EPA-SAB-11-xxx

The Honorable Lisa P. Jackson
Administrator
U.S. Environmental Protection Agency
1200 Pennsylvania Avenue, N.W.
Washington, D.C. 20460

Subject: Review of EPA's draft Methods and Approaches for Deriving Numeric Criteria for Nitrogen/Phosphorus Pollution in Florida's Estuaries, Coastal Waters, and Southern Inland Flowing Waters

Dear Administrator Jackson:

[TO BE DEVELOPED]

Sincerely,

Dr. Deborah L. Swackhamer, Chair
Science Advisory Board

Dr. Judith L. Meyer, Chair
Nutrient Criteria Review Panel

Enclosure

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1. EXECUTIVE SUMMARY

- TO BE DEVELOPED

2. INTRODUCTION

2.1. Background

Nitrogen (N) and phosphorus (P) inputs from urban and agricultural sources are known to influence water quality, and nutrient pollution has been identified as the source of impairment for estuarine, marine and fresh waters in Florida. The state of Florida has a narrative criterion for nutrients, and is in the process of developing numeric nutrient criteria for its estuaries and coastal waters. In 2009, EPA determined that numeric criteria were needed to protect aquatic life in Florida, and initiated a process to develop such criteria for categories of state waters. Criteria for total nitrogen (TN), total phosphorus (TP) and chlorophyll *a* (Chl-*a*)—a measure of water column algal abundance—were finalized for Florida lakes and inland flowing waters in 2010. Numeric nutrient criteria for estuarine and coastal waters, and South Florida inland flowing waters, are being developed separately, using a variety of approaches and ecological endpoints. The SAB was asked to provide review and advice on the proposed approaches for estuarine, coastal and South Florida waters, as described in the draft EPA document, *Methods and Approaches for Deriving Numeric Criteria for Nitrogen/Phosphorus Pollution in Florida's Estuaries, Coastal Waters, and Southern Inland Flowing Waters* (November 17, 2010 draft; U.S. EPA 2010).

An ad hoc panel of the SAB, the Nutrient Criteria Review Panel, was formed for this task. The Panel met on December 13-14, 2010 to hear EPA technical presentations and public comments, and to discuss responses to the questions in the Charge to the Panel (Appendix A). A follow-up public teleconference of the Panel was held on February 7, 2011 to discuss an initial panel draft report….

2.2. Charge to the Panel

The Charge to the Panel included questions about the conceptual model used to select assessment endpoints, data sources for the various categories of waters, and possible approaches to define criteria for each of the categories of waters: estuaries, coastal waters out to three miles, inland flowing waters (including canals) in South Florida, and South Florida marine waters. Relevant charge questions are included at the beginning of each section of the Panel's report, and the full Charge to the Panel is included as Appendix A.

3. Response to Charge Questions

3.1. Conceptual Approach

3.1.1. Conceptual Model

Charge Question 1(a). EPA has introduced a general conceptual model in Chapter 2, including the selection of assessment endpoint and indicator variables. What is your perspective of the general conceptual model?

The purpose of the conceptual model is to provide relationships between nutrient levels (nitrogen and phosphorus) and biological responses that will allow EPA to develop a set of numeric criteria to interpret the current narrative criterion being used by the state of Florida. The consensus of the panel was that the general model approach provides a strong basis for choosing numeric criteria although there were numerous concerns about the details on how and where the models would be applied, and the adequacy of the data. The EPA conceptual model (Figure 1 below) proposes to relate nutrient levels to the aquatic life use (balanced natural populations of aquatic flora and fauna) using three general approaches:

1. Identify reference conditions for a water body type based on available data or best professional judgment;
2. Use predictive stressor-response relationships and nutrient/algal thresholds; and/or
3. Use numerical water quality models to predict nutrient loadings that would be protective of system biology.

These conceptual approaches would translate Florida's objective of "balanced natural populations of aquatic flora and fauna" into numeric criteria for three biological endpoints: sea grasses, phytoplankton, and faunal communities. While agreeing that these endpoints are appropriate, the Panel strongly felt that these endpoints need to be much better defined and, in some cases, connected to the explanatory variables that would be the basis for setting numeric criteria. The term "balanced" is not defined in the document and is subject to a great range of interpretation. EPA needs to provide a definition of "balanced" early in the document. EPA also should define how it will determine these three endpoints, preferably in quantitative terms. More information on the methodologies that will be used needs to be included in order to determine if the general conceptual approach is workable within the time constraints. The Panel recognizes that details on methods are to some extent specific to type of water body and appropriate for later chapters, but further information on the methods is needed in this chapter as well.

Each of the three approaches has strengths and weaknesses. The use of nutrient reference conditions for a system implies that nutrient concentrations and loadings to a system are known with enough certainty that target values protective of biological endpoints can be determined. In cases where data specific to a system are not sufficient, best professional judgment could be used to determine suitable target values. There has been extensive hydrologic modification of Florida's waters and extreme weather events, a number of which have occurred in the last 15 years, which complicates defining reference conditions in the context of spatial and temporal variability. Current reference conditions may not represent historical conditions. EPA may need to state explicitly the general hydrologic ranges over which these targets will be useful and have clearly stated goals in cases where remediation is suggested. When using the reference condition approach, EPA also needs to pay careful attention to the data sets used in setting values, especially if relatively short-term data sets are heavily influenced by recent hurricanes. Comments by Briceno et al. (2010) may be useful in this regard. The use of predictive stressor-response relationships and thresholds assumes that data on nutrient-organism interactions from Florida waters and other regions, or countries, could be appropriately applied to setting protective target values.

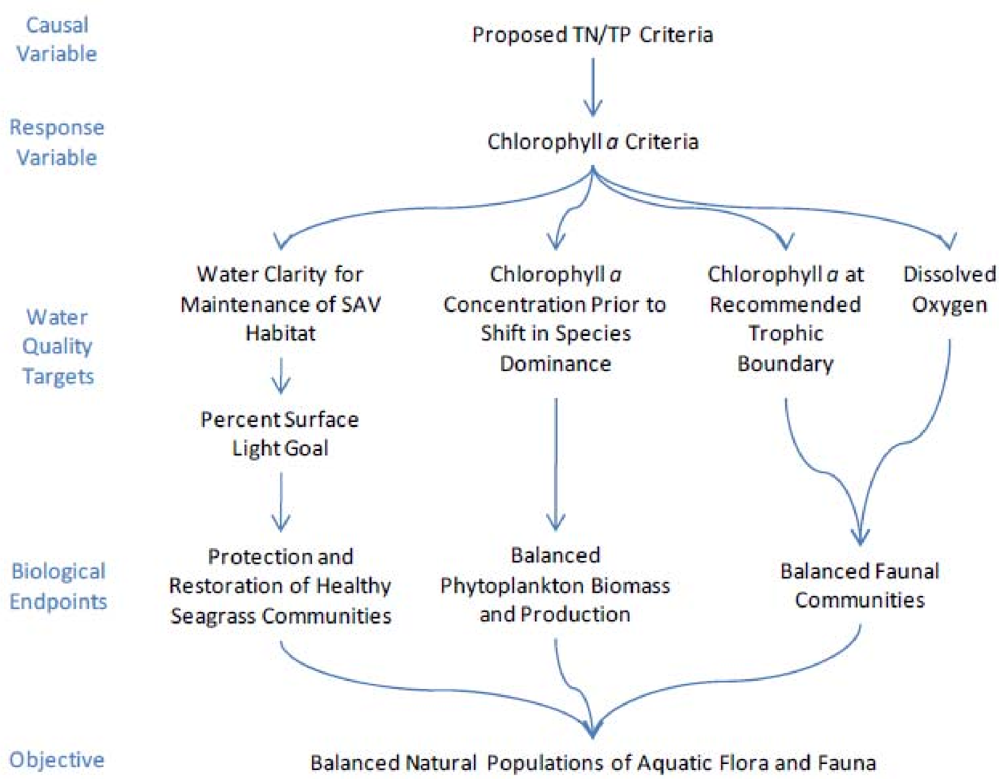

Figure 1. EPA's Proposed Conceptual Diagram Relating TN/TP Criteria to Florida's Narrative Nutrient Criterion (Source: U.S. EPA 2010; Fig. 2-1)

The use of numerical water quality models assumes that models would be a useful and realistic representation of nutrients and other water quality parameters. For practical application of numerical models, there still remain questions as to the appropriateness of selected models, availability of data, and level of detail required to adequately populate each model approach. For example, the EPA document states that a watershed model will be run with all anthropogenic sources removed to determine background TN and TP levels. More information and justification is needed to provide assurances that the models being used can adequately accomplish this with the stated degree of certainty. Most water quality models have been developed to assess and predict fate and transport processes as a result of anthropogenic activities and not for determining pristine conditions. Detailed validation of such "off label" uses is needed, which means calibration with non-impacted watershed loads. However, there are few non-impacted watersheds with conditions that reflect baseline concentrations, in relation to determining water impairment. A key factor involved in using numerical models would be their validation and some analysis of uncertainty for each of the systems where they are applied. It may not be feasible to apply these models to a large number of estuaries in a short period of time.

The three approaches listed above are being applied somewhat differently within the different categories of Florida waters, and each approach has different data requirements and more importantly different assumptions, limitations and uncertainties. The EPA document notes that EPA may use one, two or all three of these approaches for a particular water body. There would be a greater confidence in the criteria if all three approaches were applied, or as many as possible, to each of the systems if data are available. This would provide an ensemble approach and a range of values for setting numerical criteria. However, this could result in three different answers as to what numeric values would be protective. This is understandable given the different conceptual bases for each approach, but the EPA document should discuss how the results from multiple approaches would be integrated to develop the final numeric criteria.

Specific suggestions on conceptual model approaches for different ecosystem types follow. Further discussion of EPA's proposed approaches can be found in the responses to the charge questions for specific categories of waters.

Protection and Restoration of Healthy Sea Grass Populations

Chapter 3 of the EPA document describes in more detail how a healthy sea grass population might be determined using historical data and colonization depth. This is a specific and quantifiable parameter. A brief explanation of this is needed in Chapter 2 to outline the approach. We did not find specific decision criteria for determining when management objectives have been met for impaired water bodies or what sort of magnitude changes would be considered a significant change (i.e., what percent of historical sea grasses coverage would be set as a target for restoration?). This needs to be included if the numeric criterion is to be applied.

The Panel is concerned about relying upon water column Chl-*a* as the sole criterion to protect sea grasses. No numeric criteria directly related to macroalgae or epiphytes are being proposed. In systems where the nutrients are largely taken up by the phytoplankton, Chl-*a* will reflect the major impact of nutrient loading. However, there are systems where even with nutrient increases, water column Chl-*a* remains low due to short water residence times, but macroalgae proliferate. In these systems water column Chl-*a* is a poor measure of nutrient effect. Hauxwell et al. (2001, 2003) found that light levels in benthic macroalgal mats prevented young eelgrass shoots from being established. Epiphytes can also increase in systems where water column Chl-*a* levels remain fairly low.

EPA could consider an approach linking nutrient loading with sea grass areal loss for protecting sea grass communities. This approach has been successful in Tampa Bay (Greening, 2010). It was also applied to a range of systems in New England (Latimer and Rego 2010 Estuarine, Coastal and Shelf Science), with data on eelgrass loss for a number of estuaries being compared to calculated nutrient loadings. The Latimer and Rego study found eelgrass loss began to occur at N loads above 50 Kg ha^{-1} y^{-1} and eelgrass disappeared at 100 Kg ha^{-1} y^{-1}. It may be possible to develop a similar relationship for Florida sea grass systems, and the panel recommends that EPA consider this approach.

Phytoplankton production and biomass

The Panel agreed that Chl-*a* concentrations are both sensitive to nutrient inputs and an important measure of ecosystem health and therefore a reasonable endpoint in itself. However, Chl-*a*, which measures biomass, cannot be used to infer anything about whether or not populations were "balanced" in terms of species composition or relative abundance/dominance. In testimony, EPA provided examples where toxic blooms are known to occur at high Chl-*a* values. There also are data in the literature (e.g., give references…) to suggest that undesirable species are more prevalent in areas with higher nutrient loading (and higher algal biomass), but low biomass does not assure that a toxic species will not occur or that species composition has not changed. Similarly, while Chl-*a* is a measure of biomass (standing stock), it is not a measure of production (a rate) and cannot be used to assess the biological endpoint of production. In sum, while we support using Chl-*a* as an endpoint, its limitations need to be recognized.

Balanced Faunal Communities

The conceptual model in Chapter 2 of the EPA document does not include a direct metric for balanced faunal communities, but proposes that healthy faunal communities rely upon sufficient concentrations of DO. The document cites studies where low DO causes mortality and impairment of marine life. Thus, EPA proposes to use the Florida State DO standard to maintain the biological endpoint of balanced faunal communities. They propose to look for relationships between TN and/or TP and DO, and use those relationships to determine numeric criteria for TN and TP that are protective (i.e., that are associated with attainment of the existing DO standard). How these linkages will be made and which faunal metrics will be assessed needs to be more fully explained and clarified. Chapter 3 (p. 49) of the EPA document implies that the absence of hypoxia will be an indicator for the presence of balanced communities, which would imply that ambient nutrient levels where hypoxia is absent would guide setting the numeric criteria. Chapter 3 also notes that DO can be computed in water quality models from TN and TP loading. The Panel is concerned with the absence of any reference to faunal metrics (see also response to charge question 2).

Conceptual Diagram

Overall, information is given on how these three basic conceptual approaches and three biological endpoints would be applied in each of the categories of Florida waters. The conceptual diagram (Figure 1 above) is a good representation of important linkages. The upper three levels (Causal Variable, Response Variable, and Water Quality Targets) are dealt with at great length, but the bottom two levels (Biological Endpoints, Objective) are not discussed in sufficient detail. Terminology that implies TN and TP are causal variables should be dropped in favor of terms such as driver variables. While there is a cause/effect relationship between nutrients and Chl-*a*, there are many other factors that control Chl-*a*. In Figure 2-1, Chl-*a* also is shown to be a water quality target that relates to balanced faunal communities. While there are mechanisms by which these two are linked in addition to changes in DO (e.g., through increased sediment loading) we did not find any discussion of mechanistic links in the EPA document. Also, while low DO is closely linked with eutrophication, it is not the only mechanism of nutrient impacts, which is what is implied in the diagram. The Panel suggests that EPA alter the diagram or include an explanation on how numeric criteria for Chl-*a* will be linked to balanced faunal communities. EPA also should provide more background and theory on the relationships between biological endpoints and water quality targets. There are many factors that regulate "balanced" ecosystem functions in addition to the few listed in Figure 2-1, including predation, harvest, salinity, substrate, species turnover, and N:P ratios.

Dissolved Oxygen Targets

Additional concerns arose during the discussions and submitted testimony about using a single DO standard. Some sea grass meadows routinely exhibit low oxygen conditions at night even in the absence of any nutrient impairment. This diel cycling of oxygen—from supersaturated during daylight hours to undersaturated, and at times hypoxic, at nighttime—has recently been found to be common in shallow vegetated and unvegetated habitats (Verity et al. DATE ; Moore et al. DATE; Tyler et al.DATE). Similar conditions appear to occur in some Florida waters (see submitted testimony by Boyer and Briceno on Dec. 2). Another issue is that oxygen is less soluble under higher temperatures and higher salinity, conditions seen in many of Florida's warm temperate and subtropical waters. Hence, low DO criteria may be better characterized by percent saturation. Although the Florida DO numeric standard is not a subject of the current review, the Panel raises these issues to point out some of the challenges in relying simply on a DO standard to protect healthy biological communities.

TN and TP Criteria

In the document TN and TP are listed as "causal variables" and defined (p. 39) as concentrations (mg/L) of total (organic and inorganic) N and P. This may lead to confusion. As the table on page 36 of the document points out, TP and TN <u>loading</u> are normally considered to be the ultimate driver of ecosystem changes while TP and TN water column concentrations are "associated with influent loading over the long term". Hence this would make water column concentrations of both TP and TN explanatory or response variables. The narrative (p. 53) also refers to loading as the causal variable and water column concentrations as a response variable. This distinction is important when considering using TP or TN to predict other parameters. Many assessments have been based upon loading. Loading data, when available, would be expected to be a better predictor of Chl-*a*, hypoxia and sea grass loss than concentration. TN and TP may co-vary with Chl-*a*, since both are contained in phytoplankton, so they also are not completely independent from Chl-*a*. This presumes TN and TP are measure on unfiltered samples; yet the document does not clearly state that. Given the availability of data, there may be excellent reasons to use TN and TP concentrations as numeric criteria but they should be considered response variables. It also would be useful to characterize these variables in more detail, including the temporal and spatial scales over which they would be measured (i.e., weekly or monthly averages, surface values, depth-integrated samples, discrete depths).

The issue remains of whether TN and TP or "reactive N and P" (i.e., DIN and DIP) are the most relevant variables to link nutrient enrichment to specific effects on biological endpoints (i.e., primary production, biomass as Chl-*a*, and cascading effects such as food web alterations and hypoxia). This issue has been the subject of considerable research, discussion and controversy for decades. Much of the uncertainty regarding whether to use TN and TP or more "reactive" dissolved forms of these nutrients revolves around the bioreactivity and roles of organic forms of these nutrients. Bioreactivity may be system-specific (or even system-component-specific), adding to the complexity and uncertainty of measuring responses and impacts on water quality and habitat condition. It is important for EPA to discuss this issue in the context of developing numeric nutrient criteria for nutrient-sensitive waters, both in Florida and nationally.

While we know that nutrients are being delivered to coastal systems far in excess of preindustrial loadings and the negative consequences of these excessive loadings, there is little consideration of the linkage between the Causal Variable and Objective. How the three general approaches proposed by EPA will incorporate data on what constitutes balanced populations of flora and fauna needs to be expanded. The numeric criteria are being determined to meet the Objective, but there is inadequate information on the objective. A clear definition of what constitutes a balanced (or imbalanced) natural population is critical, given that Florida's existing narrative nutrient criterion states:

> "In no case shall nutrient concentrations of a body of water be altered so as to cause an imbalance in natural populations of aquatic flora or fauna."

Some consideration for what portion of TN and TP loading in a system is from natural sources versus anthropogenic sources is needed. This is particularly important for open coastal waters where conditions may be influenced by non-anthropogenic nutrient sources from outside the geographic boundaries of the coastal zone. More emphasis needs to be placed on defining what balanced populations are and determining existing conditions of these populations in Florida waters. While we know that reducing nutrients is key to restoring ecosystems in general, the difficulty lies in setting criteria that can be realistically achieved. In setting TN and TP criteria, careful consideration needs to be given to all sources of N and P that in combination affect the biological endpoints for a system.

Uncertainty

Throughout the document uncertainty is briefly mentioned as being introduced because some environmental variables can covary with explanatory variable of interest. However, uncertainty issues related to numeric criteria should be described further and how they might influence the use and appropriateness of specific numeric criteria. It is essential that predictions explicitly state and detail the level of uncertainty inherent in those predictions and those predictions be "ground-truthed" (not "validated") using site-specific data. The uncertainty among the various factors that are involved in the cause-effect relationship for a particular system of interest should be assessed.

The morphology of the aquatic system, habitat, and spatial and temporal relationships within the water body all are important in modifying the relationship between nutrient concentrations (both N and P) and observed endpoints. In fact these factors may dominate the cause-effect pathway so that nutrients are not the primary explanatory variables within the expected limits of the system. These factors need to be better documented, so that the uncertainty of the relationship can be reduced. A statistically significant stressor-response relationship can be derived that may represent only a small portion of the variability in the data. Relying solely on this relationship would result in a tremendous amount of uncertainty in the final criterion.

3.1.2. Categories of Florida Waters

> <u>*Charge Question 1(b).*</u> *EPA has delineated the State of Florida into 4 general categories of waters—Florida estuaries, Florida coastal waters, South Florida inland flowing waters, and South Florida marine waters—for purposes of considering approaches to numeric nutrient criteria development. Are these categories appropriate and scientifically defensible?*

Separation of estuarine and coastal waters is appropriate given the differences in natural populations of aquatic flora and fauna between higher salinity coastal systems and lower salinity estuarine and inland systems. Freshwater management in the region is complex and the separate consideration of South Florida is warranted, although the Panel recomments that the term "marine waters" be replaced with "estuarine and coastal waters" for clarity and consistency. A finer classification based on degree of impact may be useful; for example, to separate the Caloosahatchee and St. Lucie estuaries from the other Florida estuaries, given their unique (i.e., strong human influence) hydrological relationship to Lake Okeechobee. While nutrients clearly influence the biota in these systems, salinity levels play a stronger role than is typically the case in other Florida estuarine systems (cf. Kraemer et al. 1999; Doering et al. 1999; Steinman et al. 2002).

The category of South Florida inland flowing waters seems to be a grab bag for waters that don't fit anywhere else. It would be preferable to have a strong scientific rationale for this classification, as opposed to a default category. It would be helpful if some on the details presented by EPA staff on the delineation of Florida's waters were included in the text.

Comment: I do not know what is being asked for here. Can a clearer reference be made?

3.2. <u>Florida Estuaries</u>

3.2.1. Delineation and Data Sources

<u>*Charge Question 2(a).*</u> *Are the data sources identified appropriate for use in deriving numeric criteria in Florida's estuaries (as discussed in Sections 2.4 and 3.2)? Is the SAB aware of additional available, reliable data that EPA should consider in delineating estuaries or deriving criteria for estuarine waters? Please identify the additional data sources.*

In general the geographic delineations of estuaries seem appropriate. We were unclear why a salinity of 2.7 psu was used to delineate the upper reaches of systems. Traditionally, such salinity would denote an oligohaline region of an estuary. Why not use "head of tide" or a salinity of < 0.5 psu? In any case, sediment nutrient dynamics change in this salinity transition zone (from approximately 0.0 to 5.0 psu). For example, at the toe of salt, P releases from sediments can increase sharply. Wherever the upper boundary is fixed, such issues need to be considered. EPA also should consider adding another unit to the estuary delineation that would focus on tidal creeks. A case was made that these systems are common but have different characteristics than the open estuaries and therefore should have different nutrient criteria.

The Panel has few, if any, issues with the data sets presented. The summary tables in the EPA document indicate a careful review of data sources, including attention to time-series data. We encourage continued searching for appropriate data. In public comments to the Panel, one researcher (Dr. Tom Frazer, University of Florida) indicated that additional data are available for some estuarine areas and have yet to be utilized. It may be that County agencies have data sets not yet considered. This effort could be especially useful in the Big Bend area, where offshore seagrass beds are extensive, satellite data on Chl-*a* are not useful, and existing data sets from prior studies are rare. All data sets would need to meet EPA requirements for QA/QC, but the Panel encourages EPA to continue consultations with state and local agencies and researchers to access additional data and local knowledge where possible.

3.2.2. Assessment Endpoints

<u>*Charge Question 2(b).*</u> *Are the assessment endpoints identified in Sections 2.3 and 3.2 (healthy seagrass communities; balanced phytoplankton biomass and production; and balanced faunal communities) appropriate to translate Florida's narrative nutrient criterion into numeric criteria for Florida's estuaries, given currently available data? Does the SAB suggest modification or addition to these assessment endpoints?*

Healthy Seagrass Communities

Florida seagrass beds are an extremely valuable natural resource, and the two largest contiguous seagrass beds in the continental United States are found in the Florida Keys and Florida's Big Bend region. Approximately 2.2 million acres of seagrass have been mapped in estuarine and near-shore Florida waters by researchers at the Florida Fish and Wildlife Research Institute in St. Petersburg (reference?). However, when seagrass beds growing in water too deep to easily map are included, the total area of seagrasses within Florida waters and adjacent federal waters is likely over 3 million acres. Florida's seagrass beds improve water quality and reduce shoreline erosion, but their most important ecological role is to provide food and shelter for many economically important finfish and shellfish species (reference?). Estimates of the ecological services provided by seagrass beds range from $5,000 to $20,000 per acre per year (reference?), and it is entirely appropriate that EPA use healthy seagrass communities as one of its assessment endpoints.

There are, however, several issues relating to seagrasses that deserve further consideration. First, as acknowledged in the draft EPA document, Chl-*a* usually explains a significant amount of variation in water clarity, but frequently does not explain the majority of this variation, which is often greatly influenced by colored dissolved organic matter (CDOM) and inorganic material in the water column. Of greater importance, seagrasses in the shallow waters of Florida are typically shaded more by epiphytes growing on their leaves and by associated macroalgae (see Dixon 1999, and review by Burkholder et al. 2007) than by Chl-*a* in the water column. Thus, EPA should consider a measure of epiphyte abundance in addition to the proposed determination of Chl-*a* in the water column. Epiphyte abundance is most often controlled by the animals that feed on these epiphytes (Hughes et al. 2004; Burkepile and Hay 2006; Heck and Valentine 2007; Baden et al. 2010). This control by consumers is often referred to as top-down control and while this subject is outside the scope of the draft document, it is notable that both nutrients and food web structure affect the condition of seagrass meadows.

Comment: This statement argues against using epiphyte biomass as a measure of excess nutrients impacting the condition of seagrasses. This is not consistent with what is recommended in the previous sentence.

Balanced Phytoplankton Biomass and Production

As noted earlier (3.1.1) EPA needs to provide a clear definition of "balanced". If EPA is not referring to species composition and relative abundance, but rather the entire phytoplankton or benthic microalgal communities, then Chl-*a* or other indicators of biomass (i.e., dry weight, particulate C, total cell counts) will suffice. If EPA is referring to species diversity or some other index of biological diversity, then more specific techniques will have to be employed, including microscopic species identification, photopigment analyses, molecular analyses, etc. We recommend community-level biomass metrics, using Chl-*a* or other indicators of biomass, as this is best related to nutrient and C-flux, hypoxia, and other drivers/indicators of impacts of and responses to nutrient inputs. Endpoints that require taxonomic-level resolution (e.g., to characterize harmful algal blooms) will need more specific suites of indicators to identify, quantify and characterize factors such as toxicity and food web effects. Such taxonomic analysis may not be possible with current monitoring programs in the systems and regions of interest.

Balanced Faunal Communities

There is little discussion in Chapter 3 of how "balanced faunal communities" are defined, and this is a concern for several reasons. First, given the generally shallow nature of Florida estuaries and our general impression that water clarity is (or was) high, it is likely that these systems were (or are) benthic-dominated. If this is the case, a variety of benthic autotrophs and heterotrophs could provide strong metrics for estuarine health. Other estuary programs have used this approach effectively (e.g., …reference). However, there apparently are not sufficient data to pursue this approach for Florida estuaries. Second, a strong shift from one common benthic species to another (e.g., a pollution tolerant species) can provide a good indicator of benthic habitat condition or deterioration, although the "species pair" might differ among

Comment: What about estuarine IBI approaches?

estuaries. Has this approach, using indicator species, been considered as a measure of the health of faunal communities for Florida estuaries?

Chapter 3 indicates that hypoxia will be used as an indicator of compromised benthic habitat condition. As a first pass, this will certainly tell us something about these habitats but not all that is needed. Unfortunately, when hypoxic conditions are observed, impacts on the biota usually have already occurred. It would be useful to have indicators of stress on the faunal community before such degraded conditions develop. In addition, DO in Florida seagrass meadows during the early morning hours is often below the levels considered to be hypoxic in unimpaired Florida waters, owing to the low amounts of oxygen that can be dissolved in the high temperature and high salinity waters characteristic of Florida and the high rates of night-time respiration in the biomass rich seagrass meadows. Thus, as suggested in comments provided by Boyer and Briceno (2010), a percent saturation criterion may be more useful than an absolute measure of oxygen concentration for assessing whether faunal communities are in balance.

3.2.3. Approaches

> <u>Charge Question 2(c).</u> *EPA describes potential approaches in Section 3.3 (reference conditions, stressor response relationships, and water quality simulation models) for deriving numeric criteria in Florida's estuaries. Compare and contrast the ability of each approach to ensure the attainment and maintenance of natural populations of aquatic flora and fauna for different types of estuaries, given currently available data?*

As noted previously, the Panel recommends that EPA provide a more quantitative description of the concept of balanced phytoplankton and faunal communities, and remove the word "production" in the description of phytoplankton unless measures of production are added. Nutrient criteria development should take into account the natural diversity of Florida estuarine systems. For example, in some systems having low N/P nutrient ratios, blue-green algae may be the normal dominant species. Recognition of special system features will prevent systems from failing to meet criteria on the basis of natural background conditions. In addition, the estuarine continuum, from freshwater to the sea, often involves a transition from P to N limitation and possibly zones where co-limitation occurs. Thus, a duel nutrient strategy is warranted and we agree with EPA's decision to take this approach. Similar strategies have been adopted in the Chesapeake, Neuse and Baltic (references?).

The Panel has a general concern that the timetable for completion of this work may be unrealistic. A substantial effort already has been made to get the work to this stage, much of it solid and thoughtful. However, much work remains to be done and—in the case of Florida estuaries—it is not clear what approach will be selected, if multiple approaches will be used, and which approaches will provide useful information towards the goal of developing nutrient criteria. So, we are concerned about this large effort degrading into a footrace that will sacrifice quality work for the sake of a schedule.

The Panel emphasizes that there is no single approach that is ideal for developing nutrient criteria. This being the case, we support using multiple approaches where possible. If results for two or more approaches converge, then there is increased confidence in the results, and EPA needs to provide guidance on how to use this information to develop a criterion. If different approaches yield conflicting results, then this is strong indication that additional work is in order.

Reference Condition

Philosophically, the reference condition approach is the most satisfying, although making it operational is often difficult because sufficient data are lacking to define "reference conditions", and the problem of "shifting baselines" (Pauly 1995)—in other words, many ecosystems have been impacted by human activities for some time and we run the risk of using degraded coastal environments as reference conditions when the true (unimpacted) reference conditions have long since ceased to exist. We are aware of at least one other State (New Hampshire) using the reference approach for developing nutrient criteria and that effort yielded some useful results. Our experience also suggests that this approach, if it can be implemented, might be the most "time-efficient" pathway to developing nutrient criteria.

Stressor-Response Models

The Panel was disappointed that more attention was not given to the stressor-response approach. Given that one of the Nation's best estuarine restoration examples is Tampa Bay and that they used a stressor/response approach to developing local criteria, suggests that this approach should be more developed in the State-wide effort. Limnologists have had great success with this approach. Recently, EPA staff in New England published results of an analysis relating nutrient loads to seagrass health in a variety of small coastal systems (reference?) These sorts of studies suggest that this approach needs to be more seriously considered. The EPA document simply refers to "regression models," leaving many readers with the impression that EPA is considering only the simplest forms of regression analysis. (In contrast, the discussion of simulation modeling packages is presented in considerable detail.) Although statistical models are correlative, and the amount of variance explained by the correlations can be less than that needed for criteria development, the Panel felt that a more thorough consideration of the stressor-response approach is warranted.

Water Quality Simulation Models

The level of detail on simulation modeling in the EPA document suggests that EPA has decided to use modeling as the primary tool for development of nutrient criteria. This may not be the case but we urge some caution here. The description of the model(s) sounds great, which can be quite seductive, and some issues can only be addressed with simulation models (e.g., forecasting, understanding highly interactive processes). However, using simulation models would be a major undertaking, requiring considerable time and money (note the Chesapeake Bay model has been under development for about 25 years and still does not predict inter-annual hypoxic volumes well), and useful results are not guaranteed. There are also very considerable issues related to data needed for calibration and verification of model results.

If the simulation modeling approach is selected, a reasonable representation of internal nutrient cycling needs to be included. In the generally shallow Florida systems, benthic processes will be especially important. In addition, these processes will interact with temperature and flow changes. Ultimately, nutrient concentrations reflect the net effect of these biogeochemical processes, as well as loadings.

Hydrologic Forcing

From a spatial perspective, the location of phytoplankton production and biomass responses to nutrient inputs is strongly influenced by freshwater inflow and its impacts on estuarine residence time. Under drought conditions, the biomass peak, or Chl-*a* maximum (C_{max}) will tend to be in the most upstream portion of estuaries (Valdes-Weaver et al., 2006). Under moderate freshwater discharge and flow conditions, C_{max} will form in mid-estuarine locations, while under high flow conditions, C_{max} will tend to predominate downstream (Valdes-Weaver et al. 2006; Paerl et al. 2007). Under extreme hydrologic conditions resulting from tropical cyclones, C_{max} may not form at all, but rather the maximum phytoplankton biomass response will be in the sounds and coastal waters (Paerl et al. 2006a, b). These conditions represent a special challenge, because it may be difficult to evaluate and assign numeric criteria for nutrient loads to estuaries, as the response will not occur in estuarine waters.

Comment: This is part of that overlap between stream DPV and estuarine IPV…

Comment: may not?

We are experiencing a period of increased tropical cyclone activity and intensity (Webster et al. 2005). Florida is particularly impacted, because it experiences more tropical cyclone strikes than any other state in the U.S. Therefore this aspect of climate change needs to be factored into the anticipated/predicted responses to nutrient inputs and the development of nutrient criteria. Conversely, periods of extreme (and record) droughts require additional attention and consideration in the context of the development of nutrient criteria, as the location and amounts of phytoplankton biomass responses to nutrient inputs will be dramatically affected.

Lastly, Florida is undergoing significant increases in freshwater withdrawal (for drinking water and agricultural irrigation purposes) from its lakes and rivers. This will impact freshwater discharge to estuarine and coastal waters, which in turn will impact the location and magnitude of phytoplankton (including HABs such as cyanobacteria and dinoflagellates) as well as benthic microalgal and macrophyte responses to nutrient inputs. This growing demand will need to be factored into the formulation of nutrient criteria at it will influence freshwater discharge, nutrient loads, nutrient concentrations and microalgal responses in impacted estuaries.

Climate (and temperature) Needs Consideration

In addition to climate-related hydrologic effects, changes in temperature need to be considered. Changes in the range of 1.5 °C have been noted in some systems during the past 60-70 years. Temperature change will have an influence on phytoplankton community composition (i.e., “cyanobacteria like it hot”; Paerl and Huisman 2008), as well as key biogeochemical nutrient and organic matter transformation processes (e.g., nitrification, denitrification, and sediment oxygen demand).

Groundwater and Surface Water Withdrawals

We note the interactive effects of watershed groundwater and surface water withdrawals for agricultural, industrial and municipal uses. Specifically, water withdrawals will alter the nutrient concentrations (by altering the dilution characteristics), loads and freshwater discharge, which in turn will impact nutrient-phytoplankton growth and bloom thresholds, estuarine flushing rates and residence time. These physical-chemical alterations will impact the timing, seasonality and locations of phytoplankton and benthic microalgal growth responses and blooms.

The development of nutrient criteria for specific watersheds and estuarine receiving waters should take into account these interacting effects.

Comment: I think this point is made adequately under hydrologic forcing?

Threshold Changes

Comment: In addition to the caution to beware of nonlinear, threshold effects, when setting the initial criteria, this section also is noting that climate change will made it particularly important to revisit the criteria values as conditions (of hydrology and/or loads) change? If so, can this be stated more clearly?

When setting nutrient criteria, the Panel recommends that EPA consider the possibility of threshold changes that could occur in these systems. We include here non-linear responses, lags relative to input changes and general “state changes”. These changes will in part be a result of changing nutrient loading and freshwater discharge dynamics due to changing anthropogenic activities in watersheds and airsheds. They will also be modulated by climatic changes, including changes in rainfall (and conversely drought) intensities, frequencies and geographic patterns, as well as temperature changes (i.e. warming, which will favor the growth and proliferation of nuisance taxa such as cyanobacteria). These changes need to be considered during future triennial water quality standards reviews.

3.3. Florida Coastal Waters

3.3.1. Delineation and Data Sources

Charge Question 3(a). Are the data sources identified in Sections 2.4, 4.1.1 and 4.2 appropriate for use in deriving numeric criteria in Florida’s coastal waters? Is the SAB aware of additional available, reliable data that EPA should consider in delineating coastal waters or deriving criteria for coastal waters? Please identify the additional data sources.

The EPA document defines the outer boundary of the coastal zone based on the jurisdictional definition of 3 nautical miles. Although the 3-mile limit is legally mandated for regulation, the Panel recommends that EPA also consider monitoring the sensed chlorophyll in waters further from shore. Given the dynamic nature of algal blooms in the Gulf of Mexico in particular, it is possible, and perhaps even likely, that blooms that form further than 3 miles offshore will migrate toward the coastline, thus eventually "appearing" in the 3-mile segment. It will be important to understand the source of such patches of elevated chlorophyll, and to determine whether they are found in close proximity to the shoreline because of land and estuary-derived nutrients or formed offshore.

Restricting the offshore boundary to 3 nautical miles greatly reduced the number of calibration samples compared to the available data. As there is no clear boundary in water types at three nautical miles, it is appropriate to use data from the entire shelf. Extending the outer boundary to the shelf break in this way will improve the quality of the dataset. According to EPA personnel, adding these additional data increased both the correlation and the slope of the calibration graph (Fig. 4.6 in the EPA document) considerably (e.g., r^2 increased from 0.52 to ~ 0.8). EPA might consider using anomalies relative to either seasonal or annual means—rather than absolute Chl-*a* concentrations—in their estimates (see Stumpf et al, 2003, 2009; Tomlinson et al., 2004). This will mitigate problems inherent in working close to the coast, as bottom backscatter reflectance, for example, will be constant and therefore disappear from the equation.

The coastal segmentation scheme suggested in the EPA document apparently is a result of historical precedence, rather than any underlying scientific rationale. Given the general alongshore flow that creates anisotropy with strong gradients perpendicular to the coast and weak gradients parallel to it, EPA may wish to consider segments defined in terms of bathymetry.

Another recurrent topic in panel discussions was the "missing kilometer" at the coast where *in situ* data are not being used because the satellite chlorophyll estimate is corrupted by the presence of land within the pixels and because of backscatter from shallow water. A potential solution may be to use turbidity data to connect conditions in the estuary proper with the coastal system just offshore, thus bridging the km gap. Another potential solution would be to collect airborne spectrographic imagery (CASI), but this would require a new data collection scheme.

The Panel recommends that a boundary calculation be undertaken to better understand chlorophyll levels in the coastal zone (i.e., to relate observed chlorophyll levels to TN/TP concentrations or loadings to the coastal zone). A boundary calculation might consist of a first-pass estimate of the total nitrogen and total phosphorus released into the coastal zones from all sources.

Comment: we need to clarify what is meant by "a boundary calculation." What would one do with an estimate of total N and P released into the coastal zone?

The Panel agrees that the use of remote-sensed data to develop a reference criterion for Chl-*a* is appropriate and sensible for this large, poorly sampled region. The use of these data, however, requires calibration with *in situ* chlorophyll samples, of which there are few. The panel accepted that these sources are limited, but felt that additional sampling, including opportunistic sampling (using ferries, fishing and charter boats, etc.), where feasible, would improve the dataset. While the use of a reference criterion (Chl-*a*) is reasonable, the Panel is concerned with the sole reliance on a surrogate (see below) with no direct measurements of nutrients being made.

The question also arises as to what reference level is applicable in this region. Historic nutrient concentrations were likely very different from today (although little data are available to provide quantitative information), yet the document assumes that these areas are currently supporting a balanced phytoplankton community. Although the Panel recognizes that a longer data record is not available, it is not clear whether the ten-year dataset available from satellite observations constitutes an adequate baseline, given decadal-scale variability.

The Panel notes that reliance on satellite observations may not be as feasible in the future. The life of the SeaWiFS and MODIS sensors are near their end, and while VIIRS may be launched in time, there is also question about that sensor's capability to produce high quality data for chlorophyll. Therefore, the Panel recommends that the EPA ensure that data from the existing U.S. and European satellites, as well as future sensors, be cross-calibrated to ensure as complete a data record as possible.

3.3.2. Assessment Endpoints

<u>*Charge Question 3(b).*</u> *Is the assessment endpoint identified in Section 4.2 (chlorophyll-a to measure balanced phytoplankton biomass and production) appropriate to translate Florida's narrative nutrient criteria (described above) into numeric criteria for Florida's coastal waters, given currently available data? Does the SAB suggest modification or addition to this assessment endpoint?*

EPA is considering a "reference-based approach with satellite remote sensing Chl-*a* observations (Chl_{RS}-*a*) to derive numeric values that translate Florida's narrative criteria and ensure support of a natural balanced population of aquatic flora and fauna. This approach is likely to be effective in Florida coastal waters, because they are optically amenable to remote sensing of chlorophyll, color (CDOM) and turbidity" (Hu et al., 2005; Muller-Karger et al., 2005; Palandro et al., 2004). Remote sensing technology has evolved sufficiently to begin using calibrated imagery for estimating chlorophyll.

Comment: Where is this quote from?

The Panel acknowledges that Chl_{RS}-*a* is the most feasible indicator of nutrient status for coastal waters, given available data. However, we caution that Chl-*a* levels in these waters also are influenced by seasonal water temperatures, circulation and mixing, and influx of nutrient-rich waters from advection or upwelling. Walker and Rabalais (2006), cited in the EPA document, found only about 40% of the variance in phytoplankton production could be ascribed to nutrient concentration, and this was in an area of the northern Gulf of Mexico known to be affected strongly by nutrient inputs from the Mississippi River.

The Panel agreed that Chl-*a* will not be useful as an indicator of species composition, as has been discussed earlier. Given the weak relationship between nutrient concentrations and chlorophyll concentration, Chl-*a* may be more appropriate as a monitoring tool for Class II and Class III waters (i.e., to show whether phytoplankton blooms are increasing or decreasing) than as a regulatory endpoint. There is certainly a potential relationship between nutrients and organic carbon production, but this can vary depending on parameters such as season or relative availability of N and P, as shown clearly in Fig. 4.4 of the EPA document. Also, the carbon: chlorophyll ratio within phytoplankton can vary by an order of magnitude (Banse, 1977), while *Trichodesmium* blooms can arise in low N regimes because these organisms are nitrogen fixers.

Comment: This statement conflicts with statements elsewhere in the report. Does my inserted paragraph just above help explain the concerns?

As stated above, the Panel suggests moving away from using direct measurements of Chl-*a* and instead to consider using anomalies as a means of removing known interferences.

Comment: This recommendation is unclear. How would anomalies be used to set nutrient criteria? If we are going to recommend this, we have to be much clearer about what we mean by this.

3.3.3. Approaches

<u>*Charge Question 3(c).*</u> *Does the approach EPA describes in Section 4.2 appropriately apply remote sensing data to ensure attainment and maintenance of balanced natural populations of aquatic flora and fauna in Florida's coastal waters? If not, please provide an alternate methodology utilizing available reliable data and tools, and describe the corresponding advantages and disadvantages.*

The Panel notes the thorough approach to calibration, but has several recommendations for consideration:

- According to the document, EPA used a 3 x 3 km pixel matrix but only used *in situ* calibration data taken within 3 hours of the satellite overpass. This should be sufficient, as tidal current amplitudes, particularly off the Florida panhandle and over the wide West Florida shelf, are generally small (O<10 cm/s; He and Weisberg, 2002). Largest values are about 20 cm/s in the vicinity of the Big Bend and Florida Bay. Tidal ellipses here tend to be perpendicular to the bathymetry except very close to the coast, where they tend to parallel it (He and Weisberg, 2002; Koblinsky, 1981).
- The Panel recommends that satellite data within a larger, "coastal" context be used for the calibration, i.e., including data from outside the 3-mile zone. Because the calibration presented was not strong ($r^2 = 0.52$), this inclusion of additional data should improve the skill of the model, The Panel also recommends that EPA adopt ongoing calibration with the SeaWiFS satellite and other existing sensors (see above).
- Another issue on calibration concerns the relation between remotely-sensed chlorophyll and water column measurements. EPA calibrated the satellite data to chlorophyll measured in the uppermost two meters of the water column. The ratio between the chlorophyll concentrations in the upper two meters and the full euphotic zone needs to be established.
- The Panel recommends that obvious antecedent bloom data points be removed from analyses as these are likely not representative of desired "reference conditions" (p. 83, paragraph 1, regarding *Karenia* blooms).
- Care should be taken with Type I/Type II waters where calibrations may change. Some, but perhaps not all, of the problems inherent in the change from one water type to another may be covered by using Chl-*a* anomalies rather than absolute measurements.

Comment: I don't understand what is intended here. Above we say that SeaWIFS isn't going to be around for very long. Why are we asking for ongoing calibration.

3.4. <u>South Florida Inland Flowing Waters</u>

3.4.1. Rationale for Criteria

The Panel recognizes the considerable time and effort that has been put into identifying current data sources, assessing endpoints, and developing two approaches to deriving nutrient criteria for inland flowing waters of South Florida. However, the Panel is not convinced from the material provided that nutrient criteria are appropriate for these uniquely artificial and highly managed ecosystems. We identify a number of specific concerns in this introductory section before addressing the specific charge questions. We acknowledge that these comments and questions may not have explicit answers; however, they deserve some thought and consideration.

South Florida's inland flowing waters have a long history of being highly manipulated and managed, and in this regard they represent a special challenge to developing numeric nutrient criteria. The underlying problem is that the canals are classified as Class III waters, although their primary purpose is management of water quantity. Specific concerns include the difficulty in determining what constitutes an appropriate reference condition for these systems, and the related issue of whether or not appropriate data are available to help define reference conditions.

A second concern involves the potential confounding problem of internal nutrient loading from sediment accumulation in these canals. If sediments are a major source of nutrients (and based on SFWMD (2010), sediment accumulation and P mass are quite variable), this internal source could confound relations between water column nutrient concentration and ecological response.

A third concern is that of legacy nutrients. How will hereditary or legacy losses or inputs of N and P to water bodies be considered and accounted for in the proposed approach? This begs the next question facing water resource managers who set targets for nutrient load reduction, that if no water quality improvement or indicator biological response is seen, is this because the targets / criteria are too low, legacy nutrient inputs are an increasingly significant contributor, or because the monitoring interval is not long enough to capture the response of dynamic ecosystems and watersheds? How will continued legacy or hereditary inputs of stressor inputs (N and P) be distinguished from management change-related decreases? Internal recycling of nutrients can mask water quality improvements brought about by nutrient loss reductions affected by land management changes. Given the role of legacy nutrients in influencing water quality in these systems, an adaptive management approach (e.g., as part of the triennial review of water quality standards) is needed to incorporate new monitoring data and revise criteria or loading targets as appropriate.

Finally, South Florida inland flowing waters involve a spatially and temporally dynamic interaction between surface and groundwater flows and as such, biological condition of these waters may be more responsive to hydrology than to nutrients. For instance, N and P loadings can occur at different times of the year and can influence biotic responses depending on timing of inputs. In other words, it is not just how much or in what concentration, but at what time. In dry years, ground water will greatly influence surface water chemistry/quality compared with wet years. There is also concern that cross watershed / ecoregion / system transfers of water and nutrients in ground waters could confound the ability to relate ecological response to water column nutrient concentrations or loadings.

An alternative approach to assessing these South Florida inland flowing waters is to view them as a source of nutrients to adjacent, more oligotrophic systems, rather than for any valued ecological attributes that may be unique to them. This would be consistent with the canal science summary document (SFWMD, 2010), which describes the aquatic life in the canals (macroinvertebrates, fish, alligators), but acknowledges that the ecological value of the canals is secondary to their use for water conveyance. These canals, especially those that drain the agricultural areas, serve as a nutrient conduit. Hence, the nutrient content in the canals can serve as a proxy for "potential impact" to the more natural wetlands and water bodies adjacent to and downstream from canals. It is suggested that EPA consider developing a "Canal Stressor Index" that would serve to assess these impacts to receiving waters.

Comment: this sounds like canal DPVs to protect downstream waters, rather than IPVs?

3.4.2. Delineation and Data Sources

<u>Charge Question 4(a).</u> Are the data sources identified in Section 2.4 and 5.4 appropriate for use in deriving numeric criteria in South Florida's inland flowing waters (as discussed in Chapters 2 and 5)? Is the SAB aware of additional available, reliable data that EPA should consider in delineating or deriving criteria for South Florida's inland flowing waters? Please identify the additional data sources.

There is considerable debate as to whether or not the data in Sections 2.4 and 5.4 of the EPA document are sufficient to derive numeric criteria for South Florida's inland flowing waters. These data sources are certainly the most logical beginning point. However, EPA should look into datasets potentially available from the local water/drainage districts (not water management districts), such as Lake Worth and Loxahatchee, as well as from agricultural interests that border the canals (e.g., U.S. Sugar), although the latter data may be proprietary.

Comment: Not sure I know what these are

The proposed inventory of inland flowing waters that catalogues and distinguishes natural streams and canals should provide very useful information. The EPA document explores the use of the Landscape Development Index (LDI) as a potential approach (and data source) for determining reference conditions in inland flowing waters (reference conditions where LDI < 2, p. 105). It is well established that surrounding land use can have substantial impacts on receiving water bodies, so this approach has conceptual and intuitive appeal. However, insufficient information is available in the EPA document to determine the appropriateness of the LDI approach for South Florida's inland flowing waters. Further concerns include why only a 100-m buffer along a canal is considered; would not the canal's water quality to be determined by the entire area that drains into it? The document cites a study by Fore (2004) to justify this approach. However, Fore (2004) was based on streams throughout the state and not just canals; there are considerable differences in hydrology and land-water interactions between canals and natural stream channels. The 100-m buffers proposed for use with the LDI (p. 105-106) may be too limited, particularly where stormwater pipes convey runoff from distances much further than 100 m.

Comment: The EPA document cites a study by Fore 2004 as the basis for calculating the LDI with 100-m corridor rather than the upstream watershed. In Fore 2004, the correlation for NH3 and LDI is only slightly better using the buffer than the full watershed (r=0.39 versus 0.34). And the analysis apparently used a set of streams state-wide, not just canals.

The condition of these waters is highly influenced by geology and anthropogenic activity. In this regard, there is logic to subdividing these waters according to basin and sub-basin soil types and land uses. An additional challenge is incorporating groundwater hydrologic/nutrient dynamics, which have also been altered, but are likely to be very important in determining nutrient sources and impacts. The proposed classifications in this chapter appear reasonable as it incorporates surface and subsurface flow regimes and flow lines, as well as soil types and human agricultural and urban impacts (i.e., land use). Classification of inland water regions according to soil order, land management systems or color of water; preferably a combination of several should be considered.

Comment: Not sure you would want to use land-use to classify if that is also the stressor variable? p. 104 of the document says subregions will be based on climate, soil, hydrology, geology.

As mentioned above, legacy N and P effects from past management and from natural sources also must be considered. The EPA document appears to minimize the important of legacy effects of past management (e.g., the statement on the top of page 40 in reference to Huang and Hong, 1999). There is a wealth of data on soil nutrient levels (particularly P) available from NRCS and land-grant university extension offices. These soil levels vary greatly as a function of past management, within and among fields and especially within the supposedly uniform LDIs, for which specific criteria concentrations will be set. Additionally, geologic rock deposits vary within areas assumed to have similar nutrient concentrations. Thus, more data are needed to better support the regional classification of South Florida inland flowing waters.

3.4.3. Assessment Endpoints

<u>Charge Question 4(b).</u> *Are the assessment endpoints identified in Section 5.4 (balanced faunal communities, i.e., aquatic macroinvertebrates, and balanced phytoplankton biomass and production) appropriate to translate Florida's narrative nutrient criteria (described above) into numeric criteria for South Florida's inland flowing waters, given currently available data? Does the SAB suggest modification or addition to these assessment endpoints?*

Philosophically (but with practical implications), one can question whether any assessment endpoint is appropriate for systems that have been artificially created. How does one establish an appropriate reference condition for such systems, especially when they are heavily managed? There are no easy answers for these questions, although this has certainly been done for reservoirs. However, given the limited options available to EPA, and the reality that nutrient criteria are required for these inland flowing waters, the panel believes that EPA has taken a reasonable approach.

Comment: I think this is the point. The current classification REQUIRES that criteria be set to protect the designated use (healthy, well-balanced population of fish and wildlife). Panel needs to decide whether to say, class III notwithstanding, it does/doesn't make sense to have IPV criteria for these waters—maybe omit the last sentence?

Comment: This comment conflicts with what is said in the introductory material -- namely that canals are nutrient conduits and the approach taken doesn't make sense in that context. We need some clarity from the writers on this one.

South Florida canals have been constructed continuously over the last century, so it is not clear how reference conditions can be assessed for these very dynamic and flashy systems designed to get water off the landscape quickly. Least disturbed sites tend to be in one region only and may not be transferable to other identified regions. Because canals are unique aquatic ecosystems, more information needs to be presented on how balanced natural populations are to be assessed. An initial inventory of science for South Florida canals, provided by SFWMD (2010), summarizes data on water quality and biological conditions in the canals. The closest analog to South Florida canals would be in The Netherlands where much of the inland waters flow through canals (locally called ditches). There is some literature for some of the assessment endpoints from Netherland ditches (e.g., see Verdonschot, 1987) that may be of some use in developing methods for assessing the status of flora and fauna in Florida canals.

The Panel recommends further consideration and assessment of the response variables (e.g., invertebrates and Chl-*a*) to be used. The form of the nutrients also will be important. For example, distinction is needed among nutrient forms that are of immediate availability to biological uptake —i.e., short-term bioavailability and growth response, such as inorganic nitrogen (NO_3 and NH_4) and phosphorus (PO_4)—compared with losses as particulate and organic forms of N and P (i.e., long-term availability). Some freshwater ecosystem studies have shown that Chl-*a* can be a function of grazing pressures rather than nutrient concentrations. For example, increasing nutrient concentrations in inland flowing waters can increase the number of grazers, which can lead to a lower Chl-*a* concentration; i.e., a top-down regulation of primary production (references?).

Both assessment endpoints have conceptual appeal, but their utility is not straightforward. Aquatic macroinvertebrate community structure and/or traits have been shown to be reliable bioindicators in other aquatic ecosystems. Hence, they are a reasonable starting point for South Florida's inland flowing waters. However, these systems are poorly understood, highly managed, and heavily modified. As a consequence, it is unclear at present if these proposed assessment endpoints can be applied effectively.

The Panel identified areas of uncertainty that need further attention before a reasonable level of scientific confidence can be applied to the use of balanced faunal communities and/or balanced phytoplankton biomass and production. We elaborate on these below:

Faunal Communities

The macroinvertebrate index used by Snyder et al. (1998), provided as Figure 5-8 in the EPA document, shows a good relationship between landuse and macroinvertebrate community structure. However, the macroinvertebrate data provided in a presentation to the Panel reveal a much more tenuous stressor-response relationship between total P concentrations and macroinvertebrate indices (DeBusk, 2010). It is important that EPA examine possible reasons for the lack of correspondence in these two data sets. Possible explanations include the use of different measures of stressor (land use vs. total P), different types of indices (e.g., emphasizing different taxa) or inclusion of inland flowing waters from different parts of south Florida (e.g., that experience different pressures). For example, the relatively high SCI score for the wetland sites shown in the Snyder et al. data may have more to do with habitat quality than nutrients, *per se*. The summary of canal science prepared by the SFWMD (2010) notes that "additional research is needed to select sensitive (macroinvertebrate) metrics and a quality threshold applicable to low gradient streams and canals within the peninsula and Everglades bioregions". The Panel agrees with this statement; if the different macroinvertebrate patterns in these data sets can be explained, aquatic macroinvertebrates may be a very useful assessment endpoint and one that the Panel recommends be given more attention.

Phytoplankton Biomass

There is a relative paucity of phytoplankton data (either as Chl-*a*, species composition, or productivity) in these inland flowing waters. SFWMD (2010) shows geometric mean Chl-*a* concentrations ranging from 2 (Lower East Coast) to 8.0 µg/L (Everglades Agriculture Area) in canals within the South Florida region considered here. However, these concentrations are not related to hydrologic conditions, and it is impossible to assess if they represent actively growing algae populations (as might be expected in a non-flowing canal) or algae being transported downstream (i.e., in a flowing canal) and therefore not representative of local conditions. The hydrologic status of the canal (non-flowing, slow-flowing, fast-flowing) has enormous implications for the plankton community, and this needs to be accounted for in EPA's assessment. At this point, it is unclear if there are sufficient data to know what a "protective" level of Chl-*a* should be for these systems; as a consequence, it is currently not possible to assess whether or not phytoplankton can be used as an effective assessment endpoint.

The inventory of the inland flowing waters, and subsequent screening of water bodies, is an important step and may help in the selection of appropriate endpoints. The approach provided in the technical document is a good starting point, but the Panel has identified some issues and suggestions with respect to the classification procedure. The panel identified several factors that EPA may want to consider with respect to this endpoint:

- EPA proposes a classification of inland water regions according to soil order, land management systems or color of water (see below). They should consider some combination of these, taking into account covariates.
- Currently, EPA does not appear (at least explicitly) to consider the potential influence of humic soils in their classification of inland flowing water types, with respect to their role in discoloration of waters; phytoplankton response will be very different in waters that are naturally colored (i.e., influenced by humics) vs. those that are not.

Comment: This discussion fits better under Approach?

Additional Endpoints

The Panel identified the following four additional endpoints for EPA's consideration:

- **Dissolved oxygen (DO)**: Dissolved oxygen concentration reflects the relative amount of photosynthesis (DO production) and respiration (DO consumption) in aquatic ecosystems. While there is no biotic component to this endpoint, DO might be an alternative endpoint; however, new studies would be needed to determine if DO levels are linked to nutrient loads or concentrations, and not to other factors (such as light), and if groundwater influx (low DO) confounds the use of this assessment endpoint.
- **Algal community structure**: The Panel recognizes that taxonomic analysis is more labor-intensive and requires more technical expertise than measuring chlorophyll, and therefore may not be practical. However, there is far more information in taxonomic structure than in Chl-*a*. Given the potential problems with taxonomic structure (labor-intensive, specific expertise, lack of consistent and available data), a possible alternative would be to focus on the percentage of a particular problematic species (e.g., a certain HAB, such as *Microcystis*). In this case, the analyst would need to be able to identify only a specific taxon, with the assessment endpoint being: not to exceed some pre-determined level of a particular cyanobacteria or dinoflagellate species.
- **Primary productivity** (either in terms of carbon fixed or DO evolved): as with taxonomic analysis, this assessment endpoint may not be practical because of limited data availability and difficulty of data collection relative to Chl-*a.*
- **Benthic algal community structure**: given the geomorphology of these canal systems—their depth and steep banks—there may be insufficient light penetration to allow the growth of benthic algae. However, this endpoint is worth exploration because prior studies have clearly shown the sensitivity of periphyton community structure to P impairment in other parts of South Florida (cf. McCormick et al., 1996; McCormick and O'Dell, 1996; Carrick and Steinman, 2001; McCormick et al., 2002).

3.4.4. Approaches

<u>Charge Question 4(c).</u> EPA describes two approaches in Section 5.4 (reference conditions and stressor-response relationships) for deriving numeric criteria in South Florida inland flowing waters. Compare and contrast the ability of each approach to ensure attainment and maintenance of balanced natural populations of aquatic flora and fauna in different types of flowing water or geographical areas, given currently available data?

The two approaches that EPA is considering for determining numeric criteria for South Florida inland flowing waters are discussed in Section 5.4 of the EPA document. The first is based on reference conditions and the second is based on stressor-response relationships. It seems possible that either the reference condition or stressor-response approach could work for these waters, provided the necessary data can be collected and that they show interpretable patterns. With the reference condition approach, repeated surveys of invertebrates will show changes in community structure and diversity that could be related to changing nutrient conditions, but this is a time-consuming and expensive methodology, particularly if many sites need to be sampled regularly. The stressor-response approach should also work if a suitable relationship between Chl-*a* and nutrient load can be demonstrated in the canals (p. 103), but several of the same caveats apply here as for setting limits in coastal waters. Selecting "least disturbed sites" using an LDI < 2 also may not be feasible in this region that has been subject to active management for many years. Additional comments on each approach are provided below.

Comment: The statement about the feasibility of the stressor response and reference condition approaches conflict with what we said earlier about the fact that the canals are primarily nutrient conduits.

Reference conditions

Briefly, under the approach based on reference conditions, a set of least-disturbed sites would be identified using the Land Developmen Intensity (LDI) index. The total LDI for each site would be calculated as an area-weighted sum of the LDI coefficients for all land uses within an area of influence. Sites with total LDI below 2.0 or another specified threshold would be classified as least-disturbed and would form the reference set. The historical annual values of total nitrogen (TN) and total phosphorus (TP) for these sites would be used to fit lognormal distributions of TN and TP under least disturbance and specified quantiles of these fitted distributions – the EPA document mentions the 0.75 and 0.90 quantiles – would be used as the numeric criteria.

On the bottom of page 107, the EPA document discusses the question of the frequency with which these numeric criteria could be exceeded. This discussion is difficult to follow, but the general point appears to be this: Consider that the estimated 0.75-quantile for one nutrient is exceeded k or more times in n years. Commonly used values are 1 in 3 and 2 in 5. Under the assumption that values in different years are independent and have the same distribution as the reference set (and ignoring any error in the estimation of the 0.75 quantile), the probability of this event is given by:

$$p(k,n) = \sum_{j=k}^{n} \binom{n}{j} 0.25^{j}\, 0.75^{n-j}$$

For fixed k and n, this formula essentially provides a one-sided significance level for testing the null hypothesis that the nutrient distribution is the same as that of the least-disturbed sites. So, for example, under this null hypothesis, the probability $p(1, 3)$ of at least 1 exceedance of the 0.75-quantile in 3 years is 0.58, while the probability $p(2, 3)$ of at least 2 exceedances in 3 years is 0.16.

The Panel notes the following:

- The choice of quantile, k, and n can have a profound effect on the performance of criteria derived in this way and some discussion is needed about how this choice will be made.
- The probability calculations sketched here pertain to exceedances of a single nutrient criterion. If the same rule is applied to both nutrients (and assuming that nutrient levels are independent) then, under the null hypothesis that both nutrient distributions are the same as those for least-disturbed sites, the probability that either or both nutrients exceed their respective 0.75-quantiles in at least k of n years is $1-(1-p(k,n))^2$. Thus for both nutrients, the probability of at least 1 exceedance of the 0.75-quantile of either or both nutrients in 3 years is (0.82?), while the probability of at least 2 exceedances in 3 years is (0.29?).

Comment: I think our argument is clearer if we provide an example. I'm just not sure that my math is correct.

As noted, these calculations assume that the relevant quantile of the annual nutrient levels in least-disturbed sites are estimated without error. An assessment of the impact of estimation error – including non-normality of the log of annual nutrient levels – on the accuracy of these calculations is needed.

Although these calculations provide information about the rate of Type I error (i.e., the exceedance of the criterion when the underlying distribution is the same as that for least-disturbed sites), they provide no information about the rate of Type II error (i.e., the non-exceedance of the criterion when the underlying distribution is different from that for least-disturbed sites). In the jargon of hypothesis-testing, this analysis provides no information about power. To gain such information, it is necessary to consider also the distribution of nutrients in disturbed sites.

To some extent, variability of nutrient levels in the least-disturbed sites will reflect heterogeneity in hydrology, geology, etc. Failure to account for such heterogeneity, which is also present in disturbed sites, may result in numeric criteria that are under- or over-protective for some sites. It would, therefore, seem preferable to develop criteria that account for such factors. The EPA document briefly notes (on p. 108) that EPA also is considering following the reference conditions approach using all sites as a reference set (and not only least-disturbed sites as discussed above). With the exception of the identification of least-disturbed sites, the mechanics of these approaches are the same. However, the underlying logic seems rather different – loosely speaking, one approach aims to reproduce conditions in least-disturbed sites and the other aims to maintain conditions within a specified quantile of the distribution of all sites, whatever their level of disturbance – and this needs to be discussed.

Comment: this is what I think this means. Is that more clear?

Comment: I'm not sure this last sentence is correct. If they use all sites, will they not just say that the lowest quartile represents least disturbed?

Stressor-response relationships

The second approach that EPA is considering for developing numeric criteria for South Florida inland flowing waters is based on stressor-response relationships. This involves developing a statistical model relating the level of Chl-*a* to TN or TP. The EPA document presents examples involving linear, nonparametric, and quantile regressions of log Chl-*a* as response and log TN or log TP as stressor.

The Panel notes the following:

- A fundamental question that the EPA document leaves unanswered is how such fitted regression models will be used to determine numeric criteria, i.e. how they will determine the level of Chl-*a* that will be considered protective of balanced phytoplankton and faunal communities. This is a serious shortcoming that needs to be addressed.
- As the EPA document notes, it has not been possible to develop stressor-response relationships in which a biological endpoint serves as the response. It is for this reason that EPA is considering the use of Chl-*a* as the response. However, if there is a clear relationship between Chl-*a* and TN, say, and a clear relationship between a biological endpoint and Chl-*a*, then there would ordinarily be a clear relationship between the biological endpoint and TN. The fact that it is difficult to identify this latter relationship may reflect limitations of the statistical models considered so far. For example, the effect of TN or TP on a biological endpoint may be modulated by other factors. This effect could be obscured by omitting these factors from the regression model.
- As with the approach based on reference conditions, the relationship between Chl-*a* and TN or TP is likely to be modulated by the effects of hydrological, geological, and other covariates. Failure to account for such factors may lead to criteria that are over- or under-protective at some sites and it would again seem preferable to include such covariates in developing numeric criteria. Furthermore, should TN and TP be considered simultaneously (i.e., is a multiple or simple regression most appropriate)?
- Some of the variability in the stressor-response relationship could be a result of season. This should be investigated, and it may lead to the formulation of different criteria for different seasons.
- A substantial amount of effort will be put into identifying and quantifying stressor-response relationships in these waters using correlative/regression analysis. Considering the difficulty of working across the surface-subsurface interfaces in deriving nutrient loading estimates, as well as effects of these loads, the authors have done a good job of addressing these challenges. This section could however benefit from closer process/response connections (including applying modeling approaches) to receiving estuarine and coastal waters.
- The results of a distribution approach (p. 108) are sensitive to the distribution of sites along the disturbance gradient. If a larger proportion of the samples are from more disturbed sites, then using the lower percentile to set the criteria will result in a higher number than if a larger proportion of the samples are from less disturbed sites. Some

Comment: I don't understand -- Chl-a is a biological endpoint! Do they mean macroinvertebrate index???

Comment: maybe we are confusing terminology. The conceptual model shows Chl-a as a response measure, but not a true biological endpoint. The latter being the biological communities to be protected. So, yes, the Panel is saying that Chl-a is a step away from a direct measure of phytoplankton or faunal community balance or “health”

Comment: This whole issue of seasonal differences is of concern for the entire document. Something should be said about this in the response to the first charge question.

requirements with respect to the distribution of sites along the disturbance gradient are needed.

3.5. South Florida Marine Waters

3.5.1. Delineation and Data Sources

Charge Question 5(a). Are the data sources identified in Section 2.4 and 5.5 appropriate for use in deriving numeric criteria in South Florida's marine waters (as discussed in Chapters 2 and 5)? Is the SAB aware of additional available, reliable data that EPA should consider in delineating or deriving criteria for South Florida's marine waters? Please identify the additional data sources.

One general recommendation is that the waters currently termed "marine waters" in the EPA document be changed to "South Florida coastal and estuarine waters" to be consistent with the use of the terms throughout the rest of the document.

The southern part of Florida has a quite different nutrient regime than other parts of the state, with respect to its highly oligotrophic nature and the degree to which conditions can be rapidly altered by upstream water management (versus nutrient regulatory) decisions. The Panel agrees that these waters should be considered separately for purposes of nutrient criteria development. However, the proposed subdivision/subclassification of South Florida estuarine and coastal waters does not clearly relate to the oceanographic circulation and degree of connectivity in the region.

Comment: Expand, and add reference(s)

The data identified in the report seemed appropriate for use in this exercise. There also are water quality data from NOAA's Atlantic Oceanographic Meteorological Laboratory (AOML) that have been collected for Florida Bay, Biscayne Bay, the Florida Keys and SW Florida Shelf for more than a decade as part of the NOAA South Florida Program (www.aoml.noaa.gov/sfp). There are some possibly significant differences between these data and the Southeast Environmental Research Center (SERC) data, which covers the same domains. For some periods in some subregions, the NOAA data were temporally more dense (bimonthly versus quarterly) in larger domains and the nutrient methodologies were more sensitive (long path length liquid wave guide) in accordance with oceanographic practice for oligotrophic open ocean waters (as established in JGOFS, GLOBEC and other international programs).

3.5.2. Approaches

<u>Charge Question 5(b).</u> EPA describes two methods in Section 5.6 for using a reference condition approach for deriving numeric criteria in South Florida marine waters (least-disturbed sites or bionomial test). Compare and contrast the ability of each approach to ensure attainment and maintenance of balanced natural populations of aquatic flora and fauna in South Florida marine waters, given currently available data?

There are two approaches to nutrient criteria being considered for South Florida coastal and estuarine waters. The first approach is to identify criteria that are inherently protective based on a statistical evaluation of data from least-disturbed sites. We note that in some of these zones least-disturbed sites may be those most distant from land-based sources; but this becomes tricky where the least-disturbed locations are seaward (e.g., on the east coast and Keys) because least disturbed may also be a result of dilution with naturally highly oligotrophic waters, and that dilution is not likely to occur nearshore in many places. Hence by including sites diluted by oligotrophic ocean water, the criteria may be overly protective. The second approach is also based on a statistical evaluation, but in this case raw data are analysed using a binomial test and two criteria are generated – an average concentration and an upper percentile concentration that is more sensitive to higher concentrations. Both approaches have merit and the Panel encourages the application of both to provide a more robust evaluation of criteria.

Comment: I tried to clarify this sentence, but I am not quite sure what point is being made.

It is also critical to address how the two approaches would be applied. For example, if a baseline (i.e., reference) condition is established using the median or geometric mean of a decade of data for the undisturbed condition, there still remains the major issue of how concentrations that exceed the criteria will be determined. Will each new year be assessed against the baseline (the approach taken with the CERP System Status Report and the SFERTF Scientific Indicators) or will five years of data be required to determine if 2 (or 3) had "exceeded" the baseline? How would the variability in the two data sets (baseline and evaluation) be incorporated? Given how variable some of these numbers are, it is a lot "weaker" (less chance of seeing a change) to ask if the means of the two datasets (in the example above, 10 and 5 yrs) differ versus whether a particular year was significantly above the baseline mean.Furthermore, there may be major ecological differences between two successive years of concentrations that exceed the criteria versus two years separated in time, and the document does not discuss this. The Panel recommends that more thought be given to these implementation issues.

Comment: less powerful? (in a statistical sense…)

The Panel recommends a reconsideration of the rationale for doing both a principle component and cluster analysis. EPA proposes to use a combination of principal component analysis (PCA) and cluster analysis to define coastal regions based on multivariate measurements with sites. As the goal of cluster analysis is precisely to identify groups of similar sites, it is unclear why PCA is being proposed in this context.

Comment: Solow placeholder. More can be added if needed.

Consider past alterations.

The coastal and estuarine waters of South Florida have experienced enormous changes over the last 100 years. (Several surveys were done beginning in the 1960's, but widespread data collection in the region really only started in the mid-1990's.) For example, the Florida Bay of the early 1900s was a true estuary with low and highly variable salinities for most of the year. Following widespread damming for the Flagler railroad, salinities were lowered throughout much of the Bay; the effect on salinity was relatively minor, but the effect on residence time was significant. Then, with canal construction from the 1920s to mid-1960s, the vast majority of the water flowing out of Lake Okeechobee is now shunted out to sea before reaching the southern-most waters of the state. Although there were a number of animal studies conducted, there were few nutrient or chlorophyll measurements made because the water was so clear that light penetrated to the bottom. In the 1970s, *Thalassia* covered Florida Bay, believed to be a result of the artificially high salinities resulting from the eastward and westward shunting of water that used to flow south into Florida Bay (citation needed). A major drought in the mid-1980s resulted in Florida Bay salinity going as high as 70, which killed off *Thalassia* and other sea grasses. Although nutrients were not a cause of the sea grass dieoff, the result was that enormous amounts of nitrogen and phosphorus that had been sequestered as detritus in the sediment were no longer protected by the dense sea grasses. A subsequent large storm event then mixed large amounts of sediment nutrients into the water column. The result was eutrophication, yet the ultimate cause was a change in salinity that killed off sea grasses years before (citations needed). Based on this brief history, the Panel has the following recommendations:

- When setting reference conditions, EPA should consider historical water management and structural changes and regional climatic variability that affect water delivery to South Florida estuaries and coastal waters.

- Sea grasses coverage and the extent of epiphytic colonization should be considered as endpoints, in addition to water column chlorophyll (see also 3.2.2).

- Salinity should be considered for its role in maintaining water quality as well as nutrients, particularly with respect to sea grasses. We note, however, that salinity is relevant (and in fact variable as a result of water management) only in a very restricted part of this domain.

Clarify geographic areas to be included.

South Florida contains a number of parks and marine protected areas. This situation has been clarified to a large degree by the formation of the National Marine Sanctuary. The document should clarify which coastal and estuarine areas will be under the jurisdiction of the EPA document under review vs. other regulations. We note that the Florida Keys National Marine Sanctuary (FKNMS) domain (and that of the three NPS parks: Biscayne, Dry Tortugas and Everglades – a.k.a. Florida Bay) are not the only federally protected waters, and there are also state protected waters of various types. It is our understanding that what is set by EPA will constitute a "de minimus" standard for these areas, which could receive additional protection. Similarly, the EPA document should clarify the relationship of the South Florida coastal and estuarine nutrient criteria to the Comprehensive Everglades Restoration Plan (CERP) and the standards being established in the courts.

Comment: What is this referring to?

3.6. <u>Downstream Protection Values</u>

3.6.1. DPVs for Estuaries

<u>Charge Question 6(a).</u> Are the methods EPA is considering for deriving downstream protection values (DPVs) for estuaries (excluding marine waters in South Florida) as described in Section 6.1-6.4 appropriate to ensure attainment and maintenance of downstream water quality standards, given available data? Please describe additional approaches and their advantages and disadvantages that EPA should consider when developing numeric criteria to protect these downstream estuarine waters (excluding marine waters in South Florida), given available data?

Rationale for DPVs

The 1972 Clean Water Act (CWA) states that:

"In designating uses of a water body and the appropriate criteria for those uses, the State shall take into consideration the water quality standards of downstream waters and shall ensure that its water quality standards provide for the attainment and maintenance of the water quality standards of downstream waters."

This provision has been the basis for ensuring that water quality standards in one state provide for attainment and maintenance of water quality standards of downstream states and Tribes. The recently published nutrient criteria for Florida's lakes and flowing waters (75 FR75762-75807, December 6, 2010) explicitly included the concept of downstream protection values (DPV) as a concentration or loading value in a stream at the point of entry into a lake, set at a value to ensure that lake nutrient criteria are attained. The rule also notes that wasteload and/or load allocations from an approved total maximum daily load (TMDL) may be used as the DPV.

In the present document, the concept of DPV is included as a means of ensuring that upstream N and P water quality criteria will be set at levels that will protect downstream estuarine designated uses. However, the entire Panel was not convinced that DPVs contribute to water quality protection beyond that which is already achieved given existing regulations for water quality standards and TMDLs. To illustrate this, consider Figure 2 below. Water quality criteria (WQC) are required for all waterbodies in this figure – the estuary and the streams. If streams A1, B1, and C1 meet their WQC, yet the estuary does not, additional pollutant load reductions to the estuary are required. These reductions could come from direct loading, the atmosphere, or the tributaries (A1, B1, C1). Standard practice is to model the estuary and watershed to determine the additional pollutant load reduction needed, and then to allocate the load reduction based on input from state and local officials. One possible result of implementation of the required load reduction is that one or more streams may require WQC more stringent than those initially established in order to attain WQC in the estuary. Regardless, this regulatory-driven analysis will achieve compliance with all WQC without the additional regulatory entity of DPV criteria.

A few additional points to think about when considering the wisdom of DPVs:

- There are other sources such as direct (e.g., groundwater and atmosphere) loadings to the estuary. How might those be addressed in the determination of tributary DPVs?
- Referring to Figure 2 (below), why should there not be DPVs for streams B2a and B2b in order to protect stream B1?
- If DPVs are implemented, the concept of equal allocation of load reductions among tributary streams should be reconsidered, as it is common practice to allow state and local governments to select the load reduction allocation strategy.

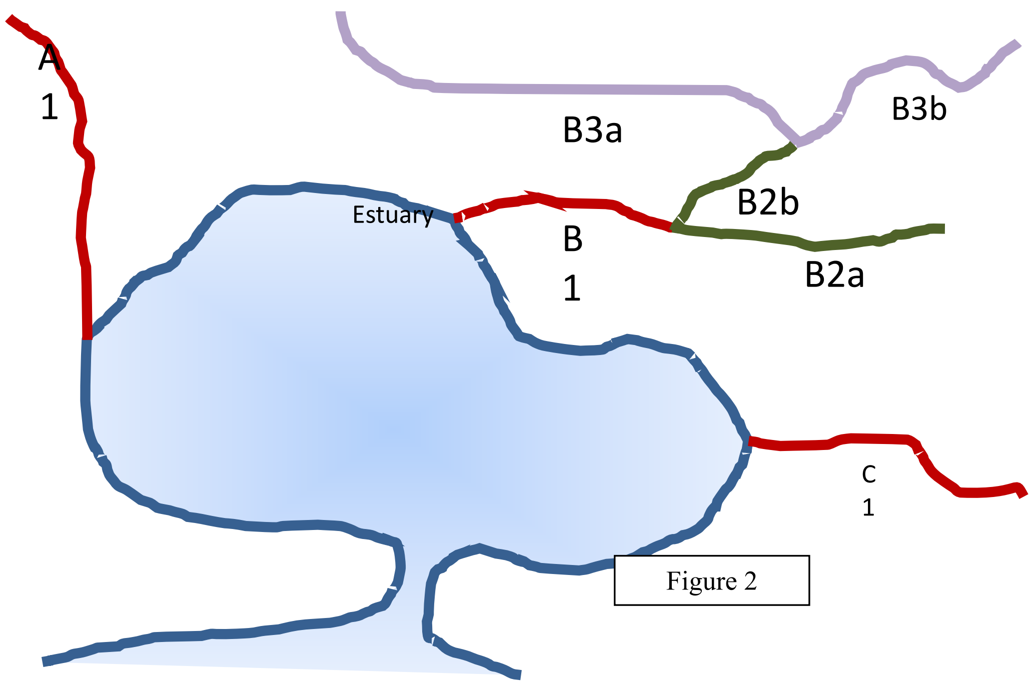

Figure 2

Approach to Setting DPVs

EPA's proposed assessment of DPVs is based on watershed modeling (to be undertaken using the LSPC model) which results in an apportioned pollutant load reduction for each tributary to the waterbody (e.g., estuary) of interest. EPA proposes to apportion the pollutant load reduction (required to achieve compliance with the waterbody water quality criterion) as an equal fractional load reduction for each tributary to the waterbody. This EPA DPV proposal for Florida appears to formalize, and unnecessarily restrict, the standard pollutant load allocation process that already occurs for TMDL pollutant load allocation when a water quality standard violation occurs.

Consider an estuary in Florida with impaired water quality; several streams that also have water quality criteria violations flow into this estuary. Water quality criteria violations must be addressed, typically with a TMDL. If, after all stream violations are successfully addressed, the estuary is still not in compliance with its water quality criterion, then additional pollutant load reductions from these streams will be necessary. At this point, the DPV LSPC model could be applied to relate the point and nonpoint sources in the entire watershed to the estuarine water quality criterion. This modeling analysis could then provide the basis for determining the necessary additional specific load reductions to achieve compliance. This is a scientifically-defensible approach and is a standard approach for load allocation or TMDL implementation.

However, the approach proposed in the EPA document requires equal allocation of the remaining pollutant load reduction. Allocation (implementation) of pollutant load reduction is normally left to state and local governments, who decide among equal allocation, minimum cost allocation, or allocation based on some other criterion acceptable to those affected. Since the watershed modeling must be undertaken anyway to determine the allowable pollutant load to achieve compliance with the waterbody water quality criterion, independent of the DPV program, and since equal allocation of the load reduction is a decision that is more appropriately made at the implementation stage, it seems that the approach proposed for DPVs is redundant and restrictive.

That said, the Panel has the following suggestions for the modeling of load reduction apportionment for upstream segments:

1. The watershed segment approach is valid, but care should be taken in selecting segments to take into account available data and other watershed characteristics such as predominant land-use.

Given a need to complete watershed modeling for the purpose of determining DPVs, the division of the watershed into segments for the purpose of predicting loadings at the “pour point” into the estuary or marine receiving waters should not be limited to simple hydrologic division of the watershed. This may conflict with the premise of using a 12-digit HUC, but the segmentation process needs to take into account predominant land uses for a segment, and those land uses that may be significantly different. For example, urban areas, with high impervious surface cover and altered stream channels, are likely to behave in a way that is distinctly different then less developed areas. Therefore, a simple model delineation of subwatersheds may not be suitable and some expert analysis and adjustment of the segments would be more appropriate.

2. The impacts of urban environments should be considered.

Urbanized areas have a distinct influence on normal stream processes given their large areas of impervious cover. In addition to changes in stream habitat, runoff from impervious surfaces as well as municipal and industrial discharges may contribute to stream nutrient loads. For this reason, the Panel recommends that large urbanized areas be given special consideration in any modeling approach that might be used to generate DPVs.

3. Given that a complete uncertainty analysis cannot be accomplished, it is essential that, in all text in the revised report where uncertainty is mentioned, readers are clearly told what is included (excluded) in any uncertainty analysis undertaken or contemplated.

4. EPA should provide justification for the choice of the LSPC model and explain why it is the most applicable model for this case.

The LSPC model is an updated version of the older HSPF program. While the model can be integrated with GIS, it is not a GIS-based approach. Numerous models exist for watershed management, physical flow, and water quality modeling that may better utilize the strengths of current GIS platforms, with some of these models having been developed by the EPA. Given the complexity of watershed modeling at the proposed scale and the complex nature of the problem being addressed, it may be prudent to invest the time in building watershed models that will be able to take advantage of a wider array of GIS-based tools and data for the current project and in future applications during implementation.

Comment: could we provide some examples here?

Comment: But the Panel is already concerned that EPA cannot accomplish its current tasks in the time available…

5. The time frame of modeling is important and should be linked to the response of the endpoints in the receiving waters.

In the EPA presentation on development of DPVs, it was indicated that adjustments for seasonal effects and flow levels are being considered. This is a very important consideration and the EPA is encouraged to analyze available data in the context of seasonal changes in the watershed and for the differences between baseflow and storm event conditions. Seasonal changes in the watershed may results from both natural processes (e.g., biotic activity) and from anthropogenic factors (e.g., agricultural practices). The differences in loadings seen during baseflow and storm events may be dramatic, with the majority of loading of TN and TP coming during a few large storm events. This is particularly true for N and P species associated with suspended sediment. Using an annual average value may grossly underpredict the impact of large storm events. Therefore, EPA should evaluate the sensitivity of the selected biological endpoints to the potential influences of shorter-term (e.g., days to weeks) events that may result in high levels of TN and TP loading to determine if annual or seasonal averages are sufficient to protect estuarine biota.

6. In-stream/watershed P transformations should be considered in more depth for streams, lakes and canals.

Species/fractions of N and P are often a part of TMDL modeling. If DPVs are to be developed in Florida, expressed as loads, and serve in a TMDL-like role, then DPVs might be expressed as nutrient fractions (for a biotic estuarine water quality criterion). In the discussion of nutrients, EPA correctly identifies the role of N species/fractions, but does not consider P species/fractions.

The dynamics of P in watersheds, lakes and canals is important to any effort to produce DPVs or similar water quality criteria. Foremost is the need to recognize the mobility, reactivity and bioavailability of the different P species: soluble reactive phosphorus (SRP); dissolved phosphorus (DP), which is the sum of SRP and total hydrolysable P (THP); and total phosphorus

(TP), which is the sum of DP and particulate phosphorus (PP). These phases exist in natural waters in varying degrees that are dependent on processes within the waterbody and on external inputs. Furthermore, transitions among forms occur during transport within watersheds and within sediments in streams, lakes and canals. Following is a brief overview of some of the extensive literature on P cycling and transformations within watersheds.

Comment: should some of this detail be moved to an appendix?

While streams are often viewed as simply a transfer mechanism for P, recent work has investigated processes that occur during transport. There are mechanisms that transform P within different physico-chemical fractions within the stream channel (Melack, 1995; Evans and Johnes, 2004; Evans et al., 2004), and the speciation of soluble P phases and fractionation of P are critical for any evaluation of transport or retention within a watershed. Various processes transform P including sorption, co-precipitation, and redox reactions (e.g., House 2003), and SRP interacts with stream sediments. Stream sediments act as both sinks and sources for SRP within the stream depending on the SRP concentration in the stream water and may change both temporally and spatially within a watershed (e.g., Jarvie et al., 2006; Ryan et al., 2007). This would suggest that EPA should evaluate existing data sets with regard to SRP and TP concentrations.

In comparing rural versus urbanized watersheds, Owens and Walling (2002) found that PP increased in stream sediments receiving point source discharge high in SRP, and that PP (inorganic and organic) may be the most significant mechanism for P transport. Up to 20% of the PP in stream sediment is likely to be easily bioavailable as inorganic P phases dominate. These mechanisms may also be active in lake or canal sediments. Given the short-term bioavailability of some fraction of the PP, it is important to evaluate TP in the context of SRP, DP and PP with some evaluation of the immediacy of the impact of each fraction.

Phosphorus retention within watersheds is typically dominated by calcite co-precipitation within bed sediment and physical trapping of sediment by reduction of flow velocity. Lake sediments may act as both sinks and sources for P cycling, with a large fraction of the inorganic P in surface sediments in equilibrium with the water column (Golterman 1995). The cycling of P is most prevalent in stratified lakes with anoxic hypolimnion, but significant cycling of P also occurs from oxic sediments (Bostrom et al., 1989; Jensen and Andersen, 1992; Rydin and Brunberg, 1998) found in nearshore environments, stream sediments and likely in canals.

P mobilization occurs under both oxic and anoxic conditions, and exchangeable and Fe-bound P are generally mobile (Rydin 2000). Organic-associated P is about 60% mobile, with greater mobility in anoxic sediments. P associated with Al and Ca is immobile and may be considered permanently bound. P release from aerobic sediments may deplete the Fe-bound P despite Fe remaining in the solid phase (Jensen and Andersen 1992). The release process involves a complex relationship between nitrate concentrations and microbial activity resulting in seasonal effect of increasing sediment P retention during winter with subsequent release during late summer and autumn. Biota also play a role in P cycling in lake sediments (e.g., bioturbation, rooted macrophytes that alter the sediment biogeochemistry). The likely lack of available data on the fractionation of P between the various physico-chemical phases will limit a detailed evaluation; however, it is important that modelling of P transport include some recognition of the biogeochemical processes involved in P cycling.

7. How are nutrients, especially P, from natural or geologic sources separated from anthropogenic sources.

Further compounding the issue of apportionment and determination of DPVs is the issue of background values for nutrients, especially P. Given that some areas of Florida have bedrock geology with high P concentrations, understanding background is critical. In watersheds where high P loadings are the result of natural factors, DPVs may not be applicable.

8. The continuum of fresh to saline waters in going from watersheds to the receiving estuarine or coastal marine waters must be considered in the process of determining DPVs.

In many instances, fresh water systems are P-limited with respect to nutrient balance and the potential for the development of eutrophic conditions. The opposite is often the case for estuarine or marine waters where N is the limiting nutrient. This raises the potential where the application of watershed water quality standards that may be focused at reducing P inputs could be protective of the watershed, but create a situation in the brackish or saline receiving waters that creates a nutrient imbalance. The development of DPVs and implementation of recent inland water criteria should address this issue.

3.6.2 DPVs for South Florida Estuarine and Coastal Waters

> Charge Question 6(b). *Are the methods that EPA is considering for deriving downstream protection values (DPVs) for marine waters in South Florida as described in Section 6.5 appropriate to ensure attainment and maintenance of downstream water quality standards, given available data? Please describe additional approaches and their advantages and disadvantages that EPA should consider when developing numeric criteria to protect downstream marine waters in South Florida, given available data?*

Unlike for estuaries in other parts of the state, the EPA document is not proposing an upstream apportionment of load reduction by stream segment because of the greatly managed hydrology in South Florida. Instead, the document proposes setting a protective load at the terminal reach of each tributary, i.e., at the point where the tributary empties into estuarine or coastal waters. The EPA document discusses several schemes for allocating acceptable estimated nutrient loads among tributaries, including allocation based on flow-weighted concentration, flow-only, or total load for each tributary.

As noted previously, DPVs appear to result in an additional regulatory entity (DPV criteria) that quantifies a tributary pollutant load that would otherwise be determined in the TMDL implementation process. If the effort is to continue, more justification and details are required for this part of the project, and consideration for temporal and land use variations are necessary. The Panel has the following additional comments:

1. Provide more information on how canals will be evaluated.

A number of primary canals empty directly into coastal waters, so it will be important to incorporate all available data on TN and TP for the terminal reach of these canals and to provide a more detailed approach on how DPV criteria will be developed.

2. The time frame of modeling is important and should be linked to the response of the endpoints in the receiving waters.

Although numeric models are not being used for the canals, the time frame of discharge and, therefore, loading rates, is still important. Given that nutrient concentrations vary widely, an effort to create DPV criteria should consider loading rates. Furthermore, given that this is a highly managed system, loading rates could be adjusted in near real-time. For example, if a large storm resulting in large discharges from canals is expected, sampling for TN and TP could occur before the storm event and loading rates calculated to protect the receiving waterbodies.

Given the wide variation in flow conditions for canals, the concentrations of nutrients in canal waters are likely highly variable. Hence average nutrient concentrations in canal waters when released to estuarine and coastal marine waters may not adequately represent the concentrations needed to protect receiving waters. If additional information is not available for nutrient concentrations in the canals, discharge of canal waters to the receiving waterbodies needs to take into consideration loading rates on a daily basis that will ensure the receiving waterbodies meet their water quality standards.

3. In-stream/watershed P transformations should be considered in more depth for streams, lakes and canals

Although less is known about P transformations within the canals of South Florida, the physical and chemical processes that control P transport within a watershed should be the same for canals. Additional consideration, however, must be given to the special situations that result as a function of the wide-ranging flow situations for the canal system. Furthermore, it is important to understand the temporal parameters and their range of variability. These factors will determine, in part, the mechanisms that are most important under different sets of flow conditions.

4. The continuum of fresh to saline waters going from watersheds to the receiving estuarine or coastal marine waters must be considered in the process of determining DPVs.

For canal waters discharging to estuarine and coastal marine waters, the issue of the continuum to fresh to saline water is the same as discussed in response to the first charge questions, above.

4. Concluding Remarks…

If any…

4.1. &&&&&

&&&&&

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APPENDIX A: Charge to the Panel

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