Empirical Approaches to Establishing Numeric Nutrient Criteria for Southwest Florida Estuaries

Prepared for:
Tampa Bay Estuary Program

Sarasota Bay Estuary Program

Charlotte Harbor National Estuary Program

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FOREWORD

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1.0 Background

The U.S. Environmental Protection Agency (USEPA) is developing numeric nutrient water quality standards for Florida waters, including lakes and flowing waters, and estuaries and coastal waters. The schedule for estuarine and coastal water criteria has been recently modified and requires USEPA to propose estuarine and coastal waters nutrient criteria and downstream protective values in Florida by November 14, 2011 to allow for peer review by the Science Advisory Board (SAB) and to allow for public comment, followed by USEPA revision of the proposed numeric nutrient criteria.

This document presents potential empirical methods for use in developing numeric nutrient criteria for southwest Florida estuaries. The methods discussed below build on research developed by the Tampa Bay, Sarasota Bay, and Charlotte Harbor National Estuary Programs and on previous USEPA work (USEPA, 2009) and reviews by its Science Advisory Board (SAB, 2010).

After review and comment by USEPA and any further peer review, we plan to apply these methods to selected southwest Florida estuaries, and make recommendations for appropriate estuarine numeric nutrient criteria and commensurate Downstream Protective Values for upstream waters.

1.1 Definition of Numeric Nutrient Criteria and Regulatory Framework

Numeric nutrient water quality criteria define levels of nutrients (i.e., nitrogen and phosphorus) protective of the designated uses of water bodies from over-enrichment, as prescribed by the Clean Water Act (CWA). Over-enrichment of water bodies by nitrogen and phosphorus typically stimulates plant and microbial growth, and can result in biological and physical responses that adversely affect water quality and aquatic life. The USEPA nutrient criteria guidance recommends development of criteria for both total nitrogen (TN) and total phosphorus (TP), the primary causal constituents, and for chlorophyll *a* and water clarity, the primary response constituents, while not precluding the use of alternative causal or response constituents (USEPA, 2009).

The numeric nutrient criteria to be proposed by USEPA are the result of legal action brought against USEPA. In 2008, Earthjustice, a public interest law firm, filed a lawsuit to require USEPA to promulgate numeric nutrient water quality standards for Florida waters. The lawsuit was filed on behalf of the Florida Wildlife Federation, Sierra Club, Conservancy of Southwest Florida, Environmental Confederation of Southwest Florida, and St. Johns Riverkeeper. On January 14, 2009, USEPA made a determination under section 303(c)(4)(B) of the CWA that numeric nutrient water quality criteria for lakes and flowing waters and for estuaries and coastal waters were necessary for the State of Florida to meet the requirements of the CWA section 303(c), and that Florida's existing narrative nutrient criteria were insufficient to ensure protection of the State's water bodies. This determination recognized that although Florida had expended considerable effort to diagnose and control nutrient pollution, water quality degradation from nutrient pollution was significant and likely to worsen in the State. The January 2009 determination noted USEPA's intent to propose numeric nutrient standards for lakes and flowing waters within twelve months, and for estuaries and coastal waters within 24 months.

In August 2009, USEPA entered into a Consent Decree with Earthjustice to settle the 2008 litigation. USEPA committed to proposing numeric nutrient standards for lakes and flowing waters by January 2010, and for estuaries and coastal waters by January 2011. Final standards were to be established for lakes and flowing waters by October 2010 and for estuaries and coastal waters by October 2011. As noted above, both parties to the USEPA Consent Decree recently agreed to extend the timeline for proposed estuarine and coastal waters nutrient criteria to November 14, 2011, and to August 15, 2012 for finalized criteria.

The State of Florida had responded to the CWA in the late 1990s by developing a method to identify water bodies that exceeded water quality standards. Section 303(d) of the CWA requires states to submit lists of surface waters that do not meet applicable water quality standards and establish Total Maximum Daily Loads (TMDLs) for these waters. TMDLs establish the maximum amount of a pollutant that a water body can assimilate without causing exceedances of water quality standards. Chapter 99-223, Laws of Florida, known as the Florida Watershed Restoration Act (FWRA), defines a TMDL as "…the sum of the individual wasteload allocations for point sources and the load allocations for nonpoint sources and natural background. Prior to determining individual wasteload allocations and load allocations, the maximum amount of a pollutant that a water body or water segment can assimilate from all sources without exceeding water quality standards must first be calculated."

The FWRA directed the Florida Department of Environmental Protection (FDEP) to develop a methodology, and adopt it by rule, that clearly defined those waters that should be included in the state's 303(d) list of impaired waters (those waters that do not meet applicable water quality standards and for which TMDLs must be developed). FDEP formed a Technical Advisory Committee (TAC) to develop a method to define impairment of lakes, streams, and estuaries. The TAC meetings started in July 1999, and in 2001 the Environmental Regulation Commission adopted by rule (Chapter 62-303, F.A.C., Identification of Impaired Surface Waters, or IWR) the scientific approach for guiding FDEP's process for identifying and prioritizing impaired surface waters in Florida.

FDEP implemented a detailed, USEPA-approved plan for the development of numeric nutrient criteria and proposed revisions to Chapter 62-302, F.A.C. (Water Quality Standards) and Chapter 62-303, F.A.C. (Impaired Waters Rule) (IWR) to establish numeric nutrient criteria for lakes and streams. The plan called for adoption of criteria by the end of 2010, but was accelerated in response to USEPA's January 2009 determination that criteria are necessary to implement the CWA. The revised schedule following the January 2009 determination, as noted above, was for proposed criteria for lakes and flowing waters by January 2010, and for estuaries and coastal waters by January 2011.

These numeric nutrient criteria were to be in addition to the narrative nutrient standard used for years by Florida, stated in Chapter 62-302.530, F.A.C., "…in no case shall nutrient concentrations of body of water be altered so as to cause an imbalance in natural populations of flora or fauna." FDEP planned to develop numeric nutrient criteria and use these to assess waters for nutrient impairment, in addition to using the current narrative nutrient impairment thresholds in the IWR. Nutrients are present naturally in aquatic systems in many different forms, are necessary for proper functioning of biological communities, and are often moderated in their expression by many natural factors.

The January 2009 determination by USEPA of the necessity for criteria also included the assumption of responsibility for development of these numeric nutrient criteria by USEPA. In a letter from USEPA Office of Water Assistant Administrator Grumbles to FDEP Secretary Sole, it was noted that section 303(c)(4) of the CWA requires the Administrator to "…promptly prepare and publish proposed regulations setting forth a new or revised water quality standard…", and that "EPA will move forward to develop federal proposed regulations setting forth numeric nutrient criteria for Florida…" As of this time, the schedule for proposed criteria for estuaries and coastal waters is November 14, 2011, and for finalized criteria by August 15, 2012.

1.2 The Role of Nutrients in Estuaries

Estuaries are semi-enclosed bodies of water that have a free connection with the sea and within which seawater is diluted by freshwater (Hobbie, 2000). Estuaries are the most productive of all aquatic ecosystems. The tidal sheltered waters of estuaries support highly productive and diverse communities of plants and animals that live at the margins of the sea (USEPA, 1999; Hobbie, 2000). Tides and rivers provide a constant flow of water and nutrients that result in a beneficial environment for primary producers that form the base of the maritime food web. Estuaries typically form transition zones between the land and the sea, and are efficient at retaining dissolved chemicals and pollutants entering the estuarine waters prior to discharging to the open sea. One study estimated the retention of nitrogen and phosphorus supplied to the Potomac River at 82% (Bennett, 1983), with most of the retention in the estuarine zone near the mouth of the river. Another study found that nitrogen retention was 87% of input, and phosphorus retention was 72%, for the Patuxent River estuary (Boynton et al., 2008).

Nutrient inputs (TN and TP) from watersheds adjacent to coastal and estuarine waters can have significant impacts on estuarine function. High rates of nutrient inputs often stimulate very high rates of primary productivity. Because of this high primary productivity, estuaries provide breeding and nursery grounds for many species of fish and shellfish fisheries. Hundreds of marine organisms, including commercially valuable fish and, depend on estuaries to provide valuable habitat during different stages of their life cycles (Day et al., 1989; USEPA, 1999; McLusky and Elliott, 2004).

Since the 1970s, many scientists and managers have been studying the deterioration of estuarine ecosystems due to increases in nutrient loads and accompanying eutrophication (Paerl et al., 2006; Bricker et al., 2008; Fisher et al., 2006). The targeting of nutrient inputs from points sources, such as sewage outfalls and industrial effluent, for loading reductions was met with much success, yielding improved water quality following implementation of advanced waste water treatment (e.g., Greening and Janicki, 2006). Unfortunately, population growth and the growing need for agricultural output have led to an increase in nonpoint source pollution. It is estimated that human activity has increased the total rate of formation of reactive nitrogen globally by 33-55% through increases in agriculture via synthetic fertilizer (Howarth, 2008). Increases in reactive nitrogen have also resulted from increases in the encouragement of biological nitrogen fixation associated with agriculture and the inadvertent creation of reactive nitrogen oxides through fossil fuels combustion (Howarth, 2008; Paerl, 2006). For example, estimated increases of TN loadings to the Chesapeake Bay system are 6-8 times since the pre-colonial period, and TP loadings have increased by 13-24 times (Boynton et al., 1995).

Eutrophication in estuaries, as expressed by increased phytoplankton production, is linked to increased loadings of nitrogen. Estuarine eutrophication is defined as an increase in the rate of supply of organic matter, which includes nitrogen and phosphorus, in estuarine ecosystems which leads to increased rates of primary production (Nixon, 1995; Pinckney et al., 2001). An extensive study including information from 63 estuaries around the world was completed, utilizing data concerning phytoplankton production, chlorophyll *a*, and associated physical and chemical variables (Boynton et al., 1982). The authors found a significant positive relationship between annual phytoplankton production and nitrogen inputs, but not between production and phosphorus inputs. The authors also concluded that nutrient loading rates and recycling rates were more useful than standing stock values for predicting productivity (Boynton et al., 1982). Relationships between nutrient loadings and primary production in estuaries have been used to determine the likely trophic status of estuaries (Painting et al., 2007).

The impacts of eutrophication and how anthropogenic impacts affect the structure and function of estuaries continues to be a research thesis for scientists and managers worldwide (Paerl et al., 2006). Eutrophication has resulted in documented cases of reduced biodiversity, habitat degradation, and food web alterations (Nixon, 1995; Rabalais and Turner, 2001; Paerl et al., 2006; Bricker et al., 2008). Important sources of nutrients that support estuarine productivity include nonpoint and point source inputs from the watershed and atmospheric deposition.

Symptoms of water quality decline are typically increased chlorophyll *a* and macroalgae, low dissolved oxygen, loss of submerged aquatic vegetation, and occurrences of harmful algal blooms (HABs) (Bricker et al., 2008). Chlorophyll *a*, a pigment used in photosynthesis, serves as a measure of biomass (abundance) of phytoplankton in estuaries. Planktonic algae provide a food source for filter-feeding bivalves (oysters, mussels, scallops, clams) and zooplankton (including the larvae of crustaceans and finfish). Chlorophyll *a* concentrations can also be used as a measure of overall ecosystem health. High concentrations of chlorophyll a in estuarine waters are a primary indicator of nutrient pollution, as excess nutrient supply fuels the growth of algae. Therefore, high chlorophyll *a* concentrations can be indicative of adverse conditions for aquatic life and human recreation.

Dissolved oxygen (DO) is a critical factor affecting the available habitat in estuaries. Dissolved oxygen can be used as an indicator of the health of the ecosystem. Cultural eutrophication (nutrient excess leading to overproduction of microalgae and associated trophic imbalances) is common in estuaries near human population centers, with evidence of increasing areas impacted by low DO conditions (Hagy et al., 2004; Diaz and Rosenberg, 2008; Rabalais and Turner, 2001). DO can exhibit extreme diel cycles as a result of estuarine eutrophication. Algal photosynthesis elevates DO levels in the water during the day, but at night, when ecosystem respiration is typically greatest, DO can drop dangerously low. Eutrophication can lead to periodic or long-term hypoxia (generally considered to be water column DO concentrations less than 2 mg O_2/l) (Diaz and Rosenberg, 1995) and anoxia (complete absence of DO) in estuarine ecosystems. Fish, crabs, and shrimp can avoid hypoxic conditions, while other marine animals demonstrate the ability to withstand prolonged exposure to hypoxic conditions. DO concentrations are often quite variable in estuarine systems due to fluctuations in temperature, salinity, basin morphology, and overall productivity. Not all low DO conditions are the result of eutrophication, however. Some estuarine systems are susceptible to low DO conditions due solely to physical factors (depth, stratification, limited tidal exchange) (Hagy and Murrell, 2007).

Seagrasses are often used as indicators of estuarine ecosystem health. Estuarine management efforts have been implemented along the southwest Florida coast to increase seagrass acreage by reducing nutrient loads, and have resulted in increasing water clarity and maintenance or increases in seagrass coverage (Corbett and Hale, 2006; Greening and Janicki, 2006). Seagrasses serve several significant functions within estuaries, which are degraded if seagrass coverage declines. Human activities can harm seagrasses by degrading estuarine water quality and promoting physical disturbances and algal blooms, often attributable to eutrophication and decreased water clarity (Dawes et al., 2004; Iverson and Bittaker, 1986). Seagrasses provide food, shelter, and essential nursery habitats for many recreationally and commercially valuable species of fish, crustaceans, and shellfish (Dawes et al., 2004). They also provide food and habitat for marine mammals, such as manatees, listed species, and waterfowl. Seagrasses also help maintain water clarity by trapping fine sediments and particles with their leaves and stabilizing the estuarine sediments with their roots (Fonseca, 1989; Short et al., 2000). Seagrasses are very effective at removing dissolved nutrients from water that can enter from land runoff. The removal of sediment and nutrients improves water clarity, thereby improving overall ecosystem health.

Water clarity is a measure of light penetration through the water. Relatively clear waters are indicative of a healthy estuary, although many factors impact water clarity. Excess suspended sediments from runoff and rainfall can negatively impact water clarity. Nutrients, mainly nitrogen and phosphorus, can fuel the growth of photosynthesizing algae. High chlorophyll *a* concentrations associated with high algal biomass can decrease water clarity. Reduced light transmission can decrease seagrass abundance (Boynton et al., 1996; Stevenson et al., 1993), which can in turn impact the entire food web. Decreases in seagrass abundance also reduce habitat availability to the hundreds of species which depend on them.

1.3 Estuarine Responses to Changes in Nutrient Concentrations and Loadings

Increased nutrient loads are normally expected to result in increased algal biomass in an estuary. The response of estuarine water quality to varying nutrient loads is dependent on other factors as well, including freshwater inflow, residence time, water clarity, estuarine physiography, and trophic interactions (Howarth and Marino, 2006; Swaney et al., 2008). Freshwater inflow and tidal exchange determine hydraulic residence time, and hence the time available for nutrients to react in the estuary (Hagy et al., 2000; Bricker et al., 2008). Freshwater inflow can affect algal abundance and therefore chlorophyll *a* concentrations via enhanced nutrient supply, changing the location of peak chlorophyll *a* abundance, or decreasing chlorophyll *a* abundance because of reduced residence time (Hagy, 1996). When freshwater inflow is low, the chlorophyll *a* maximum is typically located further up-estuary than during times of high flow. Low flow also allows a longer residence time for chlorophyll *a* and nutrients. Longer residence times tend to promote slower-growing taxa which include dinoflagellates and cyanobacteria (Pinckney et al., 1999), and also allow faster-growing algal taxa to increase even more. During high flow conditions, flushing is more rapid and residence time in rivers and the estuary is reduced (Sheldon and Alber, 2006; Miller and McPherson, 1991; Hagy et al., 2000). These conditions tend to favor fast growing phytoplankton such as chlorophytes (green algae) and various flagellates (Pinckney et al., 1999). At times, high flows can be excessive, depending on the morphology of the estuary. Very high flows may not result in higher chlorophyll *a* abundance due to the relationship between the residence time of water within the system and uptake and growth rates of the phytoplankton community. Changes in flow can also impact the community composition with less desirable species dominating during times of low flow and longer residence times (Bricker et al., 2008).

The dependence of water clarity on colored substances and suspended solids is of particular importance in southwest Florida estuaries, more so than in most other estuarine regions. Dissolved and suspended matter may have more influences on water clarity than does algal biomass (Corbett and Hale, 2006; Christian and Sheng, 2003). The watersheds of southwest Florida estuaries are characterized by significant wetland extents that contribute relatively large amounts of colored dissolved organic matter (CDOM) to the estuaries (McPherson and Miller, 1987; Dixon and Kirkpatrick, 1999). CDOM must be considered when developing relationships between incident light and light at depth within southwest Florida estuaries.

2.0 Objective

The objective of this document is to identify potential methods for derivation of technically defensible numeric nutrient criteria for Southwest Florida estuaries (Figure 2-1), based on scientifically sound and robust methods. Input from USEPA and others is encouraged.

To this end, the remainder of this document contains the following:

- Approach and Rationale – A discussion of the approaches available to derive numeric nutrient criteria, identification of the appropriate approach, and the methods we will employ to use this approach for numeric nutrient criteria derivation.
- Implementation – A discussion of factors to consider when developing an implementation strategy, including both technical aspects (e.g., temporal and spatial extents, seasonality, natural variability) and regulatory aspects (e.g., time period for compliance evaluation, etc.).

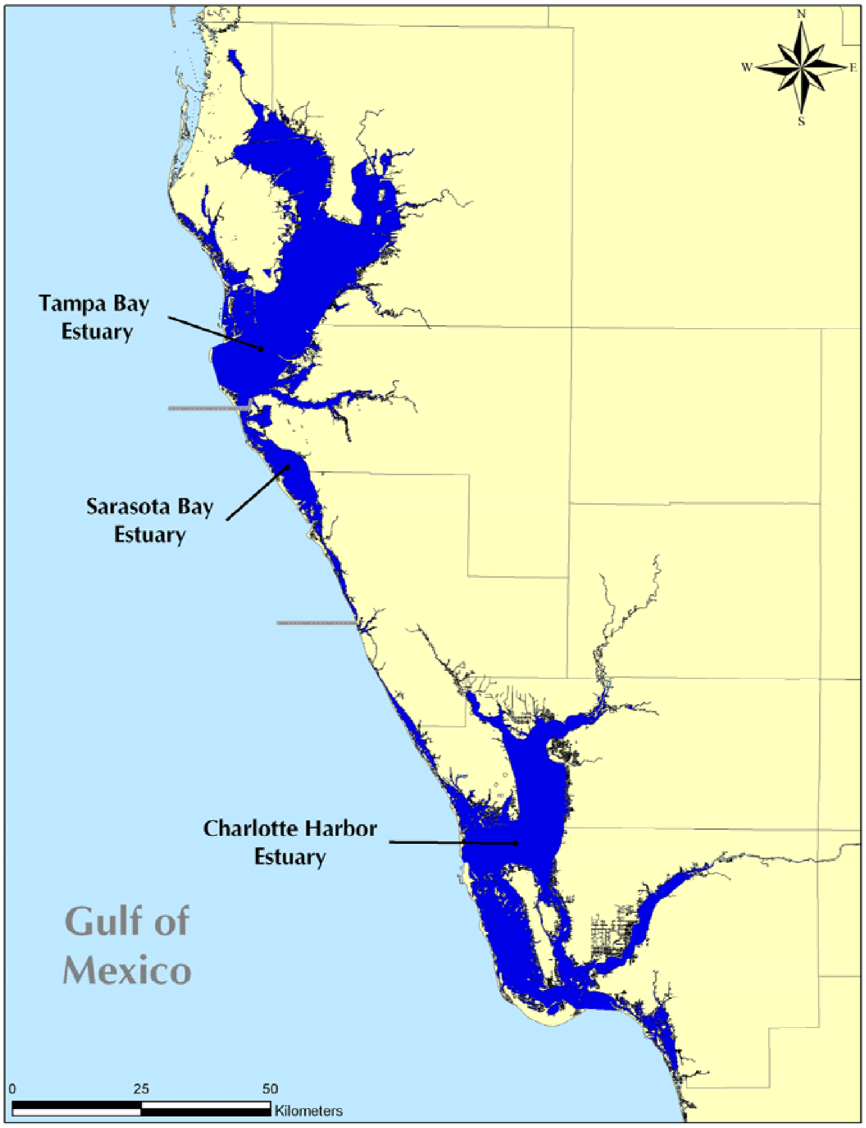

Figure 2-1. Southwest Florida estuaries – Tampa Bay, Sarasota Bay and Charlotte Harbor.

3.0 Approach and Rationale

The USEPA has previously furnished guidance for deriving nutrient criteria in the form of peer-reviewed technical approaches and methods (USEPA, 2000a; 2000b; 2001; 2008; 2009). USEPA clearly states their view on development of nutrient criteria in the recent Science Advisory Board Review Draft (USEPA, 2009).

> "USEPA's view is that the criteria derivation process for the toxic effect of chemical pollutants is not applicable for nutrients because effects, while linked to widespread and significant aquatic degradation, occur through a process of intermediate steps that cannot be easily tested in simple laboratory studies. As a result, nutrient criteria derivation relies in large part on empirical analysis of field data."

3.1 Approaches

USEPA and others have identified three analytical approaches for the development of nutrient criteria:

- the reference condition approach,
- stressor-response analysis, and
- mechanistic modeling.

The approaches are briefly described below.

3.1.1 Reference Condition Approach

As implied by the name, the reference condition approach is based on determining criteria based on a group of reference waterbodies. The reference waterbodies are selected from among a group of like waterbodies (e.g., the same class of waterbodies) that represent minimally disturbed conditions (Stoddard et al., 2006) and have similar characteristics (e.g., black-water streams).

Data from the reference waterbodies are assembled and the distributions of either causal or response variables are analyzed (Figure 3-1). Because these reference waterbodies typically are intended to represent minimally disturbed conditions (or at least an acceptable level of disturbance), USEPA has used specific percentiles derived from these systems (USEPA, 2009) to develop nutrient criteria. Generally, some percentile of the reference stream distribution is chosen to represent the criterion value but caution should be used in selecting the appropriate value to serve as the benchmark criterion value (Rohm et. al., 2002; Suplee et al., 2007).

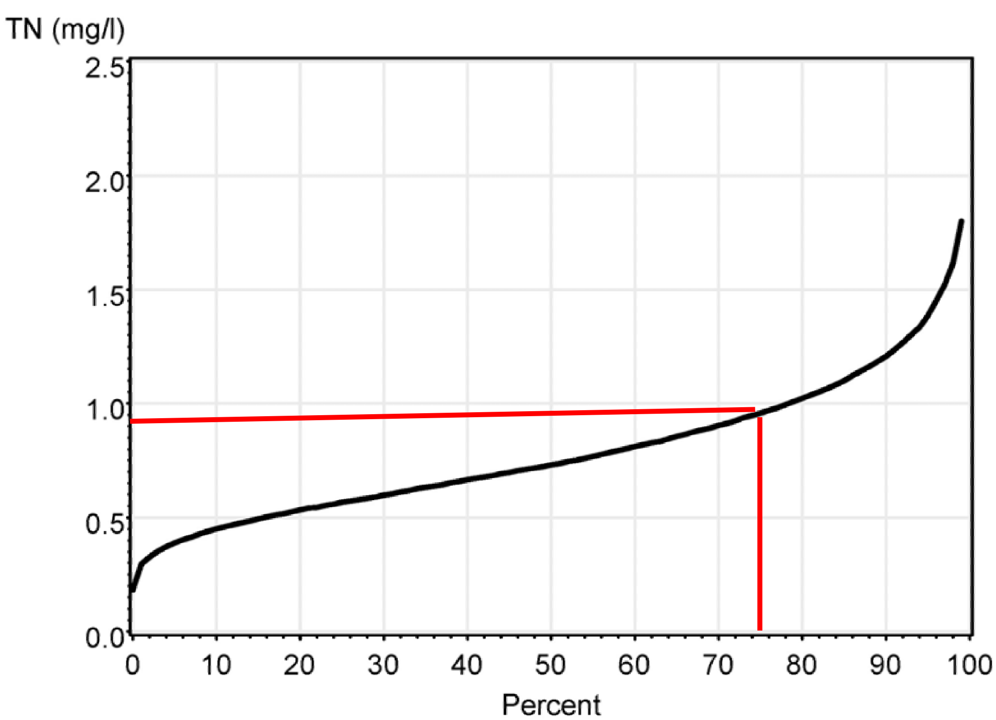

Figure 3-1. Example of a Reference Condition approach to set a numeric TN criterion.

3.1.2 Stressor-Response Approach

The stressor-response approach consists of developing relationships between nutrient concentrations or loads and biological responses. The biological responses should be related to the "designated use of a waterbody (e.g., a biological index or recreational use measure) either directly or indirectly, but ideally quantitatively" (USEPA, 2009). After quantitative relationships have been developed, the nutrient criterion that is protective of the specific designated uses can be determined (Figure 3-2).

USEPA (2009) has provided guidance on the development of stressor-response relationships using empirical data analysis approaches and a review of these approaches by the Science Advisory Board (SAB, 2010) has provided additional insights as to how evidence of stressor-response relationships may be used in establishing numeric nutrient criteria. A more comprehensive discussion of stressor-response methods for determining numeric nutrient criteria can be found below.

3.1.3 Mechanistic Modeling Approach

The mechanistic modeling approach is used to predict specific constituents based on a series of equations and algorithms that represent physical, chemical, biological, and ecological processes. Mechanistic models include a wide variety of water quality models, some of which were briefly described in previous USEPA nutrient criteria guidance documents (USEPA 2000a, 2000b). A much more in depth discussion of water quality modeling theory and practice can be found in a wealth of references (e.g., Chapra, 1997; Martin and McCutcheon, 1998; Edinger, 2002).

Mechanistic models are valuable tools and where available and useful we intend to explore their use in supporting the development of numeric nutrient criteria for southwest Florida estuaries. Mechanistic models tend to integrate information on the interactions of major ecosystem processes to derive quantitative estimates of effects and maybe valuable in interpreting the stressor-response relationship (SAB, 2010). However, their first-order approximations can underestimate the variability and uncertainty in the predictions.

Since reference system approaches tend to be based on subjective decisions on what constitutes the reference condition, we consider them to be the least desirable of the three approaches. For cases where limited data exist this may be the only viable option to set site-specific water quality criteria. The remainder of this document focuses on defining proposed methods for developing the stressor-response approach and implementing the results for developing numeric nutrient criteria in southwest Florida estuaries. However, it is important to note that all three methods will be evaluated and potentially utilized to set numeric nutrient criteria for southwest Florida estuaries.

3.2 Framework for Establishing Numeric Nutrient Criteria

USEPA has explicitly stated that any of the methods described above may be used individually or some combination of methods may be used to derive numeric nutrient criteria (USEPA, 2009). For southwest Florida estuaries where sufficient data are available, the stressor-response approach is preferable for delivering a weight-of-evidence that can be used in the development of appropriate numeric nutrient criteria.

As part of its review of the USEPA *Empirical Approaches for Nutrient Criteria Derivation*, the SAB recommended revisions to the 5-step framework presented in that document. The revised framework provides a very useful means to organize our discussion of potential methods for establishing numeric nutrient criteria for southwest Florida estuaries (Figure 3-2).

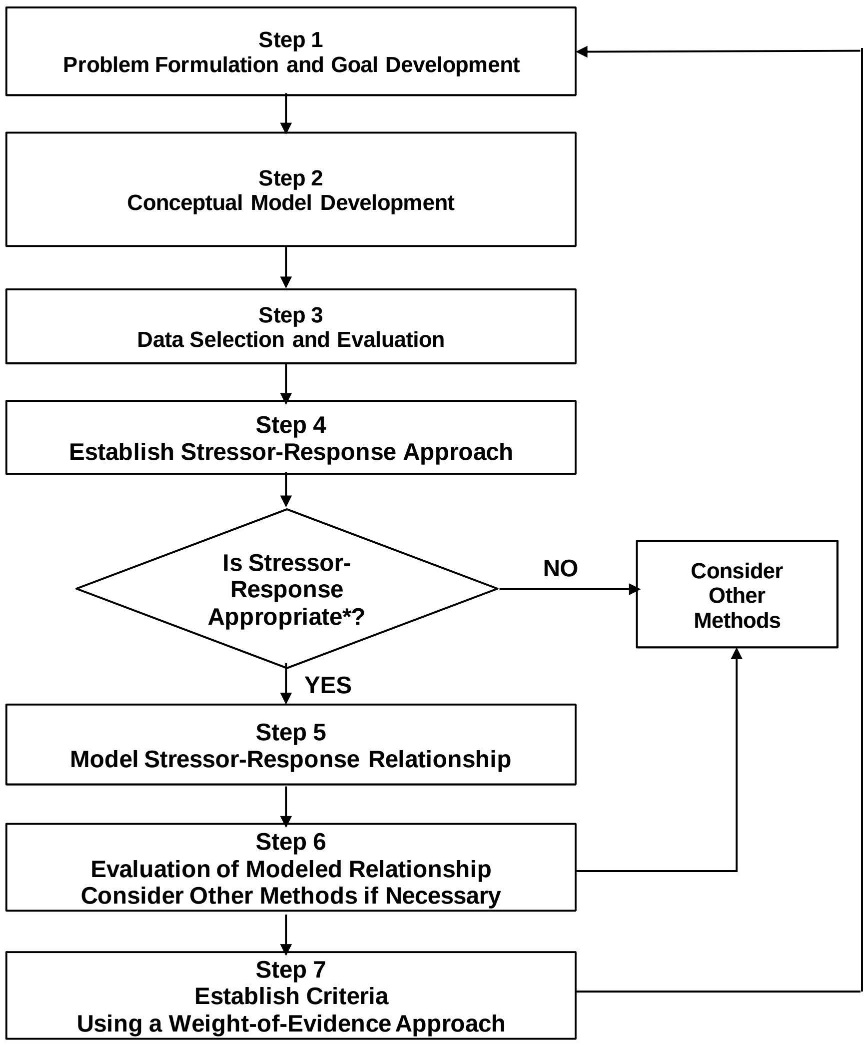

Figure 3-2. Framework for Establishing Numeric Nutrient Criteria (modified from SAB, 2010).

The use of a weight of evidence approach will be a critical aspect of our intended development of proposed criteria. Aside from the USEPA guidance on empirical approaches to developing numeric nutrient criteria, there are many peer reviewed articles on the development of criteria that will be utilized as guidance in our efforts. What follows is a detailed description of our proposed analytical approach with citations to documents that were instructive in providing guidance in formulating our proposed approach. It is important to note that we have a variety of analytical tools available to us and that we have demonstrated capability to employ these methods towards this issue.

Our principle analytical software that will be used in this project is Statistical Analysis Systems (SAS V9.2). This software is the gold standard for analysis due to rigorous testing and validation of its statistical procedures, documentation provided by the source code, and log files and output listings documenting all steps taken in the analytical process. This allows for the easy replication of results which will benefit the USEPA in its evaluation of our methods. However, recent advances in sophisticated methods such as recursive partitioning (aka. changepoint analysis) have been popularized in several efforts to develop numeric nutrient criteria and these methods are commonly executed in the R software language. As described in the following sections, we intend to use both SAS and R in the analytical phase of this project. This allows for a full complement of analytical techniques to be employed to develop numeric nutrient criteria for southwest Florida estuarine waters. Where R is used in developing predictive linear models, the modeling efforts will be replicated in SAS as a quality assurance step. What follows is a detailed description of each step in our intended analytical approach. Methods are described and examples are provided with pertinent literature cited that will allow the reader to have a full understanding of the analytical flow path.

3.2.1 Problem Formulation and Goal Development

To a large extent the goals have been defined by the USEPA - to develop numeric criteria for total nitrogen (TN) and total phosphorus (TP) in southwest Florida estuarine waters which fully support their designated aquatic life uses. Therefore, a definition of the appropriate stressor (causal) and response variables is required.

There has been a substantial amount of work in southwest Florida directed towards defining seagrass, water quality, and nutrient loading targets, as well as defining the stressor-response relationships for these estuarine waters (Janicki et al., 1994; Janicki and Wade, 1996; Tomasko et al., 2005; Corbett and Hale, 2006; Greening and Janicki, 2006; Janicki et al., 2008; Janicki Environmental, 2009; Janicki et al., 2009; Dixon et al., 2010). We fully intend to utilize existing technically sound information as a basis from which to identify the response variables of interest corresponding to designated uses and the anthropogenically influenced stressor variables that result in adverse effects to estuarine health. USEPA has employed chlorophyll *a* concentrations (as a surrogate measure of phytoplankton concentration) as a biological response endpoint. This endpoint is also commonly used in a regulatory framework for evaluating water quality and is often used as an index defining the trophic state of waterbodies.

The Tampa Bay Estuary Program and the Sarasota Bay Estuary Program currently have target chlorophyll *a* concentrations that are used to provide management level indicators of year-to-year variability in estuarine condition. These target chlorophyll *a* concentrations are related to the light attenuating properties of phytoplankton and subsequent effects on the light available to seagrasses. Higher chlorophyll *a* concentrations reduce the incident light available to seagrasses for photosynthesis. These targets provide important links to seagrass extent management goals defined by the National Estuary Program Comprehensive Conservation and Management Plans. Chlorophyll a concentrations are also used by the FDEP as a primary criteria for determining impairment of waterbodies in both freshwaters and estuaries. Chlorophyll *a* not only affects light penetration through the water column but the degradation and decay of phytoplankton can also affect water column dissolved oxygen concentrations. FDEP also regulates standards for DO that are used to determine impairment. The value of using these parameters as response variables is in their direct linkage to nutrient enrichment in the causal pathway as described above.

Therefore, at a minimum, chlorophyll *a* and DO concentrations will be used as the primary response variables of interest. The methodology presented in this document is intended to be used to develop numeric nutrient criteria for total nitrogen (TN) and total phosphorus (TP) using chlorophyll *a* and DO concentrations as the biological endpoints. Therefore, these nutrients are defined as the stressor variables of interest. Relationships between nutrient enrichment and estuarine eutrophication are well known and accepted; however, there are many factors such as hydrologic retention times and seasonality that influence the ability of phytoplankton to take up nutrients. Further complications arise when trying to assess stressor-response relationships between nutrients and response when the limiting nutrient is not considered or when co-limitation may occur over the range of the relationship. Therefore, we intend to consider how covariates such as residence times, seasonality, colored dissolved organic matter and other potential confounding factors influence the relationship between nutrients and phytoplankton responses in southwest Florida estuaries. We intend to document the physical, chemical and biological variables comprising the morphological relationships (e.g., habitat, spatial, and temporal) that define the aquatic system of interest, and which may be important in modifying the relationship between nutrient concentrations (both nitrogen and phosphorus) and observed biological endpoints as recommended by the SAB (2010). Conceptual models are a valuable tool to visualize and refine the conceptual framework under which subsequent data analysis will take place.

The southwest Florida estuaries (Figure 2-1) considered here are all data rich estuaries. Many years of discrete water quality sampling in both the estuaries and their watersheds, nutrient loading estimates, and many other studies including seagrass coverage and routine fisheries monitoring are available. The forethought and insight of local scientists and resource managers in these estuaries has allowed for analysts to have available to them a wealth of empirical information from which to consider stressor-response relationship as part of the development process for numeric nutrient criteria.

It has been widely accepted that all three southwest Florida estuaries display a significant degree of spatial variability that has largely been captured by the delineation of segments within each estuary. This segmentation has also structured much the management strategies that have been employed over the past 15-20 years.

The morphology of Tampa Bay lends itself to division of the bay into distinct segments. Lewis and Whitman (1985) provided a segmentation scheme which defined seven segments of the bay (Figure 3-3), including the four mainstem segments: Old Tampa Bay, Hillsborough Bay, Middle Tampa Bay, and Lower Tampa Bay. This segmentation scheme has been utilized since the initiation of water quality assessment and pollutant loading development for Tampa Bay. The segments differ in watershed attributes, including size, land use characteristics, ratio of segment volume to watershed area, circulation and mixing, and distance from the Gulf of Mexico. Analysis of water quality data supports the segmentation of the bay, with differences in water quality obvious between the segments (Appendix 1).

Segment-specific loadings to Tampa Bay (Zarbock et al., 1994, 1996; Janicki Environmental, 2001, 2005, 2009) have been successfully linked to segment-specific water quality, for the four mainstem segments of the bay. The TBEP utilized segment-specific loading-water quality relationships based on empirical data, as detailed in Janicki and Wade (1996), to set water quality targets consistent with seagrass protection and restoration in the bay. The Tampa Bay CCMP implemented management actions based on these relationships, and has been very successful in meeting water quality targets, and as a result has seen increases in seagrass acreage in Tampa Bay (Greening and Janicki, 2006). The Tampa Bay TMDL was developed from segment-specific loading targets that Tampa Bay resource managers voluntarily committed to maintain (TBEP, 2010).

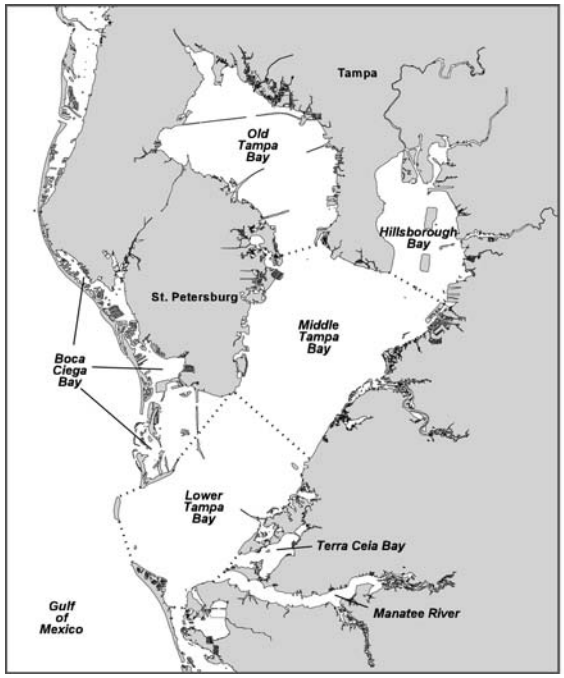

Figure 3-3. Tampa Bay segmentation as recognized by the Tampa Bay Estuary Program.

The spatial variability among the lagoonal estuarine segments of Sarasota Bay has also been recognized. Sarasota County's Ambient Water Quality Monitoring Program follows a segmentation scheme developed by Mote Marine Laboratory (Estevez and Palmer, 1990). The design was developed for the Sarasota Bay Estuary Program (SBEP) and is described in the SBEP's Framework for Action (SBEP, 1992). This segmentation scheme was developed to delineate areas with similar hydrologic features and impacts from the land. The term "Problemsheds" was introduced to represent geographic areas that most likely encompass problems causally related to essential features of the area and it was expressly stated that as more information became available, the segmentation scheme could be adjusted to reflect additional knowledge gained as a result of additional data collection efforts. Within each segment, sub-segment polygons (strata) were established to distribute sampling effort evenly throughout the segment. Sarasota County Water Planning and Regulatory (County) has been routinely sampling water quality since 1995 using this segmentation scheme at a sampling frequency of one sample per stratum per month. The sampling design is a hierarchical, fixed station design. Initially, the individual monitoring locations within each stratum were randomly generated and the locations have since remained fixed at their initial placement.

More recently, seagrass and water quality "targets" have been developed for the SBEP (Janicki et al., 2008; Janicki Environmental, 2010) using a variation of this segmentation scheme (Figure 3-4). The only change to the scheme was the combination of 4 segments in Sarasota Bay (i.e., 7, 8, 10, 11) into the "Sarasota Bay" segment for expressing seagrass and water quality targets. Otherwise, the segmentation is identical to that originally proposed by Estevez and Palmer (1990). Sarasota Bay proper is a large waterbody and while jurisdictionally divided by the Manatee–Sarasota County line it was recognized by local scientist and natural resource managers that the benefits of expressing management targets for the entire bay as a single number outweighed the intra-bay differences in water quality.

The Charlotte Harbor National Estuary Program (CHNEP) study area is expansive and includes a diverse set of estuarine waterbodies from lagoonal systems to open estuarine segments with direct tidal forcing. Several segmentation schemes have been used for various purposes as described in Janicki Environmental (2009). These segmentation schemes were compared and contrasted to identify a segmentation scheme to use for development and reporting of the status of seagrass and water quality in the study area. Based on outcomes of several meetings with local experts as well as objective analytical work it was decided that while several segmentation schemes have been developed in the CHNEP study area to serve various purposes, the segmentation scheme developed for the Coastal Charlotte Harbor Water Quality Monitoring Network (CCHMN), a probabilistic sampling design, was the best segmentation scheme to use for development and reporting of the status and trends in seagrass and water quality (Figure 3-5).

The CCHMN segmentation scheme was developed based on logistical, jurisdictional, and analytical considerations as well as professional judgment with the recognition that estuarine water quality is a dynamic gradient and is not constrained by any management segmentation scheme. There was general consensus among the meeting participates that the CCHMN segmentation scheme could serve as the foundation for a segmentation scheme for reporting on water clarity and seagrass condition over time though some segments were identified for further evaluation. Analysis of water quality data from segments identified for further evaluation revealed that the water quality constituents thought to limit light in estuaries were similar between many of the sub-segments with some exceptions. Exceptions were the distribution of color values in Estero Bay where the northern portions tended to have higher color than the southern extent and turbidity and in Pine Island Sound where the southern extent tended to have higher turbidity values than the northern sub-segment. However, seagrass trends and acreage estimates were provided that showed similar trends between most sub-segments identified.

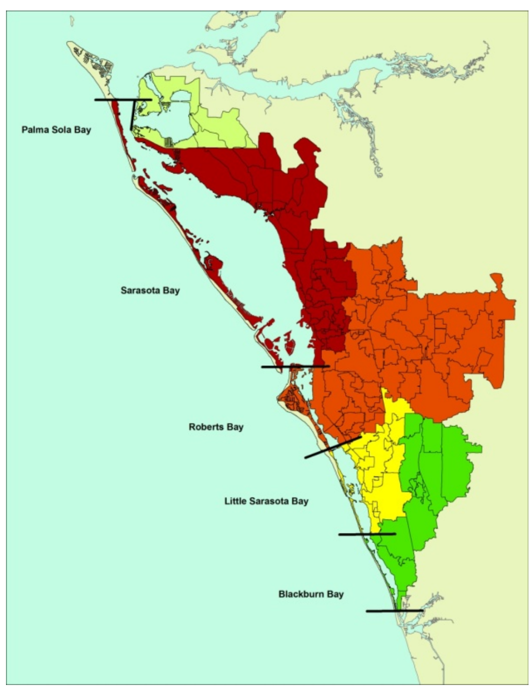

Figure 3-4. Sarasota Bay Estuary Program segments and contributing watersheds.

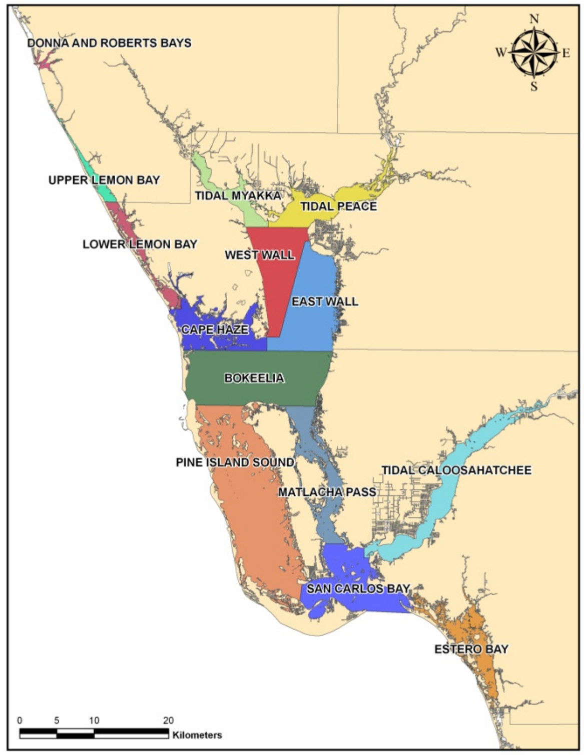

Figure 3-5. Charlotte Harbor National Estuary Program segments.

3.2.2 Conceptual Models

Conceptual models provide a framework of understanding for the paradigm considered for subsequent data analysis. We have established conceptual models as a primary step in the analytical flow path to define the paradigm under which subsequent analyses will take place (Figure 3-3 and Figure 3-4). Specific hypotheses can be generated based on the paradigms illustrated by these conceptual models.

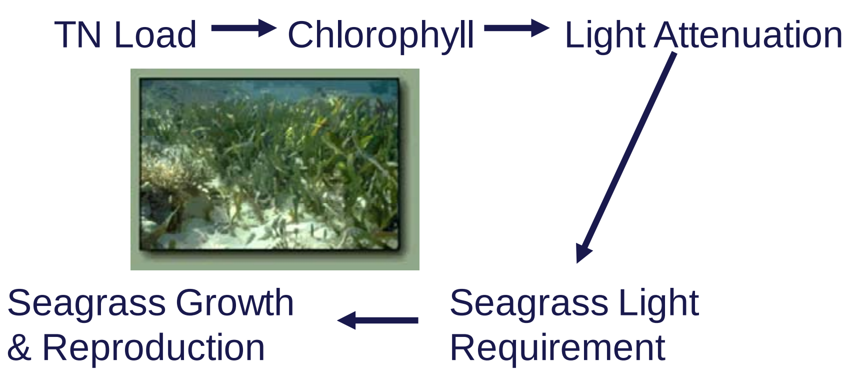

Figure 3-3. Conceptual model defining the relationships between nutrient loading and seagrasses.

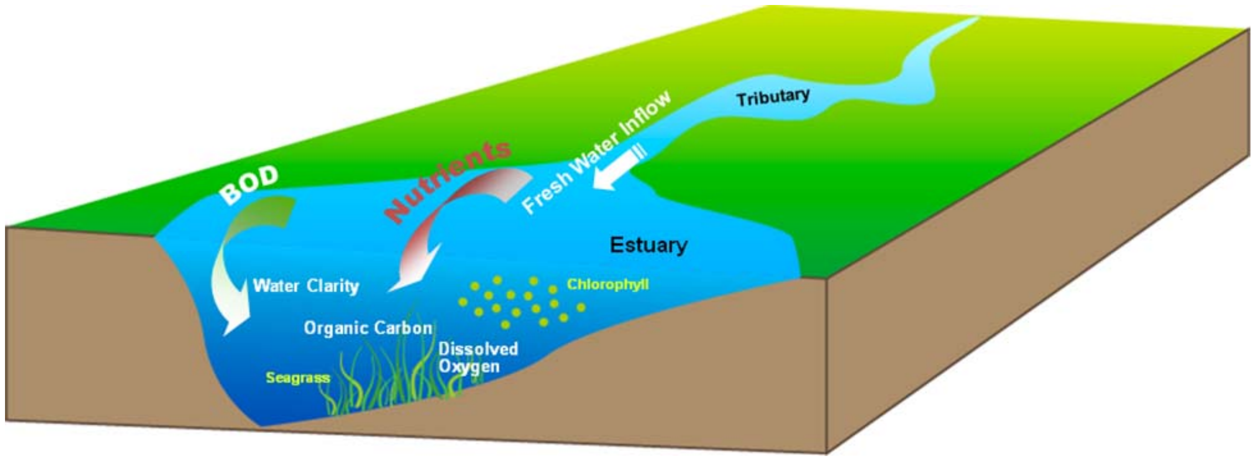

Figure 3-4. Conceptual model of relationship between watershed inputs and estuarine responses.

3.2.3 Data Selection and Evaluation

There are several principles that we intend to follow with respect to data analysis. A primary consideration is that analysis should follow the design used to generate the data. For example, analysis of data collected using a fixed station sampling design should be conducted differently than analysis conducted on data from probabilistic designs. There are advantages and compromises associated with both designs but recognition of the design allows for the proper analytical tool to be chosen. We intend to characterize the sampling design from which the data were derived for each analytical component in the development of the numeric nutrient criteria and explicitly state the assumptions and limitations of the sampling design as it pertains to any outcomes derived from an analysis. Another important component of the preliminary evaluation is the visualization of the data using exploratory data analysis methods. Exploratory data analysis methods will be based on the conceptual models developed in the previous step. Scatterplots will be generated to visually explore relationships among potential stressor and response variables. Maps can be generated that display the spatial variability for particular stressor and response variables of interest within an estuary. Univariate summary statistics will be generated as an initial exploratory data analysis step as well as Cumulative Frequency Diagrams (Cumulative Distribution Function) plots, and time series plots. Inter and intra-annual box plots are valuable tools to explore the temporal variability in the metric of interest. Simple correlation metrics such as Pearsons correlation coefficient (Rho), and Spearmans rank correlation coefficient (Rho S) will also be utilized in the exploratory data analysis phase.

3.2.4 Model the Stressor-Response Relationship

The next step in the analytical path is to model the stressor-response relationship. Our proposed method of using stressor-response relationships to develop numeric nutrient criteria is conceptualized in the illustration provided in Figure 3-5. At this point the determination has been made that a stressor-response model is the preferred method for assessment. The problem has been formulated and a conceptual "model construct" has been developed that describes the hypothesized cause and effect relationship. Once the model construct has been established, empirical analysis approaches will be developed from which analytical outcomes will be generated.

As illustrated in Figure 3-5, analytical outcomes will be generated from the data analysis and these analytical outcomes will be evaluated as to their relevance to the conceptual model construct. If the analytical result does not agree with the model construct, then the model construct will be revisited. If the outcome supports the model construct, then the outcome will be compared as to its relevance to existing information. Since there is a wealth of information established in the stewardship of these estuarine systems, the analytical outcome will be weighed against this existing information. If the analytical outcome supports existing information and the conceptual model construct, then an assessment will be conducted as to the appropriate mechanism to implement the analytical outcome as a potential numeric nutrient criterion. This will be further discussed in the implementation section.

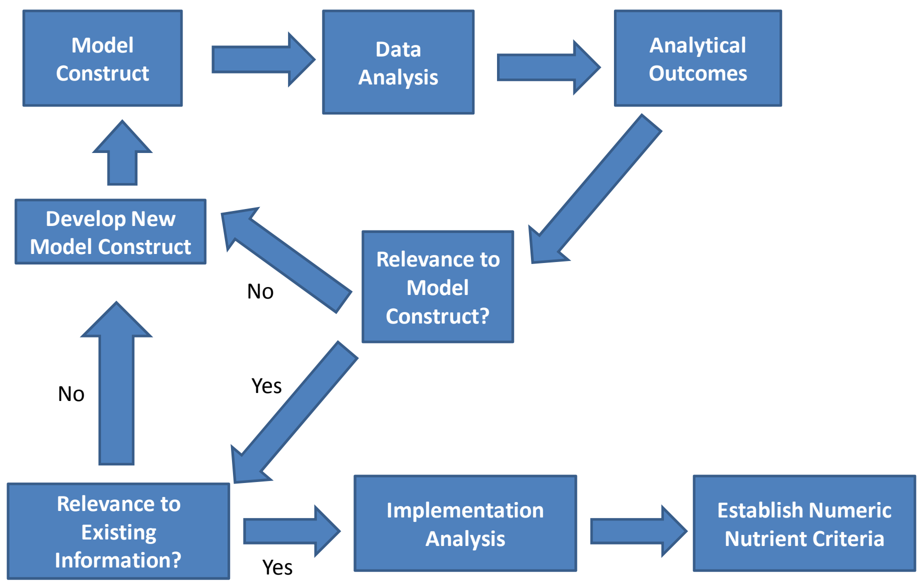

Figure 3-5. Analytical flow path to development of Estuarine Numeric Nutrient Criteria.

3.2.5 Stressor-Response Modeling

Previous research on the southwest Florida estuaries considered, the USEPA guidance on empirical approaches for nutrient criteria development (USEPA, 2009), and the SAB review (SAB, 2010) have guided the development of the methods proposed in this document. These documents present a suite of analytical tools that can be used in establishing numeric nutrient criteria given that adequate data are available for the assessment and that adequate steps are taken to ensure that the stressor-response relationship represents biologically relevant comparisons between primary production as measured by chlorophyll a and some function of variation in nutrient inputs.

The USEPA guidance on the use of empirical approaches to establishing numeric nutrient criteria has detailed several analytical tools which we intend to employ as part of our analytical approach. Linear regression, logistic regression, and quantile regression are all useful tools that are commonly used in assessing stressor-response relationships. In the USEPA Guidance document, USEPA characterizes these tools as appropriate for use when response thresholds have been previously established. The rate of change estimates provided by the slope coefficient in these relationships can be used to predict the criterion value results in the established threshold. Changepoint methods are valuable tools to identify thresholds indicating where a rate of change in response to a stressor may change dramatically indicating a nonlinear threshold value such as the assimilative capacity of a system under study. Changepoint methods are rapidly evolving from simple data mining tools to predictive models using advanced statistical algorithms to evaluate conditional probabilities in the stressor-response relationships. These methods are therefore useful to evaluate the stressor-response relationship whether or not a threshold value has been established. What follows is a brief description of each method with examples provided as necessary to convey our intended approach.

3.2.5.1 Linear Regression

Linear regression is a parametric statistical technique that is used to explore the relationship between two or more variables. In univariate ordinary linear regression, the relationship between the dependent variable (y-axis) and independent variable (x-axis) is developed. This is done by fitting a straight line through the set of points such that the sum of squared residuals of the model is as small as possible. That is to say, the vertical distances between the individual points and the fitted line are minimized. Ordinary linear regression is a well established method.

In linear regression, it is assumed that the data are independent samples from the population. For example, if one is developing numeric nutrient criteria for the tributaries of Tampa Bay, the data should come from samples that are representative of the spatial and temporal variability of the system. Another important assumption of linear regression is that the error term of the model is normally distributed, with constant variance. Often times, one or more of the variables exhibits a non-linear relationship with the other variables. While there are non-linear regression techniques that can be employed, one should try transforming the data. Ordinary linear regressions often can be developed using transformed data and these model will satisfy the assumptions of linear regression.

Diagnostic statistics and plots are commonly used to determine if the regression model meets the assumptions of linear regression. The most commonly used statistics are the statistical significance of the model coefficients and the coefficient of determination (r^2). The statistical significance of the model coefficients tests whether the slope and intercept of the model are significantly different from zero. The coefficient of determination is a measure of the variance in the dependent variable that is explained by the model. A plot of the residuals versus the independent variable can be used to judge if the assumption of constant variance is met. Additional plots of residuals versus other variables can also be instructive. For example, a time-series plot of the residuals can be used to assess whether or not the residuals vary seasonally. Additional diagnostics can be run to identify outliers and test for leverage or influential points. Data points that are identified by these additional diagnostics should be further investigated to determine if they are the result of a data entry error or other problem that merits removing them from the analysis.

Linear regression models that meet the above assumptions and satisfy the diagnostic tests should be validated using independent data or with other techniques like bootstrapping. In traditional regression, caution should be used when applying a regression model to the range of data that is outside of the bounds of the data used to develop the model. However, recent advances in specifying random effects allow for the generalization of a regression model outside of its domain.

An example of linear regression is presented below. The relationship of TN concentrations to chlorophyll *a* was investigated. Preliminary analyses between chlorophyll *a* and various water quality constituents revealed a correlation between chlorophyll *a* and TN concentration. Generally, chlorophyll *a* concentrations increase with increasing TN concentrations, though the relationship is non-linear at lower TN concentrations. Therefore, the variables were log transformed and the fit of the model was improved (Figure 3-6). Though the amount of variation explained by the model was less than 50%, the coefficients were highly significant. Obviously a large portion of the variation in chlorophyll *a* is explained by other variables that are not included in the model, suggesting the model could be improved with the addition of relevant variables and possibly interaction terms.

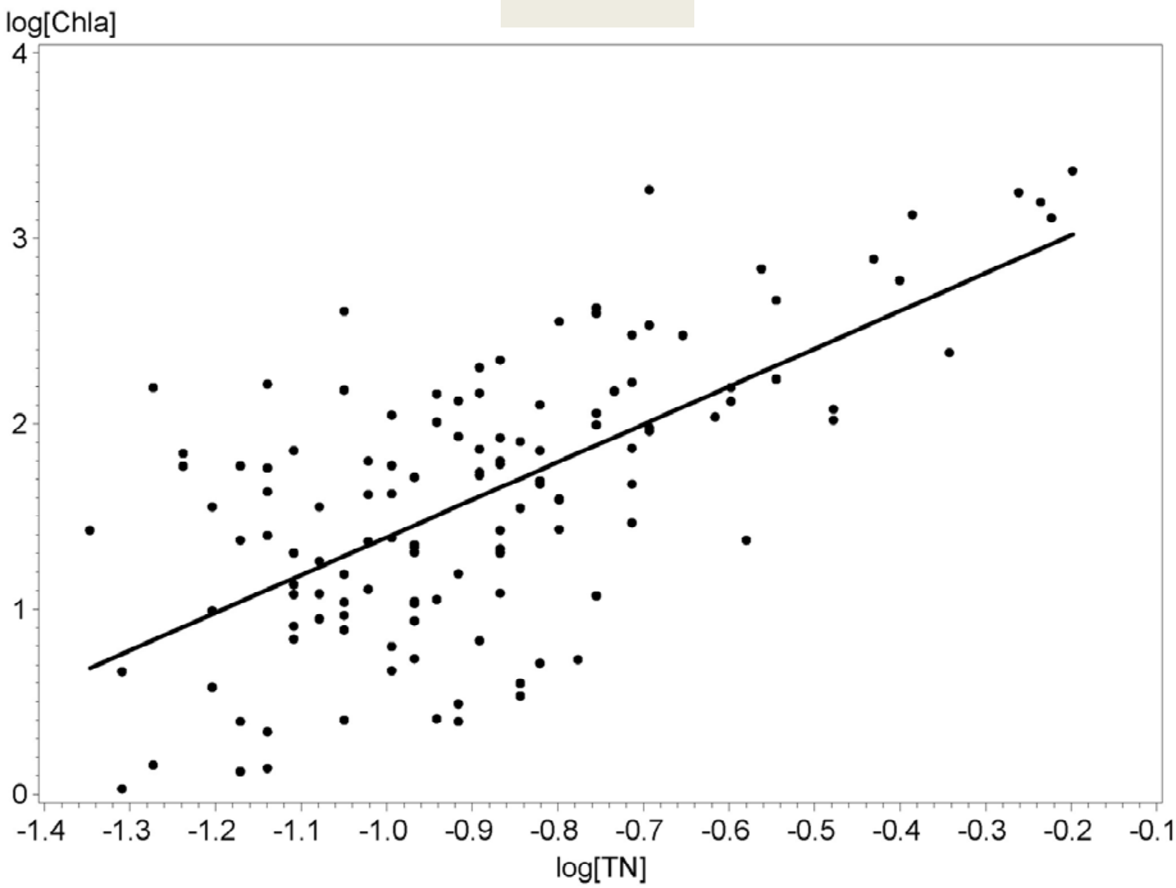

Figure 3-6. Relationship between chlorophyll *a* and TN concentrations as defined by a linear regression.

We have previously applied linear regression techniques that relate either nitrogen loading or concentrations to chlorophyll *a* concentrations in a number of segments within the three southwest Florida estuaries. The following presents some of the specific regressions that we have developed.

In Tampa Bay, a regression approach was used to define the quantitative relationship between chlorophyll *a* concentrations and nitrogen loading (Janicki and Wade, 1996). The form of the regression was:

$$C_{t,s} = \alpha_{t,s} + \beta_s (\sum_{i=0}^{2} L_{t-i})$$

where: $C_{t,s}$ = mean chlorophyll *a* concentration in month *t* in segment *s*

$\alpha_{t,s}$ = month- and bay segment-specific intercept

β_s = bay segment-specific regression coefficient

$L_{t-i,s}$ = TN load in month *t-i* and segment *s*.

Table 3-1 presents the regression coefficients derived from data for the period 1985-1991.

Table 3-1. Linear regression coefficients for relationships between chlorophyll *a* and TN loading in Tampa Bay (from Janicki and Wade, 1996).

Parameter	Value	Coefficient
Intercept		2.226599244
Month		
	1	-0.192338320
	2	0.279799163
	3	0.768803604
	4	1.270570938
	5	2.452550666
	6	3.167990897
	7	5.171125452
	8	5.378311443
	9	6.114218870
	10	4.191362957
	11	2.573604841
	12	0.000000000
Bay Segment	1	1.558367876
	2	0.320830622
	3	0.967240508
	4	0.000000000
3 month cumulative TN load*Bay Segment Interaction		
	1	0.024139861
	2	0.023465884
	3	0.010144687
	4	-0.008584323

The model was fit with 256 observations and resulted in an R^2 value of 0.73. The regression was highly significant with a probability of a greater |F| value of < 0.0001. A plot of predicted versus observed chlorophyll *a* concentrations is presented in Figure 3-7.

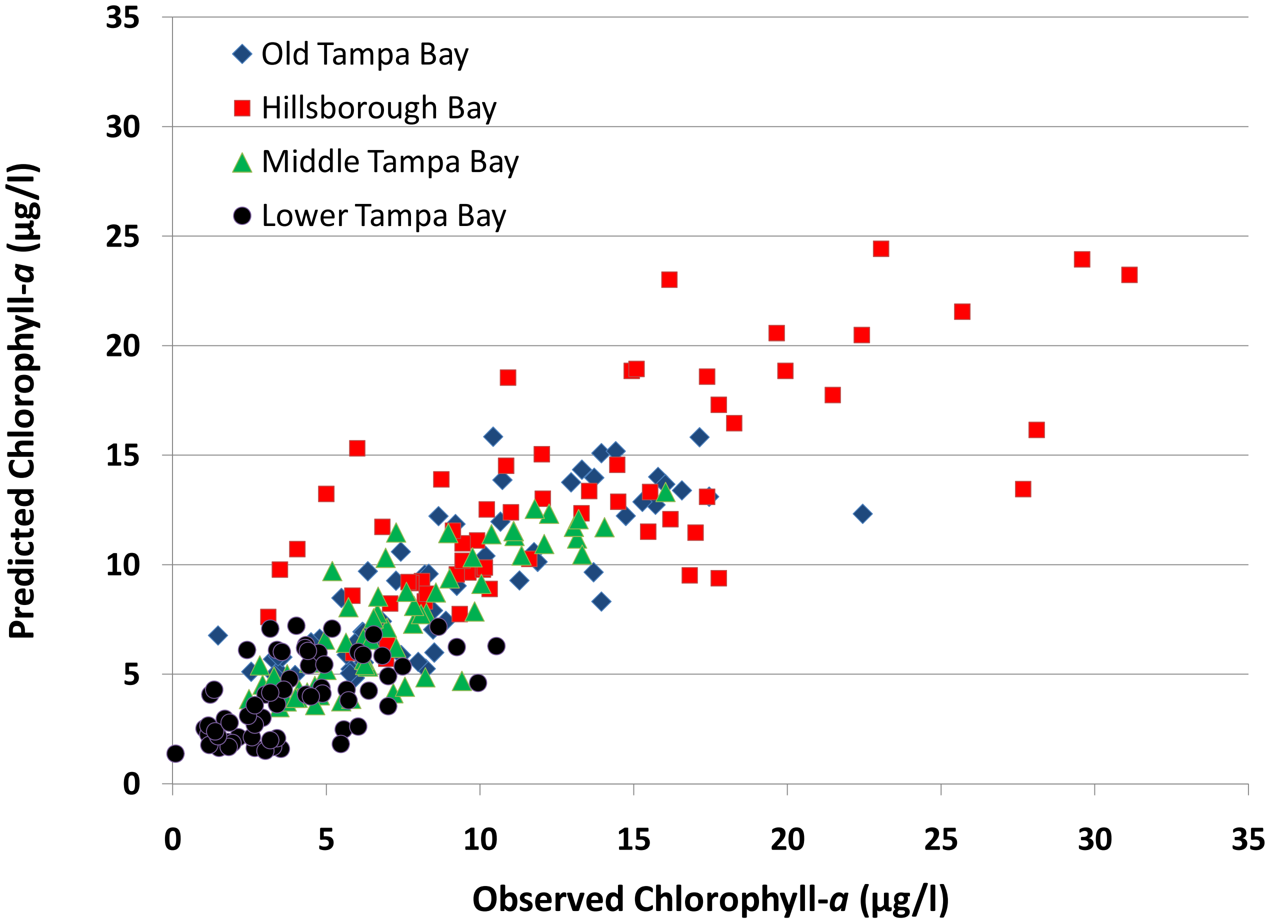

Figure 3-7. Predicted versus observed chlorophyll *a* concentrations in Tampa Bay.

In Sarasota Bay, we have been investigating the relationships between TN concentrations and loadings with chlorophyll *a* concentrations in each of the five bay segments (Janicki et al., 2010). A regression model between chlorophyll *a* and TN concentrations was developed for Blackburn Bay. Residual analysis revealed a seasonal difference in residuals. Specifically, given the same TN concentrations, chlorophyll *a* concentrations can be expected to be higher in Blackburn Bay during the wetter, warmer summer months (July-October) than during the remainder of the year. Therefore, a seasonal term was added to the regression equation. The season term is a dummy variable which equals one during July-October and zero other months of the year. The final regression equation is:

$$C_t = -3.55 + (23.72 * [TN]) + (1.73 * Season)$$

where: C_t = mean chlorophyll a concentration in month *t*

The model was fit with 133 observations and resulted in an R^2 value of 0.73. The regression was highly significant with a probability of a greater |F| value of < 0.0001. The slope and parameter coefficients were also highly significant. A plot of predicted versus observed chlorophyll *a* concentrations is presented in Figure 3-8.

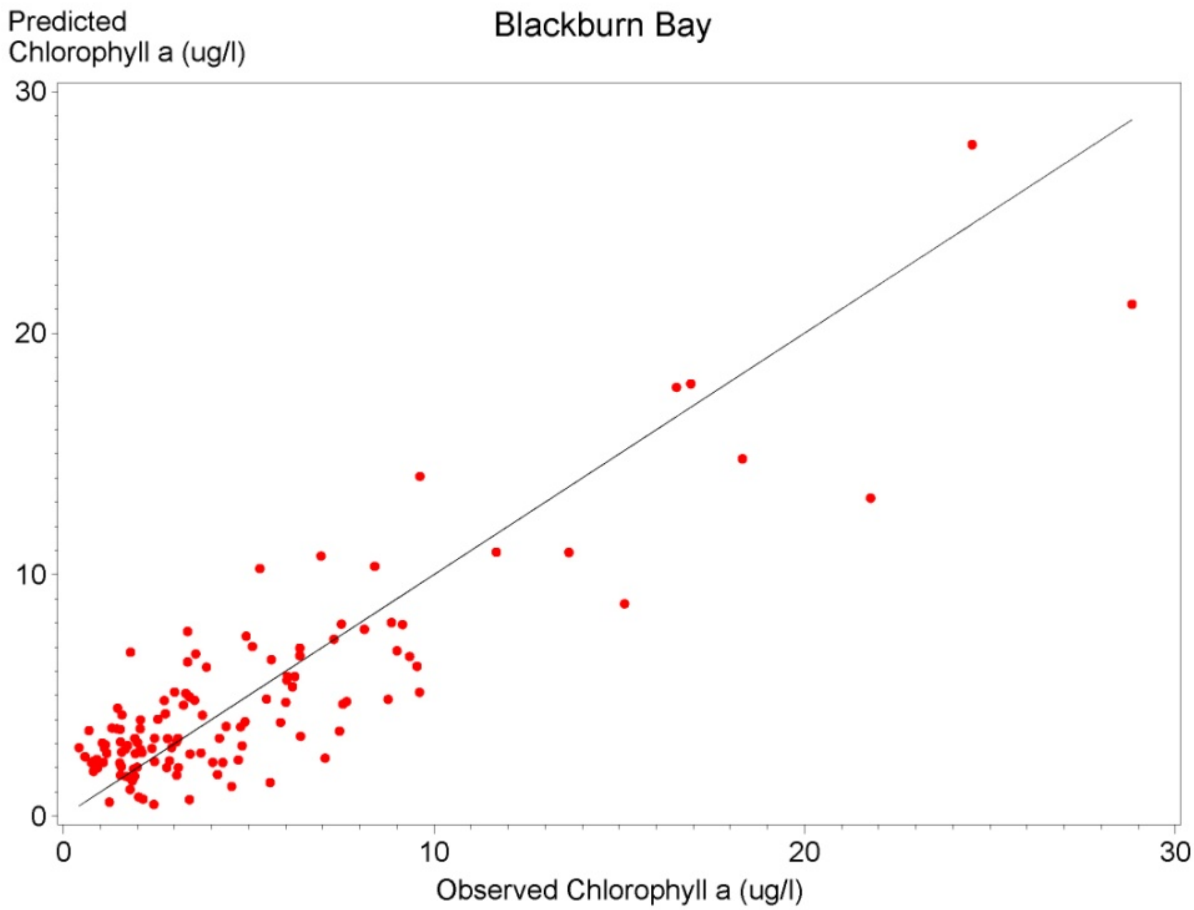

Figure 3-8. Predicted versus observed chlorophyll *a* concentrations in Blackburn Bay ($R^2 = 0.73$).

3.2.5.2 Quantile Regression

Ordinary least squares regression is used to model the relationship between one or more independent variables and the conditional mean of a response (dependent) variable. Unlike ordinary least squares regression, which models the relationship between an independent variable and the mean of the response variable, quantile regression models the relationship between the independent variable and the conditional quantiles of the dependent variable (Koenker and Bassett, 1978). Quantile regression is particularly useful when the rate of change in the conditional quantile, expressed by the regression coefficients, depends on the quantile (Chen, 2005). Therefore, quantile regression can sometimes be useful for informing criterion selection for responses that do not satisfy the assumptions of ordinary least squares regression (USEPA, 2009). Quantile regression makes no distributional assumption about the error term in the model.

As an example, quantile regression was used to develop relationships between log (chlorophyll *a*) and log (TN) for an estuary. This is the same dataset that was used in the linear regression example. The median prediction interval from the quantile regression was similar to the mean response predicted by ordinary least-squares; however, it can be seen that the slope of the 90th percentile is somewhat shallower than that of the 50th percentile (Figure 3-9).

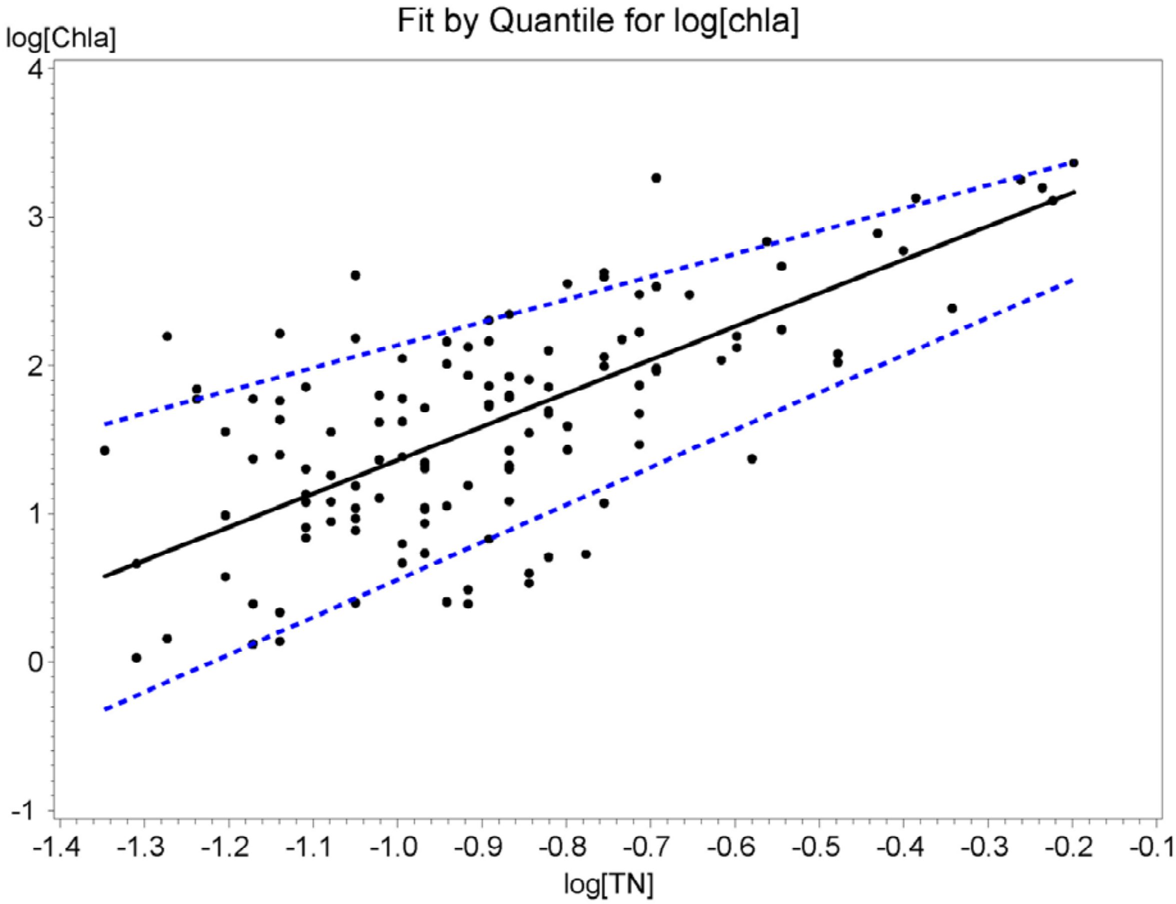

Figure 3-9. Relationship between chlorophyll *a* and TN concentrations as defined by a quantile regression.

3.2.5.3 Logistic Regression

Logistic regression models are used to predict the probability of an event occurring as a function of one or more independent variables. Because the dependent variable in logistic regression is dichotomous, this regression technique lends itself to situations where a water quality measurement results in an exceedance or impairment (e.g., dissolved oxygen < 4 mg/l). In most other respects, logistic regression is similar to ordinary least squares regression. The model can be fit to observed data using maximum likelihood estimation and statistical tests are used to determine if model coefficients differ significantly from zero (USEPA, 2009). Generally, logistic regression is most beneficial when the probability of an event ranges between near zero and near 1 over the range of some independent variable.

Further discussion of logistic regression approaches and an example are provide below in Section 3.2.7.

3.2.5.4 Changepoint Analysis

Changepoint methods are rapidly evolving from simple data mining tools to predictive models using advanced statistical algorithms to evaluate conditional probabilities in the stressor-response relationships over the range of the relationship. These methods are therefore useful to evaluate the stressor-response relationship whether or not a threshold value has been established. Collectively referred to as "Decision Trees", this methodology provides an intuitive and easily conveyed approach to identify threshold responses to environmental stressors that may be used in the development of numeric nutrient criteria. We intend to use a conditional inference tree methodology (Hothorn et al., 2006) as one line of evidence for evaluating stressor-response relationships. Conditional inference trees are a form of regression tree analysis (RTA) that has been successfully used to assist in the development of numeric nutrient criteria (e.g., Soranno et al., 2008). The approach is based on recursive partitioning. The partitioning process iteratively searches for a point in the stressors variable which maximizes the difference in the response values between two groups of response data. No *a priori* threshold is specified. The regression tree approach defines the breakpoint as that which maximizes the difference between groups by minimizing the p value associated with some statistical test. The point in the stressor variable at which the p value is minimized, after adjustment for multiple comparisons, is assigned as the breakpoint defining the split of the of the response variable into 2 groups. Once the first split is made the process continues to test for subsequent splits that are conditional on the first split. Hence, the term "conditional inference" or "conditional probability analysis" that has been popularized recently by the USEPA as a potential approach for establishing numeric nutrient criteria. Soranno et al. (2008) utilized this approach in establishing a framework for developing ecosystem specific numeric nutrient criteria in Michigan lakes and developed a bootstrap methodology with this approach to quantify the uncertainty associated with a particular detected changepoint. Bootstrapping is a powerful tool to quantify uncertainty and fits well with the analytical approach recommended by the SAB in their comment on the USEPA Guidance document regarding characterizing uncertainty in the assessment process.

Conditional inference trees embed tree-structured regression models into a well defined theory of conditional inference procedures (Hothorn et al., 2006). This class of regression trees is applicable to all kinds of regression problems, including nominal, ordinal, numeric, censored as well as multivariate response variables and arbitrary measurement scales of the covariates. These models can be specified to provide information on the strength of the stressor-response relationship, account for covariates in the relationship and model conditional probabilities. Validation techniques are built in functions in the procedures which can be invoked to estimate the predictive power of the resultant models. Below is an illustrative example of the analytical process using conditional inference trees.

Following the example conceptual model provided in Figure 3-5, the hypothesis is established that chlorophyll *a* concentrations are a function of TN and TP concentrations. The conditional inference tree is used to search for statistically significant changepoints in the relationship between TN and chlorophyll *a*. In Figure 3-10, the first split occurs at a monthly TN average value of 0.472 mg/l (node 1). Successive splits are performed conditional that the monthly average TN values are below 0.472 mg/l. The distribution of the response variable is provided in the box plots at the bottom of the figure for each of the terminal nodes. It is easy to see that there is a positive relationship between TN concentrations and chlorophyll *a* responses. That a nonlinear increase in chlorophyll *a* levels appears to occur when the TN concentrations are above 0.472 is information that may be used in a weight-of-evidence approach to develop potential criterion value(s) for TN.

Conditional Inference - Statistical Stopping

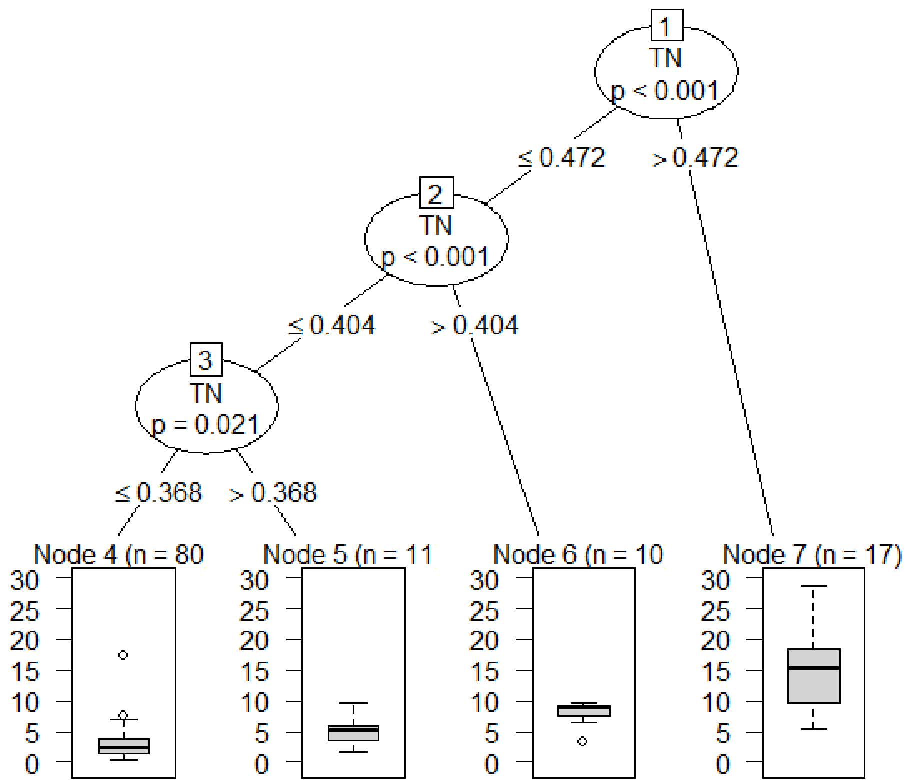

Figure 3-10. Decision tree identifying potential breakpoints between TN and chlorophyll *a* concentrations.

To further examine the relationship between TN and chlorophyll a as part of the weight of evidence approach, it is necessary to consider potential confounding variables as well as other mediating influences in the relationship between TN and chlorophyll *a*. Subsequent to the analysis above, we postulate that seasonality is likely to influence the relationship between TN and chlorophyll a. Hydrologic conditions in southwest Florida are highly seasonal and therefore discharges from the pollutant loadings model were used as a covariate in this example. The hypothesis also considered the potential for co-limitation of TN and TP at some point in the relationship between TN and chlorophyll *a*. Therefore, when phytoplankton populations increase there is the potential for phosphorus to become limiting. For this analysis we used a variant of the conditional inference model that allows for linear model formulation within the terminal nodes of the decision tree. This method is referred to as model based recursive partitioning (MOB). Since a parametric model framework is specified for this assessment the variables were natural log transformed for analysis. In Figure 3-11, the results of the MOB tree are presented. In this case, discharge is the variable that the response is being partitioned on and the relationship between chlorophyll *a* and TN and TP is defined within each of the terminal nodes. This illustration suggests that at discharges above 26 million cubic feet per month a statistically significant relationship was observed and below 26 million cubic feet, there was no relationship. The coefficients for the linear regressions are provided for each terminal node as part of the model output. The coefficients for this example are provided in Table 3-2. Interestingly, at low volumes there appears to be no relationship while at the highest hydrologic volumes there is some evidence of co-limitation of TN and TP.

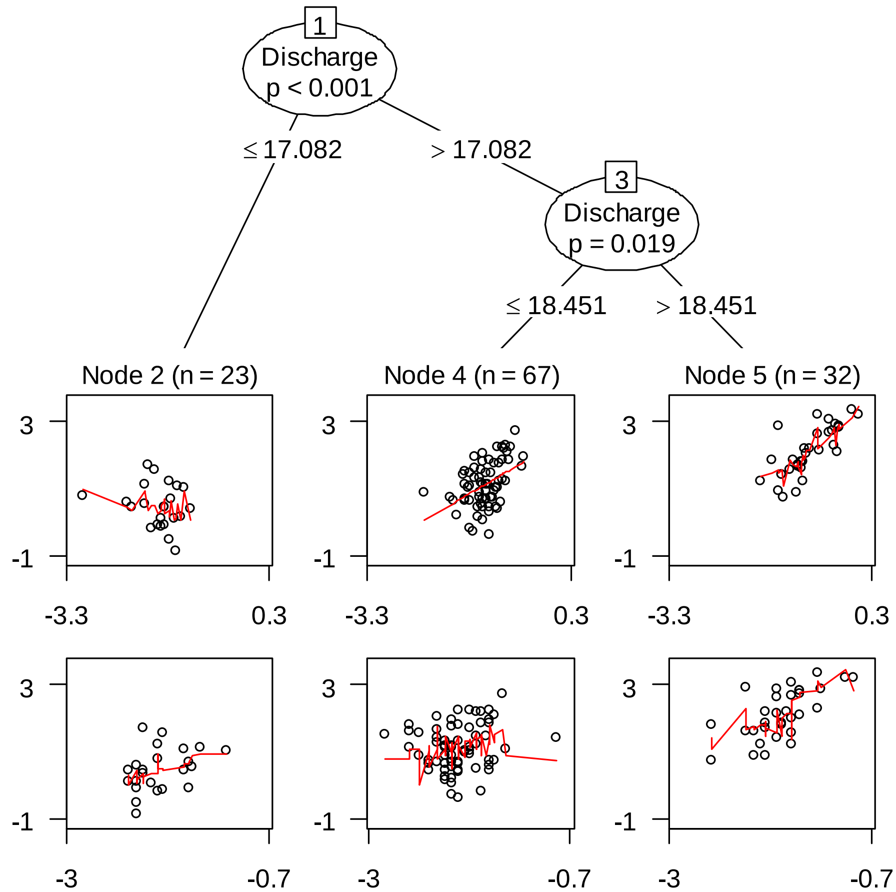

Figure 3-11. Model based recursive partitioning of chlorophyll *a* based on natural log transformed discharge, total nitrogen (top row of scatterplots) and total phosphorus concentrations (bottom row of scatterplots).

Table 3-2. Linear regression coefficients for relationships between chlorophyll *a*, TN and TP partitioned by discharge.

ln(Discharge)	Node	Intercept	ln (TN)	ln(TP)
< 17	2	1.444	-0.395 NS	0.852 NS
17 < 18.5	4	2.322	0.999 **	-0.001 NS
> 18.5	5	4.109	0.983 **	0.703**

It is anticipated that decision tree methods will provide valuable information for establishing numeric nutrient criteria in some cases and not in others. Principally, the results will depend on our ability to account for the many potential confounding variables in the relationship between nutrients and phytoplankton responses. The study area for this project includes a diverse set of estuarine water bodies from lagoonal systems with high residence times and little freshwater input relative to the estuary size to small estuarine waterbodies with large hydrologic inputs and direct influence of tidal passes. Decision tree approaches provide valuable tools for identifying potential stressor-response relationships that can be used to develop numeric nutrient criteria and we intend to fully explore their use in establishing criteria for southwest Florida estuaries. Bayesian methods, bootstrap aggregation and validation techniques are available as part of these methods which we intend to explore to minimize the potential for misspecification of the true relationship between these variable and provide weight of evidence toward the establishment of biologically relevant and meaningful criteria that can serve to protect the designated uses of these estuarine waterbodies.

It is important to note that all of these approaches are likely to result in potential criterion values that are sometimes exceeded. In other words, a changepoint can only be identified if there are values above the changepoint value. This is an important consideration in the implementation of a criterion value which will be further discussed in the implementation section.

3.2.6 Limiting Nutrient and the Need to Establish Criteria for Both N and P

In most estuarine ecosystems N is the most limiting nutrient (a nutrient whose concentration in an organism's environment that determines the growth and productivity of that organism) (Boynton et al., 1982; Howarth, 1988; Chapra, 1997; National Research Council, 2000; Pennock et al., 1999). Aquatic ecosystems are commonly characterized by their N:P ratios. Receiving waters with ratios less than 10:1 (based on concentrations) are considered nitrogen limited, while ratios higher than 10:1 are assumed to be phosphorus limited (FDEP, 2000).

In Tampa Bay, bioassay experiments, empirically-derived nutrient-response relationships, and water quality modeling simulations indicated that controlling nitrogen loads to the bay should be the primary watershed management focus to limit phytoplankton production and allow for improvements in bay water clarity (Janicki and Wade, 1996; Wang et al., 1999; Dixon et al., 2009; Johansson, 2009). These early studies clearly established that nitrogen loads were the limiting nutrient in the Tampa Bay estuary and that phosphorus loadings to the bay from the enriched Bone Valley region were not controlling estuarine production.

In Sarasota Bay, ratios between TN and TP concentrations are typically well below 10 (Lowrey, 1992), which is consistent with presumed nitrogen limitation. In Roberts Bay, FDEP (2005) calculated a median TN:TP ratio of 2.4, indicating strong nitrogen limitation of phytoplankton growth. In addition, empirically derived relationships also point to the primary role of nitrogen in the eutrophication processes in Sarasota Bay (i.e., Tomasko et al., 1992 and 1996), as was found in the earliest studies in Tampa Bay (i.e., Johansson, 1991).

In Charlotte Harbor, the two major freshwater inputs are the Peace and Myakka rivers. Both of these rivers are distinguished by the major phosphate deposits that lie within their watersheds. As a result, it has been generally held that the most of the CHNEP segments are not phosphorus limited. We intend to further these analyses to verify the TN limitation, if it exists.

Given that TP is not the limiting nutrient in these estuaries, but that USEPA wishes to establish criteria for both nitrogen and phosphorus in all waters, a methodology for identifying a TP criterion is needed. An important factor to consider in establishing this methodology is that variation in the TN:TP ratio can result in undesirable changes in phytoplankton community structure, and thus ecosystem function. Given a period when desirable phytoplankton community structure exists, or at least the lack of any undesirable blooms, the ambient water quality conditions (i.e., TN:TP ratios) can serve as an appropriate baseline for use in TP criteria establishment. Following establishment of TN concentration criterion, the TP criterion can be based on the TN:TP ratio observed during the baseline period when desirable phytoplankton community structure existed.

An additional approach to establishing a TP criterion is to follow the approach used to establish the TN criteria in the four Tampa Bay segments. In this case, TP loads from the baseline period (1992-1994) can be estimated and the resultant TP concentrations can be calculated.

We intend to investigate all of these approaches and will develop a recommendation that is based on full evaluation and comparison of the results from application to the empirical data.

3.2.7 Compliance with Dissolved Oxygen Standards

There are two major questions that must be addressed with respect to establishing numeric nutrient criteria that are protective of DO in the southwest Florida estuaries:

- Is there a relationship between nutrient loads, phytoplankton responses and dissolved oxygen levels in these estuaries?
- Do the proposed numeric nutrient criteria protect DO conditions (i.e., achieve DO standards) in these estuaries?

To address these questions we intend to investigate and quantify the relationship between the existing nutrient loadings and chlorophyll *a* targets in these estuaries and the frequency of DO values less than the state standard of an instantaneous 4 mg/l and the 5 mg/l daily average (if data allow).

Initially, we will examine the spatial and temporal variability in DO concentrations in each of the estuarine segments. This will involve synthesizing available data and identifying the spatial and temporal variability in DO within each segment. Maps will be produced for each month across years to display the variability across years within a month as well as the inter-annual variability in DO. Spatial heterogeneity will be examined using the probabilistic dataset and contour plots of bathymetry will be used to the extent that valid bathymetry is available to consider the known relationship between DO levels and depth. The results of this task will be a detailed assessment of the spatial and temporal variation in DO in each estuary.

The relationship between nutrient loads and residence times and chlorophyll *a* and DO responses in each estuarine segment will be examined. These estuaries are heavily influenced by seasonal variability in rainfall with a wet season between mid June and October delivering the bulk of freshwater inflow and nutrients to the system. Water temperatures are also highest during the wet season and contribute to the reduced capacity of estuarine waters to contain oxygen. Therefore, there are both natural and anthropogenic stressors that can potentially influence DO concentrations in these estuaries. To investigate the role of these stressors, the variability in DO concentrations among them will be partitioned. Specifically, we intend to assess if there are quantifiable relationships between nutrient loads and/or concentrations, chlorophyll *a* concentrations, and DO. The quantification of these relationships will need to consider the natural influences that affect and may confound the relationship among these variables. We will explore several analytical methods in an attempt to quantify these relationships.

Initially, simple and multiple regression techniques will be applied to assess if there are quantifiable relationships between nutrient loads and/or concentrations, chlorophyll *a* concentrations, and DO. Given a defensible relationship, expected DO concentrations for any given nutrient load and/or concentration and chlorophyll *a* concentration can be predicted.

Logistic regression techniques can also be used to investigate quantifiable relationships between nutrient loads and/or concentrations, chlorophyll *a* concentrations, and DO. An example of the use of logistic regression on DO data is presented in Figure 3-12. The relationship between DO exceedances and flow was investigated. A general trend of lower DO concentrations at lower flows was documented. However, the ordinary least-squares regression was not sufficient to predict DO exceedances based on flows. Therefore, logistic regression was employed (Figure 3-13) with satisfactory results. Logistic regression can be used with categorical and continuous predictor variables and slope estimates can be translated to describe the change in odds of occurrence per unit change in the predictor variable.

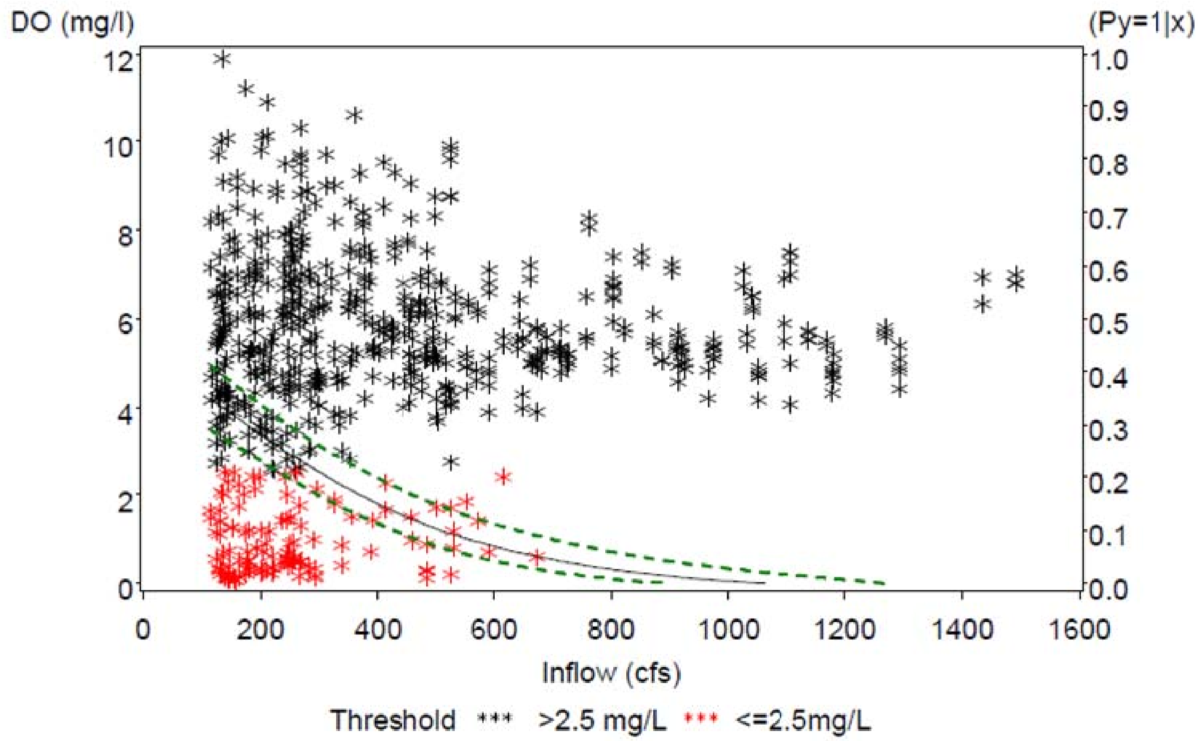

Figure 3-12. Relationship between the probability of DO < 4mg/l and flow as defined by a logistic regression.

Changepoint methods are rapidly evolving from simple data mining tools to predictive models using advanced statistical algorithms to evaluate conditional probabilities in the stressor-response relationships over the range of the relationship. These methods are therefore useful to evaluate the stressor-response relationship whether or not a threshold value has been established. Collectively referred to as "Decision Trees", this methodology provides an intuitive and easily conveyed approach to identify threshold responses to environmental stressors that may be used in the development of numeric nutrient criteria. We intend to use a conditional inference tree methodology (Hothorn et al., 2006) as one line of evidence for evaluating stressor-response relationships. Conditional inference trees are a form of regression tree analysis (RTA) that has been successfully used to assist in the development of numeric nutrient criteria (e.g., Soranno et al., 2008). The approach is based on recursive partitioning. The partitioning process iteratively searches for a point in the stressors variable which maximizes the difference in the response values between two groups of response data. No *a priori* threshold is specified. The regression tree approach defines the breakpoint as that which maximizes the difference between groups by minimizing the p value associated with some statistical test. The point in the stressor variable at which the p value is minimized, after adjustment for multiple comparisons, is assigned as the breakpoint defining the split of the response variable into 2 groups. Once the first split is made the process continues to test for subsequent splits that are conditional on the first split. Hence, the term "conditional inference" or "conditional probability analysis" that has been popularized recently by the USEPA as a potential approach for establishing numeric nutrient criteria. Soranno et al. (2008) utilized this approach in establishing a framework for developing ecosystem specific numeric nutrient criteria in Michigan lakes and developed a bootstrap methodology with this approach to quantify the uncertainty associated with a particular detected changepoint. Bootstrapping is a powerful tool to quantify uncertainty and fits well with the analytical approach recommended by the SAB in their comment on the USEPA Guidance document regarding characterizing uncertainty in the assessment process.

We intend to investigate all of these approaches and will develop a recommendation that is based on full evaluation and comparison of the results from application to the empirical data.

3.3 Downstream Protection Values (DPVs)

Downstream Protection Values (DPVs) for TN and TP are being developed as part of the numeric nutrient criteria effort by USEPA. USEPA defines DPVs as those water quality criteria in flowing waters that ensure protection of designated uses in downstream estuarine waters as required by the CWA under 40 CFR 131.10(b). USEPA (2010) previously proposed TN DPVs based on proposed protective estuarine TN loads, with the DPVs being expressed as concentrations in upstream reaches.

We will recommend a clear definition of DPV prior to deriving DPVs for the estuarine watersheds; the definition of a DPV will include:

- the temporal scales (multi-year, annual, seasonal, monthly),
- the spatial scales (drainage area, stream reach/river segment, tidal creek, bayou),
- the metrics used (average, geometric average, percentile range), and
- the units of measure for the DPV.

Derivation of DPVs must include definition of the boundary between the estuary and the drainage network discharging to the estuary. In its previous proposal of DPVs, USEPA utilized the proposed protective load to the estuary from a drainage network and the model-derived fraction of the load from an upstream basin which reaches the estuary, to derive TN DPVs as concentrations for each stream reach of the model drainage network.

Initially, identification of the terminal segments of the drainage network is necessary prior to developing DPVs. Since some hydrologic and nutrient loads enter the estuary directly from storm drains and waste water treatment plants, the appropriate spatial bounds (how far upstream) and definition of terminal segments must be developed. We will provide a working definition of terminal segments for DPV development, as well as maps depicted the terminal segments and the spatial bounds represented in the development of protective watershed and estuarine criteria.

There are several potential methods to develop DPVs for the southwest Florida estuaries.

- If the numeric criterion for the receiving estuary is expressed as a concentration, then set the DPVs for the terminal segments to the numeric nutrient criteria for the estuary.
- If the criterion is expressed as a load, then the terminal segment DPV could be estimated by taking into account the relative hydrologic contribution from the terminal segment to the overall hydrologic load to the estuary.
- If the numeric criterion for the receiving estuary is expressed as a concentration, then the terminal segment DPV could be derived from the relationship between the nutrient concentrations in the terminal segments and those in the receiving estuary. However, this method is dependent upon having water quality data available for both the downstream estuarine segment and the terminal segment, thus this method may not be applicable to all terminal segments.

We will investigate the above potential methods to derive DPVs for the terminal segments to each estuary.

USEPA also intends to define DPVs for all upstream stream reaches within the watersheds that drain to the estuaries in Florida. As for terminal segment DPVs, the definition of upstream segments will have to be developed prior to deriving upstream segment DPVs. Upstream segments may be defined as stream reaches, subbasins, or some other areal unit. After definition of upstream segments, we will define an approach for translating terminal segment DPVs to upstream segment DPVs.

It is likely that this approach will follow similar logic that was employed by USEPA in the proposed DPVs (USEPA, 2010). This logic assumed that given an estuarine load limit, the watershed load portion of the limit can be assigned to contributing areas in the watershed through a mechanism that considered nutrient inputs from the stream network. Although there were significant issues with the mechanism used by USEPA to assign loads to upstream reaches and the methods for developing the proposed loading limit for the estuary, we do find the guiding logic used by USEPA to be appropriate.

We have developed a methodology to partition the watershed load to an estuary over its watershed subbasins, so that we have the loads reaching the estuary from each subbasin. We will assess whether load attenuation should be considered in developing the upstream DPVs, and if so, how to most appropriately quantify this attenuation. In the case where attenuation is included in the DPV methodology, we will consider whether to utilize first-order decay, or whether it is more appropriate to utilize mechanistic modeling to derive factors for load attenuation from upstream segments to the estuary.

We will include an evaluation of relationships between ambient concentrations in the upstream segments and those in the downstream terminal segments. It may be that significant relationships may be found between these concentrations, so that the DPV of the downstream terminal segment can be used to derive DPVs for the upstream segments. However, this method is dependent upon having water quality data available from both the terminal segment and the upstream segment, so this method may not be applicable to all upstream segments.

3.4 Designated Uses

The primary objective in the establishment of numeric nutrient criteria is to assure that designated uses of the estuarine waters are protected. To this end, the criteria must provide fully aquatic life support in these waters. The two primary responses that will be investigated (i.e., chlorophyll *a* and DO) provide important insight into the extent to which aquatic life support is achieved.

Chlorophyll *a* and the resultant water clarity are major determinants of the seagrass success in these estuaries. The protection of seagrasses is a critical element of assuring the aquatic life support in the southwest Florida estuaries. Seagrasses serve several significant functions within estuaries, which are degraded if seagrass coverage declines. Seagrasses provide food, shelter, and essential nursery habitats for many recreationally and commercially valuable species of fish, crustaceans, and shellfish (Dawes et al., 2004). They also provide food and habitat for marine mammals, such as manatees, listed species, and waterfowl. Seagrasses also help maintain water clarity by trapping fine sediments and particles with their leaves and stabilizing the estuarine sediments with their roots (Fonseca, 1989; Short et al., 2000). Seagrasses are very effective at removing dissolved nutrients from water that can enter from land runoff. The removal of sediment and nutrients improves water clarity, thereby improving overall ecosystem health.

The effect of reduced DO (hypoxia) on biota is well-known (Diaz and Rosenberg, 1995; USEPA, 2000c). Therefore, achieving aquatic life support is also significantly influenced by estuarine DO conditions.

To examine further whether full aquatic life support, and thereby designated uses, is achieved by the proposed numeric nutrient criteria, additional data sources will be investigated. These include:

- Tampa Bay
  - Biennial aerial surveys of seagrass cover (Southwest Florida Water Management District);
  - Benthic abundance and community structure data from an annual probabilistic sampling during a late summer index period (TBEP and local counties);
  - Fish abundance and community structure from a probabilistic sampling conducted monthly (Florida Fish and Wildlife Conservation Commission); and
  - Annual bird count data (Florida Audubon).

- Sarasota Bay
  - Biennial aerial surveys of seagrass cover (Southwest Florida Water Management District);
  - Benthic abundance and community structure data from previous studies;
  - Fish abundance and community structure from a monthly probabilistic sampling conducted (Florida Fish and Wildlife Conservation Commission);
  - Annual bird count data (Florida Audubon);
  - Oyster bed surveys (Sarasota County); and
  - Macroalgae surveys (Sarasota County).

- Charlotte Harbor
  - Biennial aerial surveys of seagrass cover (Southwest Florida Water Management District, South Florida Water Management District, SCCF);
  - Benthic abundance and community structure data from previous studies;
  - Fish abundance and community structure from a monthly probabilistic sampling conducted (Florida Fish and Wildlife Conservation Commission);
  - Juvenile fish survey (FGCU);
  - Annual bird count data (Florida Audubon);
  - Oyster bed surveys (Sarasota County, FGCU);
  - Macroalgae surveys (Sarasota County);
  - Bird Rookery monitoring (FDEP);
  - Seagrass Transects (FDEP); and
  - Scallop Surveys (Florida Fish and Wildlife Conservation Commission).

We intend to provide at least a qualitative assessment of these data sources relative to the recommended numeric nutrient criteria. Where data allow, a more quantitative assessment will be attempted. For example, the abundance of species that are supported by the phytoplankton primary production will be related to the proposed nutrient criteria in the various estuaries.

4.0 Implementation

In following the analytical flow path described in Figure 3-2, analytical outcomes that are relevant to the developed conceptual models will be evaluated against the wealth of existing information that has been documented for these estuaries. Existing information will also be utilized to generate potential numeric nutrient criteria. Once potential criteria are established, there will be several important issues to assess to determine the appropriate implementation of the potential criteria. These issues include:

- identifying the appropriate temporal scale for assessment,
- considering the effects of seasonality on the assessment,
- characterizing the uncertainty of potential criterion values, and
- practical considerations.

4.1 Temporal Scales and Seasonality

The analysis of stressor-response relationships may include data averaged over various temporal scales including; individual samples, monthly averages, seasonal averages or annual averages, among others. The distribution of samples may also be used for analysis rather than a summary statistic of the distribution. However, it is recognized that the regulatory assessment protocol used by the USEPA and FDEP will likely be on an annual basis. The USEPA has previously developed criteria based on exceedance of annual geometric means (USEPA, 2010). The FDEP evaluation for exceedance of their DO criterion is based on the proportion of exceedances (i.e., 0.10 or 10%) of a state standard using the binomial distribution. These evaluation methods are very different, the binomial approach accounts for the distribution of values within a year, while the annual geometric mean does not.

We intend to utilize several methods to determine a biologically-significant exceedance frequency of the potential criteria for a given waterbody. For example, a changepoint identified for TN is likely to be exceeded some proportion of the time within a year or at least in some years. If the criterion defined as the changepoint is taken for use as an annual average, then it will be important to consider how the waterbody under study will be affected by this decision and whether the exceedance of this changepoint over a particular temporal scale results in an adverse biological response within the waterbody. Our approach to examining this issue is described in the section on characterizing uncertainty in the proposed criterion.

Seasonality is an important yet often overlooked factor to consider when estimating water quality relationships using empirical methods. Seasonal variations in temperature and photoperiod influence the ability of phytoplankton to uptake nutrients, grow and reproduce. Hydrologic regimes also generally have a seasonal signal that influences estuarine residence times which in turn affects phytoplankton productivity. In southwest Florida, rainfall and hydrology exhibit highly seasonal patterns with a wet summer season, a moderate winter season with respect to both temperature and rainfall, and typically a very dry spring between March and mid-June.

Empirical assessments of water quality relationships vary in terms of their temporal scale of inference. Annual averages are sometimes used but obfuscate any seasonal signal in the data. More commonly, water quality data are averaged based on monthly observations. In these cases, accounting for seasonality will likely be beneficial in reducing the error variance in statistical modeling techniques. Seasonality is likely to be expressed in statistical modeling either through the direct incorporation of a temperature, photoperiod term (which are often collinear) or more commonly through the use of surrogate categorical terms that estimate the differences in mean response based on the average differences due to the seasonal patterns. A hypothetical example is given below to illustrate this effect. In this example, a theoretical relationship is established between TN and chlorophyll *a* concentrations. The predictive regression equation is:

$$y_i = (14.74 * [TN]) + (1.47 * Season) - 3.60 + e_i$$

In this regression equation the season variable is dummy coded such that Winter is set to 0 and Summer is set to 1. Therefore, when a value is taken during summer, the season effect is a 1.47 μg/l shift in the mean response relative to winter which captures the effects that increased temperatures and photoperiods have on the background chlorophyll *a* concentrations. Sometimes it is also necessary to incorporate an interaction term not only to estimate the shift in the mean response of chlorophyll *a* as a function of season but also to account for the rate of change in response of chlorophyll to TN concentrations within a season. We intend to incorporate seasonality into our empirical assessments because we know that seasonality influences the relationships between nutrients, and chlorophyll *a* and DO responses. The parameter estimates and associated measures of uncertainty can also be incorporated into assessment methodologies to allow for these effects to be incorporated into the criteria evaluation process.

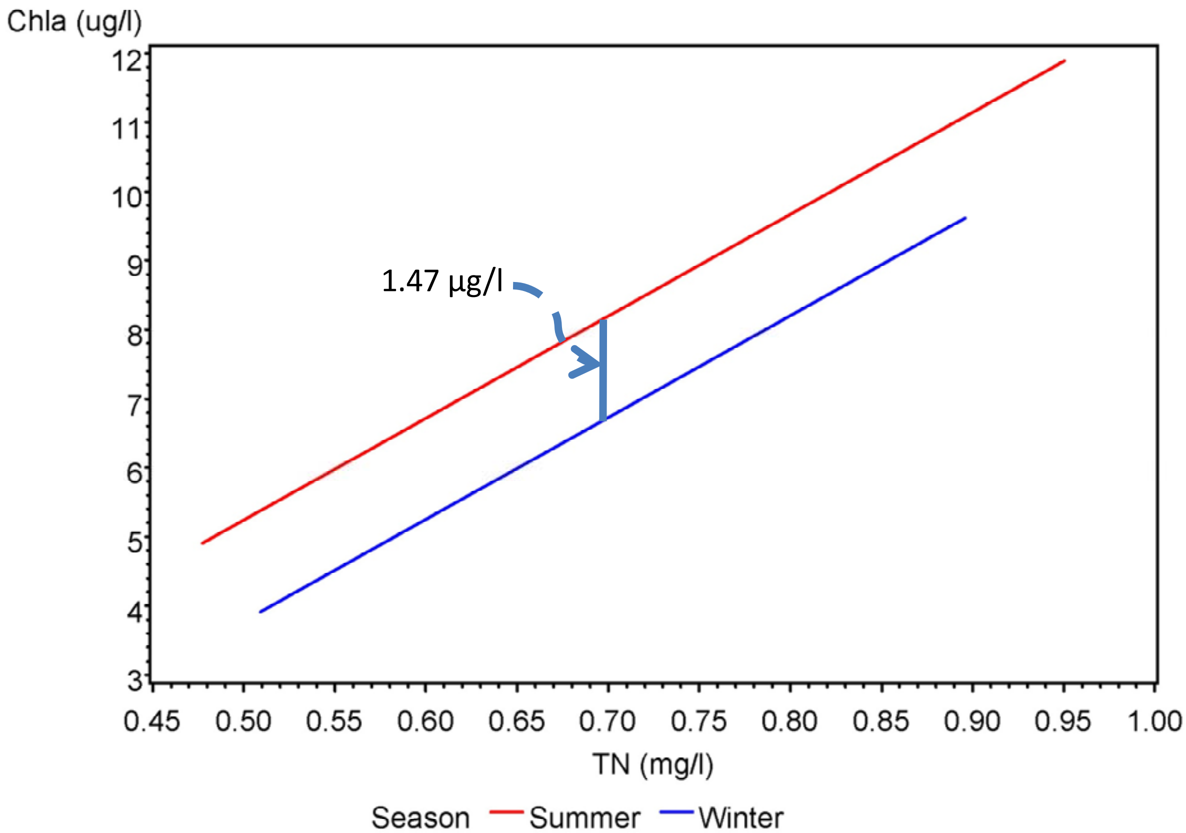

Figure 4-1. Hypothetical regression between TN and chlorophyll *a* with a seasonal effect.

4.2 Characterizing Uncertainty

The importance of characterizing uncertainty was explicitly stated in the SAB review. We intend to use parametric and nonparametric simulation methods to assess the effects of criterion specification. During the analytical approach step, bootstrap methods will have been explored to estimate the most likely changepoint value and characterize the range of potential changepoint values that best define the true relationship between stressor and response. Where parametric models are used during the analytical phase, Monte Carlo methods will be investigated as a means to evaluate the uncertainty of the parametric model and its effects on potential outcomes. Nonparametric simulation methods based on the empirical distribution will also be employed to understand how the expected distribution of values for a particular waterbody will behave in relationship to the specified criterion value. These efforts will provide valuable information on the applicability of potential criterion to serve as an appropriate standard to judge the waterbodies of interest.

Misspecification of the numeric nutrient criteria would likely result in undue burdens on local resource managers. Misspecification can result in either falsely declaring a waterbody as not meeting its designated uses (Type I error) or alternatively, in declaring that the waterbody is meeting its designated uses when it truly is not (Type II error). These simulation methods can be used to minimize the potential for misspecification of the criteria by evaluating scenarios in which the distributional aspects of the data can be adjusted and the effects assessed. An example is provided below to illustrate the use of a simple computer simulation to test the effects of establishing a criterion value and implementing an exceedance rule such as the "one in three" rule that has been proposed for evaluating compliance with numeric nutrient criteria in Florida freshwater streams.

Consider the following example of a TP distribution with a geometric mean of 0.739 mg/l (i.e., the USEPA proposed criterion for TP in the Bone Valley region of west central Florida). We impose a theoretically estimated standard deviation of 0.15 mg/l on this distribution implying that 1) there is heterogeneity in the population of TP values within the Bone Valley region and 2) that there is uncertainty in the exact value that represents the true population mean. USEPA has proposed that the criterion value should be exceeded only once every three years. We conducted a simulation by building a simulated data pool of 10,000 TP observations from streams in the Bone Valley region with a known distribution based on the mean and standard deviation (Figure 4-2).

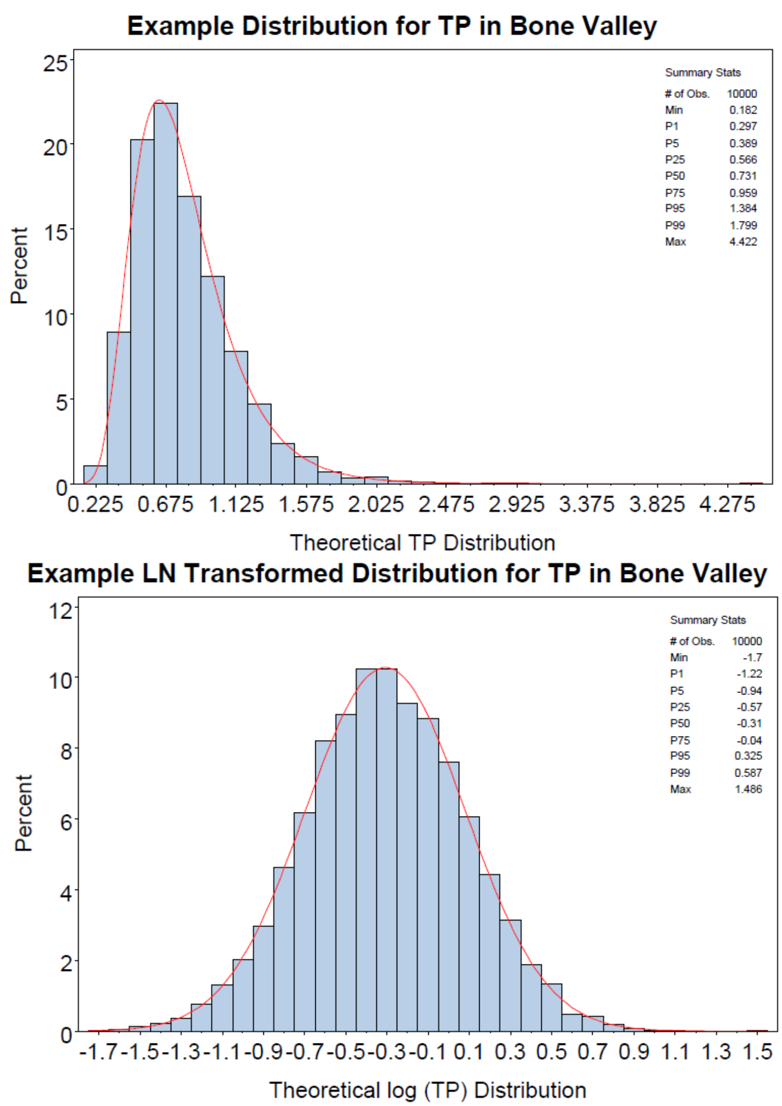

Figure 4-2. Hypothetical total phosphorus distribution of 10000 simulated observations with a geometric mean (i.e., 0.739 mg/l) based on EPA Bone Valley criterion for streams and a standard deviation of 0.15 mg/l. Distribution is show as lognormal (Top) and log transformed (Bottom). Descriptive statistics are inset.

We then pulled at random 12 observations from this pool and calculated a mean of the natural log transformed values and exponentiated those mean values to derive the geometric mean. Each trial outcome then represents the expected geometric mean for a given year. We performed 1000 trials to generate an expected distribution of annual geometric means. The true geometric mean is 0.739 mg/l (or a very close approximation depending purely on randomness). Therefore, we can calculate the number of times out of 1000 when the annual geometric mean would exceed the known (or true) geometric mean. Results suggest that when the true mean is 0.739 mg/l, and 12 observations are taken each year, the annual geometric mean will exceed the true mean nearly 50% of the time by chance. This means that the "one in three years" rule is likely to result in falsely declaring a stream reach to be impaired when in actuality it meets the criterion value.

Nonparametric confidence intervals can be constructed using simulation that would allow for uncertainty to be characterized and used to set more meaningful criteria that minimize the chances for Type I errors that might result in undue burdens on local resource managers and a misallocation of resources toward the proper stewardship of these waterbodies. These methods are robust in simulating a wide array of potential scenario's and evaluating possible outcomes which informs decision makers about risk; an important consideration in the implementation process. These methods are similar to those used by FDEP to assess compliance with the State DO standard. We intend to use simulation methods to characterize uncertainty in the stressor-response relationship, gain insights into how sampling frequencies and sampling designs may affect the evaluation of the potential criterion values and establish numeric nutrient criteria that capture the underlying uncertainties inherit in the data collection and evaluation process.

4.3 Compliance Assessment

Effective assessment of compliance with the established numeric nutrient criteria will depend upon a firm understanding of the importance of spatial and temporal variability and uncertainty in the stressor-response relationships. Also, consideration of the magnitude, frequency, and duration of any deviations from established criteria is critical especially in the context of evaluating biologically-relevant responses to these deviations. The occurrence of highly infrequent, but very significant catastrophic events (e.g., hurricanes, spills) complicates the implementation of the numeric nutrient criteria. Therefore, the methods for compliance assessment must also recognize these issues. An example of an assessment methodology used within the Tampa Bay estuary that attempts to take these issues into account is described below.

4.3.1 Consideration of the Frequency and Duration of Exceedances of Criteria

TBEP developed an assessment methodology that takes the issues discussed above into account. The Technical Advisory Committee (TAC) and Management Board of the TBEP have adopted the Decision Matrix process for tracking the status of chlorophyll *a* and light attenuation in the four mainstem Tampa Bay segments. Water quality targets have been set by the TBEP commensurate with seagrass protection and recovery in the Bay, and the Decision Matrix process provides a method to determine if the seagrass goals and water quality targets are being achieved on an annual basis (Janicki Environmental, 2000).

The Modeling Subcommittee of the TBEP TAC formalized the concepts of *magnitude* and *duration* of differences between observations and targets, as these were determined to be important considerations in determining the appropriate management response level to any target exceedances. Various levels of management response were anticipated, based on four potential outcomes for both chlorophyll *a* concentrations and light attenuation. Figure 4.3 provides the decision framework for chlorophyll *a* assessment, with the various outcomes resulting from combinations of magnitude and duration of exceedances. A similar decision framework is employed for evaluation of light attenuation.

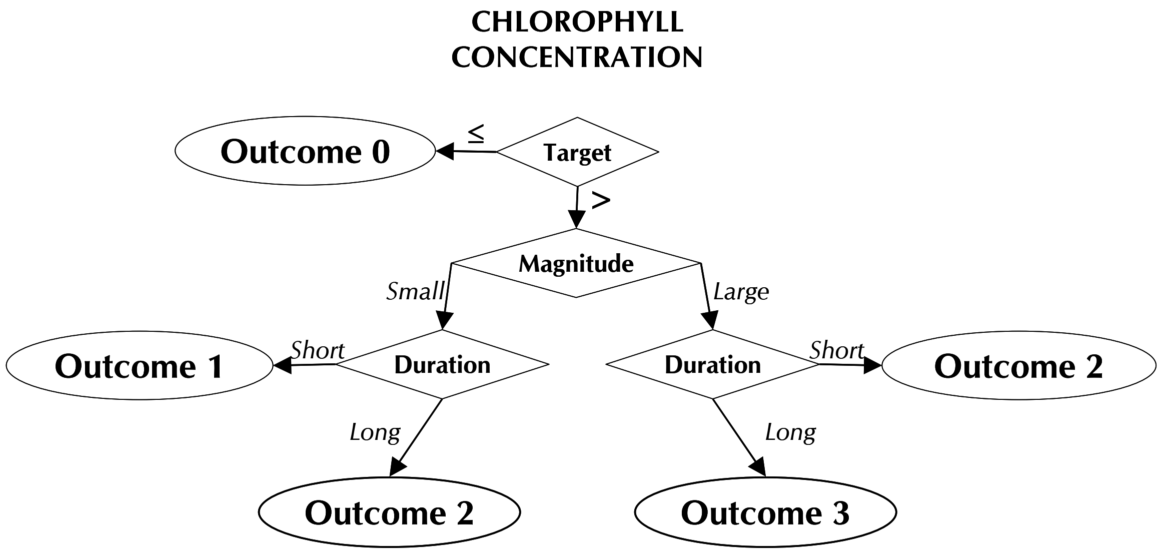

Figure 4-3. TBEP Decision Matrix framework for chlorophyll *a* (from Janicki Environmental, 2000).

Exceedances of small magnitude were defined as those more than one standard error above the segment-specific targets, while large magnitude exceedances were those more than two standard errors above the targets. Short duration exceedances where defined as those less than four years in extent, with any exceedances lasting four years or more defined as long duration exceedances (Janicki Environmental, 2000).

A method of combining the annual results from both the chlorophyll *a* and light attenuation decision frameworks was developed which determined the level of management activity recommended in response to any specific year's evaluation results. The Decision Matrix is shown in Figure 4-4, with the table cells color coded as indications of the degree of response recommended. The recommended management actions resulting from the decision matrix are classified by color into three categories, as follows:

- **GREEN** - "Stay the course"; partners continue with planned projects to implement the CCMP. Data summary and reporting via the Baywide Environmental Monitoring Report and annual assessment and progress reports.
- **YELLOW -** TAC and Management Board on caution alert; review monitoring data and loading estimates; attempt to identify causes of target exceedances; TAC report to Management Board on findings and recommended responses if needed.
- **RED -** TAC, Management and Policy Boards on alert; review and report by TAC to Management Board on recommended types of responses. Management and Policy Boards take appropriate actions to get the program back on track.

Decision matrix identifying appropriate categories of management actions in response to various outcomes of the monitoring and assessment of chlorophyll *a* and light attenuation data.				
CHLOROPHYLL	**LIGHT ATTENUATION**			
	Outcome 0	**Outcome 1**	**Outcome 2**	**Outcome 3**
Outcome 0	*GREEN*	*YELLOW*	*YELLOW*	*YELLOW*
Outcome 1	*YELLOW*	*YELLOW*	*YELLOW*	*RED*
Outcome 2	*YELLOW*	*YELLOW*	*RED*	*RED*
Outcome 3	*YELLOW*	*RED*	*RED*	*RED*

Figure 4-4. TBEP Decision Matrix. (from Janicki Environmental, 2000).

4.3.2 Consideration of Catastrophic Events and *Karenia brevis* Blooms

Southwest Florida has historically been subject to occasionally extreme events including hurricanes, blooms of the red tide causing organism *Karenia brevis,* and unintended releases of nutrient rich sources (i.e., spills). The impact of these episodic natural and anthropogenic events will likely impact water quality and should be somehow accounted for in the evaluation process for determining impairment of southwest Florida estuaries. While it is not possible to predict accurately the timing or influence of these events, an evaluation of the frequency of previous events when water quality data were available will lend insights into how these events might impact future water quality in terms of frequency and duration. We intend to explore these effects via empirical analysis and existing literature on the subject.

4.3.2.1 Catastrophic Events

Irrespective of the cause of these impacts, the evaluation process should consider how to handle these events in evaluation of a waterbody. Short-term water quality exceedances are likely event driven, while long-term exceedances are likely more indicative of chronic conditions. Time series analysis was recommended for consideration by the SAB and we intend to investigate how time series analysis might be used in the impairment evaluation process when catastrophic events occur and may produce responses within an estuarine segment outside the expected time series trend. If eutrophication is defined as a rate (Nixon, 1995), then time series analysis may be a valuable component in determining the fate of a waterbody with respect to its trophic status. Further, if a waterbody is meeting its designated uses and there is no evidence that the waterbody is degrading based on time series trend analysis, should a nutrient criterion exceedance solely caused by a catastrophic event necessarily result in the same management response simply based on a nutrient criterion exceedance?

We also intend to use the available data to assess the responses in various estuarine segments that have been impacted by a catastrophic event. Specifically, we will investigate the lag in responses as well as the time to recovery from these events within each estuarine segment. Clearly, these responses will vary across segments and the magnitude of the catastrophic event. Results from these analyses will be useful in making recommendations regarding the methodology to be used in accounting for the effects of these events in the overall assessment of compliance with the established numeric nutrient criteria.

4.3.2.2 *Karenia brevis* Blooms

Alcock (2007) has provided a significant review of the cause, consequences, and management strategies with respect to red tide blooms (e.g., *Karenia brevis)* along the West Florida Shelf. He has summarized his work and those of others into several key points. These include:

- The initiation and development of blooms occur offshore and in deeper waters.
- Upon inshore movement via wind and ocean currents, these blooms may benefit from nutrient inputs from the adjacent coastal lands.
- The predominant nutrient sources that initiate and support a bloom likely change over the course of the bloom.
- It is uncertain whether any particular combination of nutrient sources is common to all blooms.
- Conclusive evidence of a strong linkage between red tide blooms and the land-based nutrient inputs "remains elusive".

While the precise relationship between land-based nutrient inputs and red tide blooms remains uncertain, Alcock concluded that "caution against dismissing it (coastal pollution) as inconsequential." However, given the lack of conclusive evidence addressing how red tide blooms that enter the southwest Florida estuaries are specifically related to nutrient loading from their watersheds and ambient nutrient conditions, using red tide blooms to support numeric criteria development within the estuary is difficult. Clearly, the management of nutrient loading, as has been evidenced in Tampa Bay, Sarasota Bay, and Charlotte Harbor, not only benefits the longer-term conditions in these estuaries but also contributes to the reduced likelihood of extended bloom conditions when they do occur and move inshore from the Gulf of Mexico.

4.3.2.3 Other Considerations

As discussed above, the implementation of the ultimately established nutrient criteria will require facing a number of significant hurdles. The proposed numeric nutrient criteria should also be considered within the context of expected future data collection efforts to ensure that the evaluation process remains relevant and appropriate to the data sources being assessed. Clear and reasonable expectations with regard to future data needs to assess compliance effectively should be expressed. We intend to offer specific recommendations for those data needs for each of the three estuarine systems.

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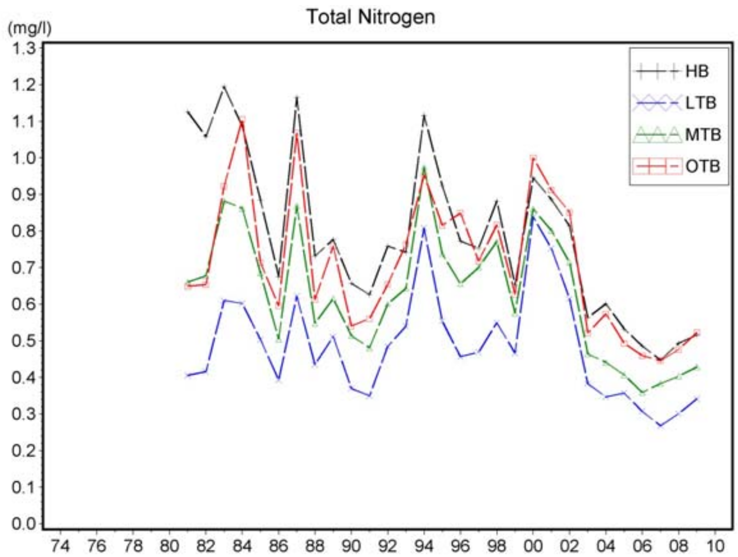
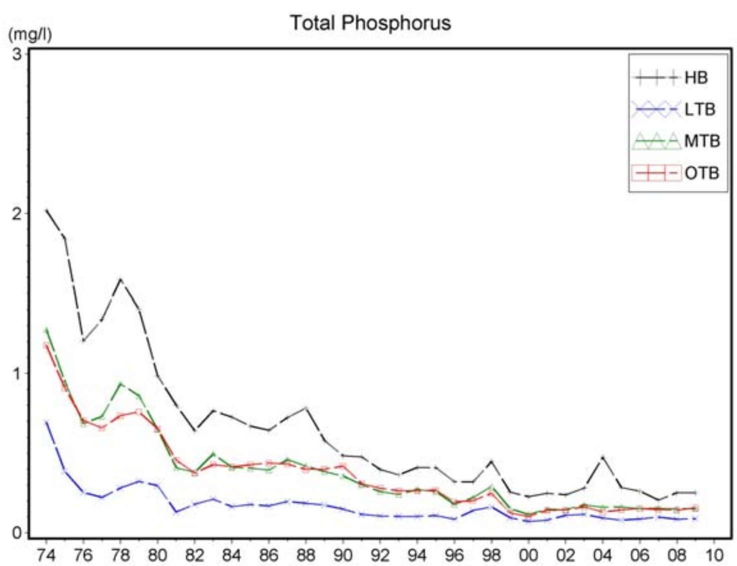
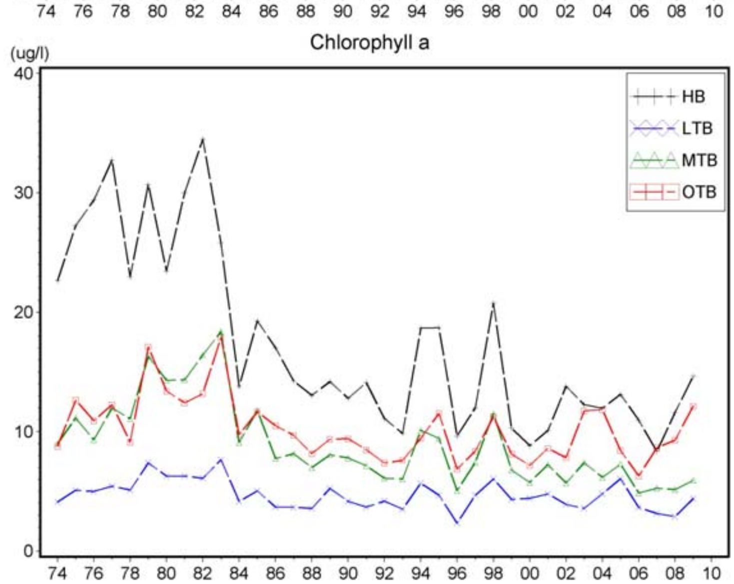
APPENDIX 1
Tampa Bay Segment-Specific Water Quality
Annual TN, TP, and Chlorophyll a Concentrations


Total Nitrogen
(mg/l)
HB
LTB
MTB
OTB


Total Phosphorus
(mg/l)
HB
LTB
MTB
OTB


Chlorophyll a
(ug/l)
HB
LTB
MTB
OTB

APPENDIX 2
Red Tide Monitoring Program

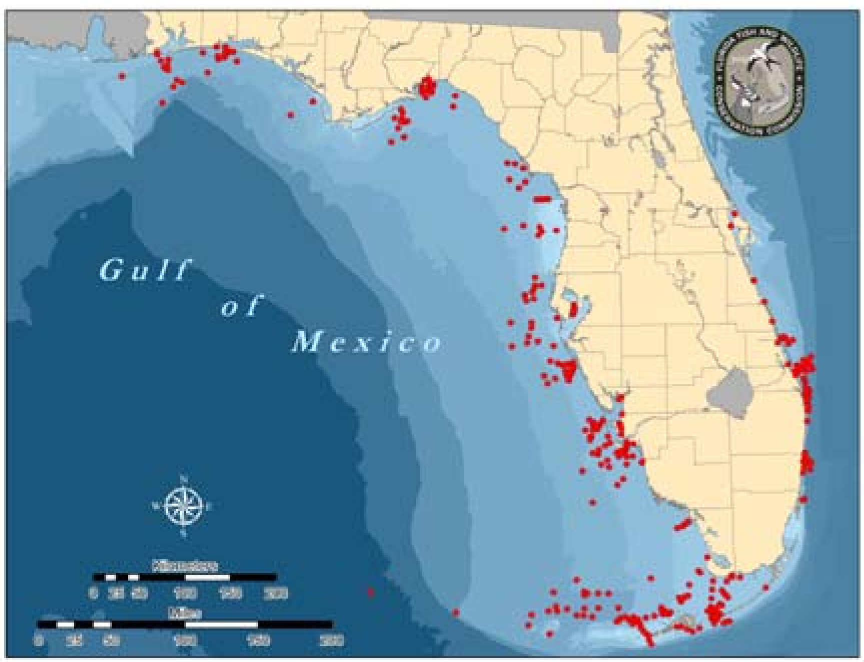
Red Tide Offshore Monitoring Program

Volunteers collect water samples to help Fish and Wildlife Research Institute scientists monitor red tides in Florida waters. Learn about the Red Tide Offshore Monitoring Program and how to become a volunteer.

BACKGROUND AND DESCRIPTION

The Florida Fish and Wildlife Research Institute's (FWRI) Red Tide Offshore Monitoring Program (RTOMP) was established in 2000. The initial purpose of the program was to help FWRI scientists monitor and detect harmful algal blooms (HABs) in Florida by asking volunteers to collect offshore water samples. The RTOMP has grown over the years and now provides increased coverage of the Gulf of Mexico, early warning of offshore algal blooms, red tide event response, and partnerships with the marine community.

Because of limited state personnel, boats, and resources, the RTOMP relies on volunteers of all kinds—charter boat captains, commercial fishermen, private citizens, divers, and more—to collect water samples from offshore areas by boat. Volunteers are located in all areas of Florida. Most samples are collected from the Gulf of Mexico, as shown in this map of sampling activity in 2008.



Sampling Activity in 2009

Once the samples are returned to FWRI, they are examined under a microscope. *Karenia brevis* (the cell that causes Florida's red tide) and other HAB species are identified and counted. The data are entered into the HAB Historical Database and are reported back to the volunteers. Data collected by volunteers are used in conjunction with statewide monitoring results to determine red tide status.

he amount of time volunteers commit to sampling and red tide event response is invaluable. In 2007, volunteers collected more than 500 water samples and spent 197 days collecting those samples in response to a red tide in the northwest and southwest regions of the state. In 2008, volunteers collected 479 water samples over 210 days.