Sons, Inc., New York.

Schwenning, L., T. Bruce, and L. W. Handley. 2007. Pensacola Bay. pp. 129-142 *in* Handley, L., D. Altsman, and R. DeMay (eds.). Seagrass Status and Trends in the Northern Gulf of Mexico: 1940-2002. U.S. Geological Survey Scientific Investigations Report 2006-5287 and U.S. Environmental Protection Agency. 855-R-04-003. 267 pp.

Short, F. T. and S. Wyllie-Echevarria. 1996. Natural and human-induced disturbance of seagrasses. *Environmental Conservation* **23**(1):17-27.

Smith, R. A., G. E. Schwarz, and R. B. Alexander. 1997. Regional interpretation of water-quality monitoring data. *Water Resources Research* **33**(12):2781-2798.

Stephan, C. E., D. I. Mount, D. J. Hansen, J. H. Gentile, G. A. Chapman, and W. A. Brungs. 1985. Guidelines for deriving numerical national water quality criteria for the protection of aquatic organisms and their uses. National Technical Information Service, Springfield, VA. PB85-227049.

Summers, J. K., J. F. Paul, and A. Robertson. 1995. Monitoring the ecological condition of estuaries of the United States. *Toxicological and Environmental Chemistry* **49**:93-108.

Tester, P. A. and K. A. Steidinger. 1997. *Gymnodinium breve* red tide blooms: initiation, transport, and consequences of surface circulation. *Limnology and Oceanography* **42**(5):1039-1051.

U.S. EPA. 1995. National air pollutant emission trends, 1900-1994. U.S. Environmental Protection Agency, Washington, DC. EPA-454/R-95-011. 219 pp.

U.S. EPA. 1998. National strategy for the development of regional nutrient criteria. U.S. Environmental Protection Agency, Washington, DC. EPA 922-R-98-002. 47 pp.

U.S. EPA. 1999a. Protocol for developing nutrient TMDLs. First Edition. Office of Water (4503F), United States Environmental Protection Agency, Washington, DC. EPA 841-B-99-007. 135 pp.

U.S. EPA. 1999b. Ecological Condition of Estuaries in the Gulf of Mexico. U.S. Environmental Protection Agency, Office of Research and Development, National Health and Environmental Effects Research Laboratory, Gulf Ecology Division, Gulf Breeze, FL. EPA 620-R-98-004.

U.S. EPA. 2000a. Nutrient criteria technical guidance manual: Lakes and reservoirs. U.S. Environmental Protection Agency, Washington, DC. EPA-822-B00-001. 232 pp.

U.S. EPA. 2000b. Nutrient criteria technical guidance manual: River and Streams. U.S. Environmental Protection Agency, Washington, DC. EPA-822-B00-002. 254 pp.

U.S. EPA. 2000c. Ambient water quality criteria recommendations. Information supporting the development of state and tribal nutrient criteria. Lakes and reservoirs in nutrient ecoregion XII. U.S. Environmental Protection Agency, Washington, DC. EPA 822-B-00-013. 77 pp.

U.S. EPA. 2000d. Ambient water quality criteria recommendations. Information supporting the development of state and tribal nutrient criteria. Rivers and streams in nutrient ecoregion XII. U.S. Environmental Protection Agency, Washington, DC. EPA 822-B-00-021. 80 pp.

U.S. EPA. 2001. Nutrient Criteria Technical Guidance Manual. Estuarine and Coastal Marine Waters. U.S. Environmental Protection Agency, Washington, DC. EPA-822-B-01-003.

U.S. EPA. 2003. Ambient Water Quality Criteria for Dissolved Oxygen, Water Clarity, and Chlorophyll a for Chesapeake Bay and its Tidal Tributaries. U.S. Environmental Protection Agency, Region III Chesapeake Bay Program Office, Office of Water, Washington, DC. EPA 903-R-03-002.

U.S. EPA. 2004. National Coastal Condition Report II. U.S. Environmental Protection Agency, Office of Research and Development, Office of Water, Washington, DC. EPA-620/R-03/002.

U.S. EPA. 2005. The Ecological Condition of the Pensacola Bay System, Northwest Florida (1994-2001). U.S. Environmental Protection Agency, Office of Research and Development, National Health and Environmental Effects Research Laboratory, Gulf Ecology Division, Gulf Breeze, FL. EPA/620/R-05/002.

U.S. EPA/OWOW, public communication. U.S. Environmental Protection Agency, Office of Wetlands, Oceans, and Watersheds, Total Maximum Daily Load Program Web site. Accessed December 2006 (http://www.epa.gov/owow/tmdl).

USGS. 2004. Seagrass habitat in the northern Gulf of Mexico: Degradation, conservation and restoration of a valuable resource. United States Geological Survey, Gulf of Mexico Program Office, Stennis Space Center, MS. 855-R-04-001. 26 pp.

U.S. House of Representatives. 2000. United States Code. 2000 Edition. Office of the Law Revision Counsel of the United States House of Representatives, U.S. Government Printing Office, Washington, DC. Title 33, beginning with section 1251 accessed at http://www.gpoaccess.gov/uscode.

Wickham, J. D., R. B. O'Neill, K. H.Ritters, E. R. Smith, T. G. Wade, and K. B. Jones. 2002. Geographic targeting of increases in nutrient export due to future urbanization. *Ecological Applications* **12**(1):93-106.

Zwick, P. D. and M. H. Carr. 2006. Florida 2060, A population distribution scenario for the State of Florida. Geoplan Center, University of Florida, Gainesville, FL. 29 pp.

Appendix A

Observations in EPA's Level III Ecoregion 75

Estuaries included in EPA's Level III ecoregion 75, defined as the "Southeast Coastal Plain," and the number of surface water quality observations used to compute median water quality values for surface water. The estuaries are located between West Mississippi Sound, MS, and Winyah Bay, SC.

Estuarine Drainage Area	Number of Observations
Altamaha River, GA	8
Apalachee Bay, FL	19
Apalachicola Bay, FL	58
Breton/Chandeleur Sound, MS	4
Broad River, SC	27
Charleston Harbor, SC	14
Charlotte Harbor, FL	48
Choctawhatchee Bay, FL	37
East Mississippi Sound, MS/AL	32
Indian River, FL	139
Mobile Bay, AL	52
North/South Santee rivers, SC	3
Ossabaw Sound, GA	13
Pensacola Bay, FL	70
Perdido Bay, FL	18
Sarasota Bay, FL	58
Savannah River, SC	14
St. Andrew Bay, FL	62
St. Andrew/St. Simons sounds, GA	21
St. Catherines/Sapelo sounds, GA	25
St. Helena Sound, SC	28
St. Johns River, FL	65
St. Marys River/Cumberland Sound, FL/GA	16
Stono/North Edisto rivers, SC	13
Suwannee River, FL	57
Tampa Bay, FL	74
West Mississippi Sound, MS	61
Winyah Bay, SC	7
TOTAL	1,043

Appendix B

Observations in Florida's Level IV Ecoregion 75a

Estuaries included in Florida's Level IV ecoregion 75a, defined as the "Gulf Coast Flatwoods Region," and the number of surface water quality observations used to compute median water quality values for surface water. The estuaries are located in the northern Florida Panhandle between Perdido Bay, FL, and Suwannee River, FL. The data are from Florida's Inshore Marine Monitoring and Assessment Program.

Estuarine Drainage Area	Number of Observations
Apalachee Bay, FL	18
Apalachicola Bay, FL	55
Choctawhatchee Bay, FL	36
Pensacola Bay, FL	67
Perdido Bay, FL	7
St. Andrew Bay, FL	60
Suwannee River, FL	56
TOTAL	299

Appendix C

List of Acronyms and Abbreviations

BOD	biochemical oxygen demand
BW	bottom water
C	carbon
Chl-a	chlorophyll-a
CWA	Clean Water Act
DIN	dissolved inorganic nitrogen
DO	dissolved oxygen
EDA	estuarine drainage area
EMAP	Environmental Monitoring and Assessment Program
EPA	U.S. Environmental Protection Agency
IMAP	Florida Inshore Monitoring and Assessment Program
μM	micromolar
mM	millimolar
N	nitrogen
N_2	molecular nitrogen
NCA	National Coastal Assessment
NH_4^+	ammonium
NO_2^-	nitrogen dioxide
NO_3^-	nitrate
NO_X	nitrogen oxide
NOAA	National Oceanic and Atmospheric Administration
O_2	molecular oxygen
ORD	Office of Research and Development
P	phosphorus
PO_4^{3-}	phosphate
R_p	plankton community respiration rate
SAV	submerged aquatic vegetation
SOD	sediment oxygen demand
TN	total nitrogen
TP	total phosphorus
USGS	U.S. Geological Survey