DEVELOPING NUTRIENT NUMERIC ENDPOINTS AND TMDL TOOLS FOR CALIFORNIA ESTUARIES: AN IMPLEMENTATION PLAN

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Technical Report 540 - August 2007

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AUGUST 29, 2007

Prepared for:
US EPA Region IX
(Contract No. 68-C-02-108)

California State Water Resources Control Board
Planning and Standards Implementation Unit

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EXECUTIVE SUMMARY

Eutrophication is one of the top three leading causes of impairments of the nation's waters, with demonstrated links between anthropogenic changes in watersheds, increased nutrient loading to coastal waters, harmful algal blooms, hypoxia, and impacts on aquatic food webs. These ecological impacts of eutrophication of coastal areas can have far-reaching consequences, including lowered fishery production, loss or degradation of seagrass/submerged aquatic vegetation (SAV), smothering of benthic organisms, nuisance odors, and impacts on human and marine mammal health. These modifications have significant economic and social costs. In California, the impacts of nutrient loading on estuaries and coastal waters have not been well monitored and no statewide criteria or guidance exist to manage these impacts. Without management actions to reduce anthropogenic nutrient loads and other factors contributing to eutrophication, symptoms are expected to develop or worsen in the majority of systems, due to projected population increases in coastal areas.

United States Environmental Protection Agency (USEPA) Region IX and the California State Water Resources Control Board (SWRCB) have previously developed a technical approach and framework for developing numeric nutrient endpoints (NNEs) for California estuaries (USEPA 2007). The stated goal of this effort is to develop a set of numeric endpoints and Total Maximum Daily Load (TMDL) tools that can be used to address impacts from eutrophication through the water quality programs of the SWRCB, Regional Water Quality Control Boards and the regulated community. The USEPA (2007) presented a scientific framework to support the development of numeric endpoints for a suite of biological response indicators (e.g., algal biomass, dissolved oxygen, water clarity, etc.) that are directly linked with estuarine beneficial uses. The framework also highlighted data gaps and research recommendations critical for the development of numeric endpoints and TMDL tools (i.e., watershed loading and estuarine load-response models). These tools are critical because they provide the linkage between the numeric endpoints, which are based on biological response indicators, and watershed nutrient loads and other factors controlling eutrophication in estuaries.

The purpose of this document is to outline an implementation plan to address these critical data gaps in California estuaries and move forward with NNE and TMDL tool development. The implementation plan proposes a statewide approach to NNE development, with regionalization of numeric endpoints possible if warranted by differences in ecology.

Tables ES-1 and ES-2 summarize available science to support development of nutrient numeric endpoints and TMDL tools. These tables show that:

- Sufficient science exists on the physiological impacts of dissolved oxygen to proceed with interim guidance for this biological response variable
- Further research and monitoring are required to provide suggested numeric endpoints for the additional biological response variables
- Critical data gaps to support the development of watershedloading models include wet-weather loading data from a variety of urban, agricultural and undeveloped land uses
- The development of statistical estuarine load-response models by estuarine classes requires a regional dataset of watershed nutrient loading and biological response variables in a representative sample of estuaries

- Cost-effective development of dynamic-simulation models of load-response (i.e., estuarine water quality models) requires data on the major processes responsible for transformation, uptake and release of nutrients in estuaries.
- A summary of data relevant to California estuaries is needed to further prioritize funding for research

Table ES-1. Summary of existing science that would support NNE development in California estuaries.

Variable	Status of Science	Recommended Action
Surface Water Dissolved Oxygen	Physiological impacts of low DO on pelagic and benthic species is well documented, though not necessarily for resident species of California estuaries. Need to interpret existing data for relevance to resident species and identify data gaps.	Assemble expert panel to recommend key indicator species and review literature to assess applicability of existing effects data on indicator species. Set preliminary thresholds and define critical data gaps for refinement of thresholds.
Macroalgal Biomass	Current science does not provide clear linkage to impairments of beneficial uses. Need to investigate impacts of macroalgal biomass on benthic infaunal communities and SAV, with ultimate linkage to food and habitat for estuarine birds and fish.	Create research plans to study macroalgal impacts on benthic infauna and SAV. Initiate research after approval of study plans by stakeholder group.
Harmful Algal Blooms	Environmental conditions for toxin production are poorly understood. Impacts of toxins are not adequately documented, such as adverse effects on viability, growth, fecundity, etc. Need better understanding of linkage with nutrient loading. No clear linkage has been established with management controls.	Assemble a statewide group of experts to identify major data gaps and identify implementation plan needed to address them.
Aesthetics	No documentation of linkage of macroalgal biomass, surface water hypoxia, or sediment sulfide production with foul odor, bad taste, or unsightliness.	Develop study plan to establish, via focus group, levels of odor or extent of macroalgal blooms associated with poor aesthetics.
Water Clarity	Thresholds exist for east coast estuaries. Need to determine thresholds that are appropriate for SAV communities in California estuaries.	Develop a study plan to address identified data gaps with respect to light requirements of West Coast SAV species.
Surface Water Chl a	No consensus on appropriate thresholds in literature. This indicator is particularly relevant for San Francisco Bay.	Assemble panel of experts to review literature and existing monitoring data to recommend course of action.
Nuisance SAV Biomass[1]	Studies of coastal lagoons have noted overproduction SAV in oligohaline areas.	Review existing monitoring data to understand prevalence of nuisance SAV in lagoons. If prevalence is great, prioritize indicator as high course of action.

[1]This indicator applies specifically to overproduction of SAV (e.g. *Ruppia* sp.) in oligohaline lagoons. Impacts to seagrass (e.g. *Zostera* spp.) are assessed through macroalgal biomass and water clarity.

Table ES-2. Summary of status of science and recommended actions for TMDL tool development.

Tool/Data	Status of Science	Recommended Action
Watershed Loading Models (applies to both dynamic simulation and simple spreadsheet model)	Development of watershed-loading models hampered by lack of wet-weather loading data from a variety of land uses.	Two actions are recommended: 1) acquire data characterizing wet-weather loads of nitrogen and phosphorus from a variety of urban, agricultural and undeveloped land uses. This dataset will be used to create watershed-loading models and statistical load-response models for TMDL development; and 2) develop regionally calibrated dynamic-simulation models of nutrient wet-weather loading.
Simple Spreadsheet Model (to predict annual watershed nutrient loads)	Simple spreadsheet tool to predict annual nutrient loads can provide cost-effective means to improve understanding of harmful algal blooms and estuarine eutrophication. Accuracy and precision in predicting annual loads needs to be better understood.	Compare precision of simple spreadsheet model and dynamic-simulation model for predicting annual loads to estuaries.
Statistical Load – Response Models	Statistical load-response models exist for some east coast estuaries; these models are not relevant for California because of major differences in dominant primary producer communities and hydrology. Need to better investigate major variables controlling biological response to loads in California estuaries. Monitoring data that relates total nutrient loads to estuarine biological response are only known to be currently available for two estuaries (SF Bay and Upper Newport Bay). Data are currently being generated for 5 more lagoons. A more comprehensive data set is required to develop load-response models.	Acquire a datasets that relates nutrient loads to estuarine biological response. Two approaches are suggested: 1) develop a statistical load-response model for an existing long-term data set and 2) collect data on an estuarine biological response through a probability-based sample of estuaries. Both approaches will be used to understand the spatial and temporal variability in estuarine load-response.
Dynamic-Simulation Models (of estuarine water quality)	Data are generally lacking on the major processes responsible for uptake, transformation, and release of nutrients in California estuaries.	Conduct a comprehensive literature review and survey of existing research in California to identify key processes that need to be modeled, locate existing and relevant data, and identify major data gaps. This report can be used to identify priorities for future funding by entities such as Sea Grant, USEPA, etc.

The implementation plan for NNE development proposes a four-phased approach that would allow for: 1) an initial statewide outreach and preparation of supporting technical documents, 2) the development of NNEs for dissolved oxygen, 3) Research to address data gaps for development of NNEs for the additional biological response variables, and 4) TMDL tool development. The conceptual framework, decisions regarding technical approaches, prioritization of studies supporting numeric endpoint and tool development, and selection of numeric endpoints would be done through a stakeholder-driven process representing the regulators, the regulated community, environmental NGOs, scientific experts, and the general public.

ACKNOWLEDGEMENTS

The authors would like to acknowledge Clayton Creager and Sujoy Roy of TetraTech for their comments on various drafts of this document; Jeff Crooks for data from Tijuana Estuary, Tammy Small for Elkhorn Slough data, Mark Page, Jennifer Wolf, and Richard Ambrose for data from San Diegito Lagoon, and Barry Lindgren and Doug Gibson for data from San Eljio Lagoon. Tijuana River Estuary and Elkhorn Slough continuous monitoring data are publicly available from the National Estuarine Research Reserve website: http://cdmo.baruch.sc.edu/.

1 INTRODUCTION AND PURPOSE OF DOCUMENT

Globally, cultural eutrophication of estuaries and coastal waters is a significant environmental issue with demonstrated links between anthropogenic changes in watersheds, increased nutrient loading to coastal waters, harmful algal blooms, hypoxia, and impacts on aquatic food webs (Valiela et al. 1992, Kamer and Stein 2003). These ecological impacts of eutrophication of coastal areas can have far-reaching consequences, including fish-kills and lowered fishery production (Glasgow and Burkholder 2000), loss or degradation of seagrass and kelp beds (Twilley 1985, Burkholder et al. 1992, McGlathery 2001), smothering of bivalves and other benthic organisms (Rabalais and Harper 1992), nuisance odors, and impacts on human and marine mammal health from increased frequency and extent of harmful algal blooms and poor water quality (Bates et al. 1989, Bates et al. 1991, Trainer et al. 2002). In addition to ecological impacts, these modifications have significant economic and social costs, some of which can be readily identified and valued, while others are more difficult to assess (Turner et al. 1998). According to the United States Environmental Protection Agency (USEPA), eutrophication is one of the top three leading causes of impairments of the nation's waters (USEPA 2001).

In California, the impacts of nutrient loading on estuaries and coastal waters have not been well monitored, with the notable exception of San Francisco Bay (Cloern 1982, Cloern et al. 1985, Cloern 1991, 1996, Cloern 1999). However, the National Oceanic and Atmospheric Administration (NOAA) National Estuarine Eutrophication Assessment Report, which characterized the trophic status and sensitivity of 18 of California's 209 estuaries and coastal lagoons, noted a high degree of eutrophication in estuaries along the central and southern California coast (Bricker et al. 1999). These estuaries tend to have restricted circulation and high nutrient inputs. Without management actions to reduce anthropogenic nutrient loads and other factors controlling eutrophication, symptoms are expected to develop or worsen in the majority of systems, primarily due to projected population increases along the coastal areas.

The USEPA and the California State Water Resources Control Board (SWRCB) have previously developed a technical approach and framework for developing numeric nutrient endpoints (NNEs) and tools to link these endpoints with management strategies to control eutrophication in California estuaries (USEPA 2007). The goal of this effort is to develop a set of numeric endpoints and Total Maximum Daily Load (TMDL) tools that can be used to address impacts from eutrophication through the water quality programs of the SWRCB, Regional Water Quality Control Boards (RWCBs), and the regulated community. USEPA (2007) presented a scientific framework to support the development of numeric endpoints for a suite of biological response indicators, and highlight data gaps and research recommendations for their development. The next step in the process is to articulate a plan to develop numeric endpoints and support the efficient and cost-effective development of TMDL tools.

The purpose of this document is to outline an implementation plan to address critical data gaps. Section 2 reviews the conceptual framework proposed in USEPA (2007). Section 3 defines each candidate biological response indicator, reviews existing literature that supports NNE development in California estuaries, and summarizes the state of science that supports TMDL tool development. Section 4 discusses technical approaches to setting numeric endpoints, and provides an overview of numeric criteria for eutrophication in existing state programs. Section 5 presents a recommended implementation plan to develop numeric endpoints and TMDL tools for California estuaries.

2 FRAMEWORK FOR DEVELOPMENT OF NUTRIENT NUMERIC ENDPOINTS (NNE) AND TMDL TOOLS IN ESTUARIES

2.1 BACKGROUND FOR DEVELOPMENT OF NNES IN ESTUARIES

The USEPA initiated the National Nutrient Management Strategy in 1998 to begin addressing the pervasive impacts of excessive nutrient loading to both fresh and marine waters (Greening and Elfring 2002). A primary objective of the strategy was to develop numeric nutrient criteria to measure the progress of the management decisions. To achieve this objective, the USEPA issued a series of technical guidance manuals for the development of nutrient criteria.

Initial national guidance on nutrient criteria development advocated the use of a statistical approach to establish thresholds based on the nutrient concentrations in surface waters (USEPA 1986). In this approach, reference conditions were based on 25th percentiles of all nutrient concentration data including a comparison of reference condition for the aggregate ecoregion versus the subecoregions. These 25th percentile values were characterized as criteria recommendations that could be used to protect waters against nutrient over-enrichment (USEPA 2000).

Several studies have since demonstrated the shortcomings of using ambient nutrient concentrations alone to predict eutrophication in streams (Welch et al. 1989, Fevold 1998, Chetelat et al. 1999, Heiskary and Markus 2001, Dodds et al. 2002) and estuaries (Cloern 2001, Dettman et al. 2001, Kennison et al. 2003). Use of ambient, surface water nutrient concentrations is generally not effective for assessing eutrophication and subsequent impacts on beneficial use because ambient concentrations reflect the biological processing that has already occurred. In addition, biological response to nutrient loading (e.g., algal productivity) depends on several mitigating factors, such as morphology, light availability, and biological community structure. Thus high nutrient concentrations are not a definitive indicator of eutrophication, and low concentrations do not necessarily indicate absence of eutrophication.

The "Nutrient Criteria Technical guidance Manual: Estuarine and Coastal Waters" was released by the USEPA in October 2001. The USEPA Region IX had already convened the Regional Technical Advisory Group (RTAG) and the State Regional Board Technical Advisory Group (STRTAG) to serve as a forum for collaboration among stakeholders, agencies, and all nine RWQCBs. RTAG and STRTAG focused on the development of nutrient criteria for fresh waters. In 2006, the STRTAG adopted the California Nutrient Numeric Endpoint framework. The development of NNEs for fresh waters proceeded prior to estuaries with the understanding that endpoints for streams could have potential downstream impacts on estuaries.

Representatives of the RWQCBs TMDL programs joined the STRTAG in 2005, combining their efforts with criteria and standards to develop a consistent and scientifically defensible approach to the development of NNEs (USEPA 2005). This collaboration allows for the refinement and evaluation of numeric endpoints on a regional basis prior to individual RWQCB's consideration for formal adoption as Basin Plan water quality objectives. The USEPA developed a conceptual framework for development of NNEs in estuaries (2007) based on the framework for streams (USEPA 2005).

2.2 CONCEPTUAL FRAMEWORK

The purpose of developing NNEs for California estuaries is to provide the SWRQCB and the RWQCBs with a scientifically-defensible framework that can serve as guidance for adopting regulatory numeric criteria. This framework is founded on an evaluation of risk relative to designated beneficial uses. Essentially, the objective is to control levels of excess nutrient loads such that the risk or probability of impairing the designated uses is minimized. If the nutrients present – regardless of actual magnitude – have a low probability of impairing use, then water quality standards can be considered met.

The California NNE framework for estuaries, based on the conceptual approach for streams, has three organizing principals (USEPA 2005):

- *Biological response indicators provide a more direct risk-based linkage to beneficial uses than nutrient concentrations alone*

  Except in extreme cases, nutrients themselves do not impair beneficial uses. Rather, biological response to nutrient loading impairs uses. Instead of setting criteria solely in terms of nutrient concentrations, it is preferable to use an analysis that takes into account the risk of impairment. The NNE framework needs to target information on biological response indicators, such as dissolved oxygen (DO), macroalgal biomass and percent cover, surface water phytoplankton biomass (e.g., chlorophyll a, water clarity), benthic algal biomass (sediment chlorophyll a), submerged aquatic vegetation (SAV) density and percent cover, aesthetics (e.g., foul odors, unsightliness), and harmful algal blooms. These biological response indicators provide a more direct risk-based linkage to beneficial uses than the ambient nutrient concentrations or nutrient loads. Given this approach, it is critical that tools be developed that link the biological response indicators back to nutrient loads.

- *A weight-of-evidence approach with multiple indicators will produce NNEs with greater scientific validity*

  The use of multiple indicators in a weight-of-evidence approach provides a more robust means to assess ecological condition and determine impairment than nutrient concentrations alone. This approach is similar to the multimetric index approach, which defines an array of metrics or measures that individually provide limited information on biological status, but when integrated, functions as an overall indicator of biological condition (Gibson et al. 2000).

- *For many of the biological indicators associated with nutrients, no scientific consensus exists on a target threshold that results in impairment*

  Site-specific factors often play a major role in determining biological response to nutrient loading. For this reason, there may not be clear scientific consensus on a target threshold associated with impairment for many of the biological indicators of eutrophication.

To address the problem of a lack of scientific consensus on thresholds of impairment, and to accommodate site-specific differences among estuaries, the California framework proposes NNEs that serve as general guidance for the RWQCBs. These endpoints can then be translated into water quality criteria for individual basin plans, accommodating changes made if necessary to reflect site-specific conditions. To this end, the framework classifies waterbodies into the three Beneficial Use Risk Categories (BURC; USEPA 2005) . These categories are as follows:

- BURC I: In these waterbodies, beneficial uses are sustained and are not exhibiting impairment due to nutrients;
- BURC II: Beneficial uses may be impaired; additional information and analysis may be needed to determine the extent of impairment and whether regulatory action is warranted; and
- BURC III: These waterbodies are exhibiting impairment due to nutrients; regulatory action is warranted.

For a given beneficial-use designation, there is general consensus that the BURC I/II boundary represents a level below which nutrients will not present a significant risk of impairment. There is also general consensus that the BURC II/III boundary represents a value that is sufficiently high that risk of use impairment by nutrients is probable. Within BURC II, additional water body-specific cofactors may be brought into the analysis to determine an appropriate target. Table 2-1 provides a summary of potential biological response indicators for eutrophication and their correspondence to estuarine beneficial uses. Ultimately, the goal is to propose preliminary numeric endpoints (i.e., BURC thresholds) for each of the biological response indicators using literature sources, monitoring data, and expert opinion. These values may change among ecoregions within California. BURC thresholds for each biological response indicator can be converted to nutrient concentration targets appropriate for assessment, permitting, and TMDLs by using simple load-response models or more complex dynamic-simulation models for biological responses in estuaries. Depending on use, data availability, and economic impact of the decision, other, more detailed and site-specific tools may be appropriate for correlating secondary indicator targets to nutrient loading targets.

2.3 ESTUARINE BENEFICIAL USES

State policy for water quality control in California is directed toward achieving the highest water quality consistent with maximum benefit to the people of the state. Beneficial uses define the resources, services, and qualities of the state’s aquatic systems that guide protection of water quality; they also serve as a basis for establishing water quality objectives. Several studies have linked nutrient enrichment to beneficial-use impairment. Table 2-1 lists the key biological response indicators that may result in impairment to specific estuarine beneficial uses. It should be noted that waterbodies are generally assigned multiple beneficial uses.

Table 2-1. Summary of response variables and their applicability to estuarine beneficial uses. DO = Dissolved Oxygen.

Beneficial Use	Key Biological Response Indicators						
	Low DO	Macro-algal Blooms	Poor Water Clarity	Harmful Algal Blooms	Foul Odor, Bad Taste, Unsightliness	Increased Chlorophyll a	Increased Nuisance SAV/Loss of Beneficial SAV
Area of Special Biological Significance (ASBS)	X	X	X	X	X	X	X
Ocean, Commercial, and Sport Fishing (COMM)	X	X		X	X	X	X
Estuarine Habitat (EST)	X	X	X	X		X	X
Marine Habitat (MAR)	X	X	X	X		X	X
Fish Migration (MIGR)	X	X		X		X	X
Preservation of Rare and Endangered Species (RARE)	X	X	X	X		X	X
Water Contact Recreation (REC1)		X	X	X	X	X	X
Non-contact Water Recreation (REC2)		X			X		
Shellfish Harvesting (SHELL)	X	X		X	X	X	X
Fish Spawning (SPAWN)	X	X		X		X	X
Wildlife Habitat (WILD)	X	X	X	X		X	X

2.4 ESTUARINE CLASSIFICATION

Estuaries within California are highly variable in how they respond to nutrient loading due to differences in physiographic setting, salinity regime, frequency and timing of freshwater flows, magnitude of tidal forcing, sediment load, stratification, residence time, and denitrification. This combination of factors results in differences in dominant primary producer communities (i.e., phytoplankton, macroalgae, benthic algae, submerged aquatic vegetation, emergent macrophytes). It also creates variability in the pathways that control how nutrients cycle within the estuary. As a result of these differences, the USEPA (2007) recommended the creation of numeric endpoints for distinct subclasses of estuaries. This recommendation was based on the fact that: 1) the dominant primary producer communities among the estuary types are different, and 2) among estuary types which share a common primary producer community, biological interactions vary such that the thresholds for "impairment," would be expected to be significantly different. Based on this premise, California's proposed classification scheme includes eight estuarine subclasses (USEPA 2007):

- Protected Embayment - This estuary type is typically semi-enclosed by land, dominated by subtidal or deepwater habitat. The inlet mouth is not restricted and is continuously open to tidal exchange.
- Perennially Tidal Lagoon - These estuaries are dominated by shallow subtidal and intertidal habitat and have a long residence time due to the restricted width of the mouth. The inlet is continuously open to tidal influence year round, either by natural forces or anthropogenic management.

- Seasonally Tidal Lagoon - These estuaries are dominated by shallow subtidal and intertidal habitat, with a long residence time due to a seasonally restricted width of mouth or mouth closure. They support fresh to brackish submerged aquatic vegetation and emergent marsh for part of the year when the mouth is closed.
- Nontidal Lagoon- These estuaries are dominated by shallow subtidal and intertidal habitat, with a long residence time due lack of surface water connection with coastal ocean. The salinity regimes of these lagoons can be fresh to brackish due to limited input of ocean water during spring tides, storm surges or advective exchange through a sand berm.
- River Mouth Estuary- This class of estuaries is the terminus of high flow, perennial river systems as they enter the coast. The estuarine portion is the mixing zone at the mouth of the river. These systems are characterized by 1) ebb-dominated flows, 2) estuarine mixing zone found within the channel during dry season, and 3) continuous disturbance of flats discourages growth of emergent vegetation during average flow years
- Creek Mouth Estuary- These estuaries are the terminus of small creeks that drain to the coast. The estuarine portion is the mixing zone at the mouth of the river. Many of these systems experience seasonal closure of their mouths due to the longshore drift of sand. These systems typically do not support emergent marsh, but are important habitats for steelhead trout and tidewater goby.
- Open Embayment/Coastal Estuarine Front- These estuaries are the open bays and zones of freshwater plumes that are found on the continental shelf. The limits of the estuary are defined by the mixing zone of freshwater with salt water. For coastal estuarine fronts, these boundaries are highly elastic/transient and can vary depending on location and magnitude of river plumes, currents, and upwelling.
- San Francisco Bay Estuary- The size and complexity of the San Francisco Bay/ Estuary makes it unique among California estuaries. This estuary contains at least four compartments that are hydrologically distinct from each other.

Due to these differences in structure and function among estuarine subclasses, different biological response variables and specific thresholds may vary, not only between subclasses, but also within a subclass across ecoregions. Thus, classifying California's estuaries into estuarine types with similar morphology and environmental setting can aid in the development of specific NNEs for each subclass. Table 2-2 shows the number of California's estuaries by estuarine subclass.

The task of developing NNEs is complicated by the lack of a common definition of "estuary" among the six Regional Water Quality Control Boards (Regions 1, 2, 3, 4, 8, and 9). This has resulted in inconsistencies in the list of estuaries subject to nutrient criteria. A type of estuary that is excluded in beneficial-use designation by one Regional Board may be included in another. Other types of estuaries, such as creek mouths, have been largely excluded from consideration. This lack of consistency will lead to an arbitrary application of numeric endpoints across the state.

Table 2-2. Summary of the total number of estuaries in each class for Northern, Central, and Southern California. The number of open embayments are not included in the table.

Estuarine Class	**Northern California**	**Central California**	**Southern California**	**TOTAL**
River Mouth Estuaries	16	3	7	**26**
Seasonally Tidal Lagoon	10	36	12	**58**
Perennially Tidal Lagoon	4	5	23	**32**
Nontidal Lagoon	5	10	11	**26**
Creek Mouth Estuaries	54	30	54	**138**
Protected Embayment	8	3	12	**23**
San Francisco Bay				**1**
TOTAL	**97**	**87**	**119**	**304**

3 NUTRIENT NUMERIC ENDPOINT AND TMDL TOOL DEVELOPMENT: STATUS OF SCIENCE

3.1 PURPOSE OF REVIEW

The purpose of this section is to: 1) review the literature that supports the selection of numeric endpoints for each biological response variable where estuarine beneficial uses become threatened and 2) summarize the state of science with respect to nutrient TMDL tool development. The review focused on three priority classes of California estuaries where most of the TMDL activity is occurring: perennially tidal lagoons, seasonally tidal lagoons, and protected embayments.

3.2 STATUS OF SCIENCE TO SUPPORT NUMERIC ENDPOINT DEVELOPMENT

3.2.1 Candidate Biological Response Indicators and Relationship to Watershed Nutrient Loads

Cultural eutrophication is the overproduction of organic matter, caused by excessive anthropogenic nutrient inputs. Cultural eutrophication of estuarine environments results in the alteration of the structure and/or function of estuarine environments, and can impact estuarine beneficial uses (Figure 3-1; (Howarth 1988, Nixon 1995, Cloern 2001). A common pattern of estuarine biological response to nutrient loading exists: primary production increases (macroalgae, phytoplankton, benthic algae, etc.), organic matter derived from this primary producer biomass decomposes, and dissolved oxygen (DO) is depleted in the surface waters and sediments (Cloern 2001, Bricker et al. 2003). As eutrophication and benthic respiration increases, the DO content of bottom waters decreases, and areas of hypoxia expand in both in area and duration (D'Avanzo and Kremer 1994, Nixon 1995, Diaz 2001, Howarth et al. 2002). Organic carbon loading to sediments increases sediment concentrations of ammonia and toxic metals, increasing the potential for toxicity to develop in the sediments. In some estuaries, planktonic algal blooms occur for months at a time, blocking sunlight to submerged aquatic vegetation and promoting hypoxia in bottom waters. Macroalgal mats can cover intertidal flats and shallow subtidal areas, altering sediment chemistry, reducing populations of benthic infauna, and hindering the forage of fish and birds for food sources within or upon mudflats. Furthermore, harmful algal blooms can produce toxins that harm estuarine fauna and damage commercial fisheries. Hypoxic/anoxic events degrade essential estuarine habitats and this degradation has the potential to reduce the capacity of the system to support production of submerged aquatic vegetation (SAV), fish, and invertebrates (Diaz 2001, Rabalais et al. 2002).

These common biological response indicators to eutrophication are thus the basis for a multi-metric approach to setting NNEs:

- Surface water DO
- Macroalgal biomass and percent cover
- Surface water phytoplankton biomass (e.g., chlorophyll a)
- Water clarity
- Nuisance SAV density and percent cover

- Aesthetics (e.g., foul odors, unsightliness)
- Harmful algal blooms

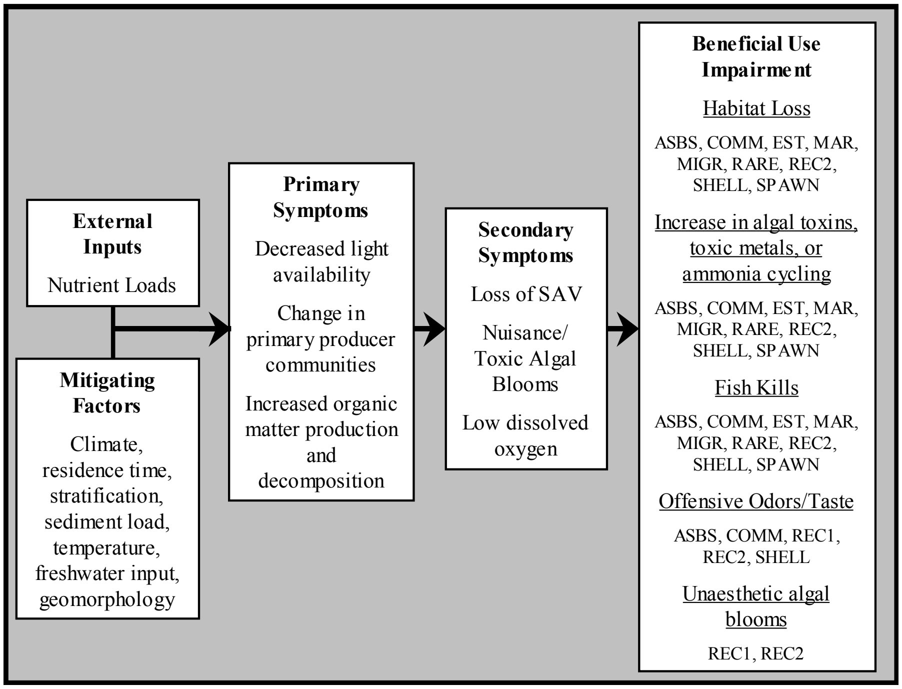

Figure 3-1. Conceptual model of primary and secondary symptoms of eutrophication and impacts on beneficial uses adapted from Bricker et al. (2003) and Mills et al. (2003).

While many of the biological response variables listed are common to most estuaries, the interplay between the various estuarine responses to eutrophication (e.g., changes in nutrient ratios and cycling, changes in biological community structure and distribution.) can be subtle and complex (Cloern 1999, Krause-Jensen et al. 1999, Smith et al. 1999). For example, nutrient loading into an estuary may result in a macroalgal bloom, but because the flushing rate of the estuary is rapid, hypoxia does not develop. In another estuary, a long hydraulic residence time results in continuous hypoxia. This variability highlights the fact that multiple indicators are required to assess eutrophication and that thresholds for specific indicators may vary by estuarine class. For this reason, caution must be used in the assumptions that dictate the degree to which indicators and specific thresholds apply to an estuary.

Management of eutrophication typically occurs through: 1) Reduction of watershed nutrient loads to the estuary, and 2) management of some of the factors that control the biological response of an estuary to nutrient loads (i.e., tidal prism, freshwater residence time, estuary bathymetry and habitat distribution). Although eutrophication can sometimes be alleviated by manipulation of these endogenous factors, reduction of nutrient loading is the primary management strategy to minimize eutrophication.

It is important to note that nutrient loading is not one of the recommended numeric endpoint indicators. Determination of the specific nutrient loading rates that result in eutrophication is complicated by site-specific attributes that serve to modulate the biological response to nutrient enrichment. Consequently, there is a large range in the sensitivity of estuaries to increased nutrient loading (Cloern 2001). Since the numeric endpoints would be based on biological response indicators rather than nutrient loads, it is critical that TMDL tools be developed that facilitate the linkage of biological response back to nutrient loads. Section 3.3 gives the status of the science with respect to the development of these tools for California estuaries; the implementation plan (Section 5) provides recommendations on specific studies and activities to develop these tools.

The sections below summarize the state of the science with respect to thresholds impacting estuarine beneficial uses for each biological response indicator.

3.2.2 Dissolved Oxygen

Oxygen is necessary to sustain the life of all fishes and benthic invertebrates. Oxygen enters the estuarine environment either directly from the atmosphere or from photosynthesis of organisms living within the surface waters or on the sediments of estuaries. Once dissolved, oxygen can be mixed into the bottom waters where it can support the life of benthic organisms. Eutrophication produces excess organic matter that fuels the development of hypoxia (i.e., low surface water DO concentration) as that organic matter is respired (Diaz 2001). When the supply of oxygen from the surface waters is cut off (via stratification), or the consumption of oxygen exceeds the resupply (via decomposition of excessive amounts of organic matter), oxygen concentrations can decline below the limit for survival and reproduction of benthic (bottom-dwelling) or pelagic (water column dwelling) organisms (Stanley and Nixon 1992, Borsuk et al. 2001, Diaz 2001). Changes in the survival and reproduction of benthic and pelagic organisms can result in a cascade of effects including loss of habitat and biological diversity, development of foul odors and bad taste, and altered food webs (USEPA 2007).

Among the biological response indicators under consideration, literature on the effects of hypoxia on estuarine organisms stands out as the best documentation of impacts to estuarine fisheries and benthic communities (USEPA 2007). The threshold for acute or chronic effects of low DO varies by organism and by life stage because some life stages are more sensitive to low DO than others (e.g., juveniles versus adults). Some empirically determined thresholds for low DO in pelagic species are listed in Table 3-1 and for benthic organisms in Table 3-2.

Table 3-1. Dissolved Oxygen thresholds for protection of pelagic species adapted from USEPA (2003).

Criteria Components	Species	Concentration	Duration	Source
Protection Against Growth Effects	resident tidal and freshwater species	> 4.8 mg/L	-	USEPA (2000)
	striped bass	> 3 to 4 mg/L	-	Brandt et al. (1998)
	juvenile striped bass	> 4 mg/L	-	Kramer (1987); Breitburg et al. (1994)
	chinook salmon	>4 mg/L	-	Geist et al. (2006)
	juvenile spot and Atlantic menhaden	>1.5 mg/L	2 weeks	McNatt and Rice (2004)
Egg/Larval Recruitment Effects	resident tidal and freshwater species	>4.6 mg/L; >3.4-3.5 mg/L; >2.7-2.8 mg/L	30 - 40 days 7 days instantaneous minimum	USEPA (2000)
	estuarine species	3 mg/L 1.7 mg/L	30 days instantaneous minimum	Chesney and Houde (1989); Breitburg (1994); USEPA (2000)
	naked goby males abandon nests	0.36 mg/L	-	Breitberg (1992)
Survival	juvenile/adult fish species	>2.3 mg/L	24 hours	USEPA (2000)
	resident tidal and freshwater species	>5.5 mg/L >4 mg/L >3 mg/L	30 days 7 days instantaneous minimum	USEPA (2000)
	striped bass	>5 mg/L	72 hours	Krouse (1968)
	striped bass (preferred concentrations)	>6 mg/L	-	Krouse (1968); Hawkins (1979); Christie et al. (1981); Rothschild (1990)
	juvenile striped bass (preferred concentrations)	> 5 mg/L	-	Kramer (1987); Breitburg et al. (1994)
Reduced Survival	copepods	<0.86-1.3 mg/L	24 hours	Stalder and Marcus (1997);
	striped bass	> 3 mg/L	72 hours	Krouse (1968)
Protection of Early Life Stages	resident tidal and freshwater species	>6 mg/L >5 mg/L	7-day mean instantaneous minimum	USEPA (1986)
	striped bass	>5 mg/L	-	Krouse (1968)
Protection Against Effects on Threatened/ Endangered Species	shortnose sturgeon	>5 mg/L >3.5 mg/L >3.2 mg/L >4.3 mg/L	30 days 6 hours 2 hours 2 hours	Secor and Niklitschek (2004); Niklitschek (2001); Secor and Gunderson (1998); Jenkins et al. (1993), Campbell and Goodman (2004)
	steelhead, coho, and chinook salmon	>5 mg/L	--	Campbell et al. (2001)
Effects on Total Fish Catch	total fish biomass	< 3.7 mg/L	-	Simpson (1995)
	total fish species richness	< 3.5 mg/L	-	Simpson (1995)
	decline in abundance and diversity of catch	< 2 mg/L	-	Howell and Simpson (1994)
50% Mortality	hogchoaker, northern sea robin	0.5-1 mg/L	24 hours	Reviewed in Breitburg et al. (2001)
	tautog, windowpane flounder adults	>1 mg/L	24 hours	Reviewed in Breitburg et al. (2001); Pihl et al. (1992)
	summer flounder, pipefish, striped bass adults	1.1-1.6 mg/L	24 hours	Reviewed in Breitburg et al. (2001); Pihl et al. (1992); Poucher and Coiro (1997); USEPA (2000)

Criteria Components	Species	Concentration	Duration	Source
	skilletfish, naked goby, silverside larvae	1-1.5 mg/L	24 hours	Breitburg (1994); Poucher and Coiro (1997)
	red drum, bay anchovy, striped blenny larvae	1.8-2.5 mg/L	24 hours	Saksena and Joseph (1972); Breitburg (1994); Poucher and Coiro (1997)
	Acartina tonsa and *Eurytemora affinis*	0.36-1.4 mg/L	2 hours	Vargo and Sastry (1977)
	naked goby larvae	>~1.5 mg/L	24 hours	Brietberg (1994)
	sea nettle	>0.7 mg/L	96 hour	Brietberg et al. (1997)
	juvenile stripped bass	>2 mg/L	24 hours	Coutant (1985)
	menhaden	0.70 mg/L 0.88 mg/L 1.04 mg/L	2 hours 24 hours 96 hours	Burton et al. (1980)
	spot	0.49 mg/L 0.67 mg/L 0.70 mg/L	2 hours 24 hours 96 hours	Burton et al. (1980)
100% Mortality	*Acartina tonsa* (copepods)	<1.43 mg/L	24 hours	Stalder and Marcus (1997)
	Acartina tonsa and *Oithona colcarva*	<2 mg/L	24 hours	Roman et al. (1993)
	copepods	<0.71 mg/L	24 hours	Stalder and Marcus (1997)
	naked goby males	<0.24 mg/L	-	Breitberg (1992)
	juvenile striped bass	< 3 mg/L	-	Chittenden (1972); Coutant (1985); Krouse (1968)
	rainbow trout	< 3.1	-	Matthews and Berg (1997)
Food web shift from predation on fish larvae (naked gobi) by juvenile and adult fish (striped bass) to invertebrates (sea nettles)		2.5-3.3	-	Brietberg et al. (1997)
Reduced Activity (swimming)	naked goby	1.5 mg/L	-	Brietberg et al. (1997)
	stripped bass	<3 mg/L	-	Breitberg et al. (1994)
Avoidance Threshold/ Emigration	weakfish, blue crab, shrimp, croaker and spot	2.1-2.3 mg/L	-	Eby and Crowder (2002)
	menhaden and southern flounder	2.6 mg/L	-	Eby and Crowder (2002)
	pinfish and silver perch	4.1-4.2 mg/L	-	Eby and Crowder (2002)
	zooplankton	< 1 mg/L	-	Roman et al. (1993)
	Gulf of Mexico demersal fish	<2 mg/L	-	Craig et al. (2001)
	logfish	2.0 mg/L	-	Beitinger & Pettit (1984)
	cod and whiting	25-40% saturation	-	Gray (1992)
	benthic flatfishes	40-50% saturation	-	Gray (1992)

Table 3-2. Benthic organism response to bottom water dissolved oxygen concentrations adapted from USEPA 2003.

Response	Dissolved Oxygen	Species	Reference
Avoidance			
Infaunal Swimming	1.1	*Paraprionospio pinnata*	Diaz et al. (1992)
	0.5	Nereis succinea	Sagasti et al. (2000)
Epifaunal Off Bottom	0.5	*Neopanope sayi*	Sagasti et al. (2000)
	0.5	*Callinectes sapidus*	Sagasti et al. (2000)
	1	*Stylochus ellipticus*	Sagasti et al. (2000)
	1	*Stylochus ellipticus*	Sagasti et al. (2000)
	0.5	*Dirodella obscura*	Sagasti et al. (2000)
	1	*Cratena kaoruae*	Sagasti et al. (2000)
Fauna, Unable to Leave or Escape, Initiate a Series of Sublethal Responses			
Cessation of Feeding	0.5	*Balanus improvisus*	Sagasti et al. (2000)
	0.6	*Streblospio benedicti*	Llanso (1991)
	1	*Loimia medusa*	Llanso and Diaz (1994)
	1.1	*Capitella sp.*	Forbes and Lopez (1990)
Decreased Activities Not Related to Respiration	0.5	*Balanus improvisus*	Sagasti et al. (2000)
	0.5	*Conopeum tenuissimum*	Sagasti et al. (2000)
	0.5	*Membranipora tenuis*	Sagasti et al. (2000)
	1	*Cratena kaoruae*	Sagasti et al. (2000)
	1	*Stylochus ellipticus*	Sagasti et al. (2000)
	1	*Streblospio benedicti*	Llanso (1991)
Cessation of Burrowing	1.1	*Capitella sp.*	Warren (1997)
Emergence from Tubes or Burrows	0.1-1.3	*Ceriathiopis americanus*	Diaz unpublished data
	0.5	*Sabellaria vulgaris*	Sagasti et al. (2000)
	0.5	*Polydora cornuta*	Sagasti et al. (2000)
	0.7	*Micropholis atra*	Diaz et al. (1992)
	1	*Hydroides dianthus*	Sagasti et al. (2000)
	10% saturation	*Nereis diversicolor*	Vismann (1990)
Siphon Stretching into Water Column	0.1-1.0	*Mya arenaria, Abra alba*	Jorgensen (1980)
Siphon or Body Stretching	0.5	*Molgula manhattensis*	Sagasti et al. (2000)
	0.5	*Diadumene leucolena*	Sagasti et al. (2000)
Floating on Surface of Water	0.5	*Diadumene leucolena*	Sagasti et al. (2000)
Formation of Resting Stage	0.5	*Membranipora tenuis*	Sagasti et al. (2000)
	0.5	*Conopeum tenuissimum*	Sagasti et al. (2000)

The response of individual organisms to low DO will depend on the intensity, duration and frequency of hypoxic events (Rabalais et al. 2002). Some species have developed adaptations to deal with short-term hypoxia (Kramer 1983, Beitinger and Pettit 1984, Dauer et al. 1992, Eby and Crowder 2002). However, longer term DO stress will lead to recruitment and growth effects (Breitburg 1992, Crocker and Cech 1997), emigration from low DO areas and over crowding of remaining habitat (Coutant 1985, Breitburg 2002), species composition shift (Howell and Simpson 1994, Swason et al. 2000), and eventually death (Burton et al. 1980). Fish species are more sensitive to low DO than benthic infauna. Furthermore, early life stages (egg/larval recruitment, growth effects, etc.) are more sensitive than adult fish with respect to survival or avoidance of low DO areas. However, protective thresholds for such early life stages are only applicable during a portion of the year during spawning and nursery stages. Thus, the most sensitive threshold would protect recruitment and early life stages during spawning and nursery seasons. This threshold could then decrease to a level that would still promote adult fish survival for the rest of the year. Species-specific thresholds in Tables 3-1 and 3-2 can be utilized to set DO criteria based on the beneficial uses in a lagoon or protected embayments.

Currently, literature on the response of West Coast estuarine habitats and species to hypoxia is limited. Most studies have focused on species response in East Coast estuaries (specifically Chesapeake Bay) and the Gulf Coast (Tables 3-1 and 3-2). Numeric endpoints based on data for East Coast species could potentially be used to extrapolate thresholds for key California species. However, such extrapolations would need to be generated by a panel of California benthic and pelagic experts familiar with similarities between east and west coast species. Such an expert panel could compile a list of the types of species, native of California estuaries, that can be considered as candidate indicator species for each estuarine class (and potentially by ecoregion). The panel could then review the literature to determine the extent to which existing data on effects could be applied to these indicator species, and identify where additional data must be gathered. From the list of potential indicator species, key species could be identified based on sensitivity to low DO. If no data currently exits for key indicator species, preliminary DO thresholds could be extrapolated using data from species within the same genus or family for which data already exists. Additional experiments could then be conducted to directly assess DO thresholds for the different life stages of key California indicator species. An initial list of fish native to California estuaries that could potentially serve as indicators for hypoxia is presented in Table 3-3. This is a priority data gap that will need to be closed in order to move forward with the implementation of DO as one of the biological response indicators of the NNE framework.

Table 3-3. Fish native to California estuaries that could serve as indicator species for hypoxia.

Type	Species
Small Fish (less able to migrate out of low dissolved oxygen areas)	Intertidal: arrow goby, longjaw mudsucker, tidewater goby (endangered) Subtidal: cheekspot goby, shadow goby, Pacific staghorn sculpins
Poor Swimmers (in SAV or lgae habitat)	Snubnosed pipefish, banded pipefish, bay pipefish, California killifish, California halibut, diamond turbot
Good Swimmers (able to migrate out of low dissolved oxygen areas)	Surf perches, shiner perch, yellowfin croaker, spotfin croaker, topsmelt, jacksmelt, striped mullet, deepbody anchovy, slough anchovy, spotted sand bass, white seabass
Juveniles (in lagoons only)	Barred sand bass, kelp bass
Sharks and Rays	Leopard shark, round stingray, bat rays
Recreational/Commercial Fish	White seabass, California halibut, sand bass, jacksmelt, yelowfin croaker, striped mullet, spotfin croaker
Anadromous Fish/Endangered Species	Steelhead trout

Appendix A provides an analysis of DO monitoring data for four perennially tidal lagoons in California. This appendix details how effects-based thresholds can be applied to monitoring data to estimate the probability that specific estuaries would have a DO concentration below that in which beneficial uses are sustained. Because no specific criteria are available for California estuaries, established criteria from the Chesapeake Bay Program are applied (*c.f.* Section 4).

3.2.3 Macroalgal Blooms

Increased eutrophication often results in a shift in primary producer communities (Hernandez et al. 1997, Valiela et al. 1997), such as the proliferation of macroalgae. These algae are typically filamentous (sheet-like) forms (e.g., Ulva, Cladophora, Chaetomorpha) that can accumulate in extensive thick mats over the seagrass or sediment surface. Although macroalgae are a natural component of these systems, their proliferation due to nutrient enrichment reduces habitat quality in four ways: 1) increased respiration at night and large oxygen demand from decomposing organic matter, 2) shading and out-competing submerged aquatic vegetation, and 3) impacts on the density of benthic infauna, which are a principle food source for birds and fish, and 4) increases in poor aesthetics or odor. Among the literature on impacts of eutrophication on West Coast estuaries, the proliferation of macroalgae, particularly in shallow subtidal and intertidal environments, is one of the most commonly cited (Fong et al. 1998, Kamer et al. 2001, Kennison et al. 2003).

Respiration may reduce the DO content in estuarine waters at night (e.g., Peckol and Rivers (1995)), while decomposition may cause a large microbial oxygen demand both day and night (Sfriso et al. 1987). The presence of macroalgae in estuarine environments can alter the nutrient and DO concentrations significantly on a diurnal scale. In the daytime DO concentrations in mats can be more than 200% saturation due to photosynthesis, but during the night DO concentrations can become anoxic as organic matter is respired (Krause-Jensen et al. 1999). Growth of macroalgal mats can absorb nutrients from the water column and act as an effective filter for the nutrient flux from the sediments (Krause-Jensen et al. 1999, Fong and Zedler 2000), and the decomposition of algae can release nutrients to the watercolumn. Thus, the presence of algal blooms can effectively disrupt the nutrient balance of estuarine systems (Zimmermann and Montgomery 1984, Smith and Hollibaugh 1989). Furthermore, the presence of large macroalgal and epiphyte blooms have been noted to have a dramatic effect on the frequency and extent of low DO events in several estuarine systems (Sfriso et al. 1987, D'Avanzo and Kremer 1994, Hernandez et al. 1997). However, there are no specific threshold values that link macroalgal biomass to specific DO levels because other factors, such as water residence time and bathymetry, may alter estuarine DO response to macroalgal blooms. Because of advancements in DO monitoring technology, it is easier to directly measure DO content than to try to extrapolate DO levels based on macroalgal biomass alone. However, assessment of macroalgal thickness and cover can be used as a biological response variable in conjunction with DO.

The role of macroalgal canopies in SAV habitat loss worldwide may equal or surpass that of phytoplankton and epiphytes in some systems (Hauxwell et al. 2001). Macroalgae have light requirements that are significantly less than either seagrasses or perennial macroalgae, and can shade perennial macrophytes such as seagrasses, contributing to their decline (Markager and Sand-Jensen 1990, Duarte 1991). SAV tend to dominate in nutrient poor environments because they can access sediment sources of nutrients; however, when nutrient concentrations increase, fast-growing macroalgae and phytoplankton can out compete SAV and become dominant (Harlin and Thorne-Miller 1981, Krause-Jensen et al. 1999). The presence of excessive macroalgae in an estuary can result in shading of SAV beds to the point where the plants can no longer sustain adequate photosynthesis. In one study, a macroalgal canopy thickness of approximately 9 - 12 cm is the threshold at which eelgrass beds begin to decline due to shading (Hauxwell et al. 2001). Holmquist (1997) observed impacts to SAV from shading with macroalgal patches of

$0.25\ m^2$ or greater. Furthermore, macroalgal canopies can alter water and sediment redox conditions, resulting in sulfide and ammonium toxicity (Valiela et al. 1992, D'Avanzo and Kremer 1994, Krause-Jensen et al. 1999) and alteration in available nutrient concentrations for seagrasses (McGlathery et al. 1987, Bierzychudek et al. 1993, Krause-Jensen et al. 1996); Table 3-4). Addition of macroalgae to seagrass environments leads to the removal of most seagrass structure within six months (Holmquist 1997). Although the structurally complex mats formed by drifting algae can provide short-term habitat enhancement for some fauna, long-term effects on fauna are negative due to the degradation of more stable seagrass beds (Holmquist 1997). In the long term, sustained eutrophication can be expected to lead to complete replacement of seagrass habitat by unvegetated coarser sediments, occasionally covered by green macroalgal blooms and dominated by opportunistic invertebrate taxa (Cardoso et al. 2004). Several studies have been conducted identifying solar irradiance thresholds that some California SAV (e.g., *Zostera marina*) require for survival (Table 3-4). However, more comprehensive studies should be conducted to determine specific levels of macroalgal biomass thickness, extent, and duration result in SAV loss for each estuarine class. Such studies are particularly important for protected and open embayments where water clarity is sufficient to sustain SAV meadows.

Increased coverage by macro-algal mats has significant impacts on the distribution and abundance of benthic invertebrates and their fish and shorebird consumers (Raffaelli et al. 1999). The presence of macroalgae alters benthic macroinvertebrate community composition, in some cases severely decreasing populations during years of hypoxia/anoxia (Lopes et al. 2000, Muli and Mavuti 2001, Powers et al. 2005). Such shifts in macroinvertebrate communities can affect the foraging strategies of fish and shorebirds within the estuary. Demersal fish have been observed to shift their diet away from benthic macroinvertebrates to less nutritional items such as plant and detrital material following hypoxia induced by macroalgal blooms (Powers et al. 2005). Bird populations have been observed to decline under increasing nutrient loading to estuaries (Fernandez et al. 2005) and in association with the decline of eelgrass habitat (Seymour et al. 2002). Macroalagal mats present on mudflats have been shown to alter the foraging habitats of shorebirds. For example, algal mats physically interfere with the deep probing action of godwits and they avoid such areas; whereas redshank appeared not to be deterred by algal cover and preferentially utilized the infauna associated with algal mats as prey (Lewis and Kelly 2001). Loss of foraging area beneath algal mats could result in increased competition among bird populations, which may force birds either to emigrate or to die (Santos et al. 2005). Thresholds for macroalgal abundance have not, to our knowledge, been systematically explored (i.e., macroalgal mat thickness, aerial extent, and duration). Most of the studies available are observational related to the presence/absence of mats (Table 3-4). In order to establish macroalgal biomass thresholds that are protective of California benthic invertebrates, and by association California shorebirds and fish populations, studies would need to be conducted to understand the extent and duration of macroalgal blooms that result in a decrease in benthic infauna.

Table 3-4. Literature-based thresholds for impacts of macroalgae on the estuarine environment

Impact	Thresholds	Source
SAV shading	>8-18 % surface light >10-20% surface light > 5-6 hours of light saturated irradiance	Dunton (1994) Duarte (1991) Zimmerman et al. (1994, 1995b)
SAV shading	Macroalgae patches >0.25 m diameter	Holmquist (1997)
SAV shading	Macroalgal canopy height > 9-12 cm	Hauxwell et al. (2001)
Benthic community composition change*	Macroalgal mat presence	(Lopes et al. 2000, Muli and Mavuti 2001, Powers et al. 2005)
Fish foraging shift*	Macroalgal mat presence	(Powers et al. 2005)
Bird foraging shift*	Macroalgal mat presence	Lewis and Kelly (2001)

*Observational studies of macroalgal mat presence/absence

3.2.4 Aesthetics (Foul Odors, Bad Taste, Unsightliness)

The presence of eutrophication in coastal estuaries can create a number of impacts on the community that can lead to economic losses (Beloff and Beaver 2000, Gillen et al. 2006). Foul odors can lead to reduced enjoyment of surrounding public lands and private homes because of an inability to recreate out doors or utilize open air ventilation of facilities, etc. The overgrowth of macroalgae reduces the aesthetic beauty of waterfront or water-view property, loss of enjoyment of fishing, swimming, boating, and other water sports, and a reduction in REC2 beneficial uses. Foul odors can result in negative health symptoms (e.g., headache, nausea, loss of appetite, etc.); however, there is no quantifiable or established dose-response relationship between odor and negative health impacts (Gillen et al. 2006). Assessing thresholds for impairment of aesthetics is encumbered by the subjectivity of both odor perception and personal opinions of unsightliness. No specific thresholds for odor related to eutrophication or level of macroalgal biomass considered to be unaesthetic were located in the literature. However, several studies exist in which chemical or animal odor concentration, duration, and character impacted human well-being (annoyance, health, etc.; (Nimmermark 2004, Gillen et al. 2006). These assessments typically utilized a panel of individuals who would be tested for their threshold of detection, ranking of odors as pleasant or unpleasant, and the level of annoyance in response to specific concentrations of odors. Similar survey studies could be conducted for California estuaries to identify odor and macroalgal biomass levels that would impair REC2 beneficial uses.

3.2.5 Chlorophyll a and Phytoplankton Community Structure

Chlorophyll a is a measure used to indicate the amount of microscopic algae, called phytoplankton, growing in a water body. High concentrations are indicative of problems related to the overproduction of algae. Impairment issues related to phytoplankton blooms are similar to macroalgal blooms. In some estuaries, nutrients cause dense phytoplankton blooms for months at a time, blocking sunlight to submerged aquatic vegetation. Decaying microalgae from the blooms use oxygen that was once available to estuarine fauna. In other estuaries, these or other symptoms may occur, but less frequently, for shorter periods of time, or over smaller spatial areas. In still other estuaries, the assimilative capacity (ability to absorb nutrients) may be greatly reduced, though no other symptoms are apparent. These eutrophic symptoms are indicative of degraded water quality conditions that can adversely affect the use of estuarine resources, including commercial and recreational fishing, boating, swimming, and tourism. Eutrophic symptoms may also cause risks to human health, including serious illness and death that result from the consumption of shellfish contaminated with algal toxins, from direct exposure to waterborne or airborne toxins, or from contact with enteric bacteria that flourish under eutrophic conditions.

Nutrient over-enrichment can also change ecological structure through mechanisms other than anoxia and hypoxia. Phytoplankton species have wide differences in their requirements for and tolerances of major nutrients and trace elements (Atkinson and Smith 1983, Arrigo et al. 1999, Geider and LaRoche 2002). Some species are well adapted to low-nutrient conditions where inorganic compounds predominate, whereas others thrive only when major nutrient concentrations are elevated or organic sources of nitrogen and phosphorus are present. Uptake capabilities of major nutrients differ by an order of magnitude or more, allowing the phytoplankton community to maintain production across a broad range of nutrient regimes. A decrease in silica availability in an estuary and the trapping of silica in upstream freshwater ecosystems can occur as a result of eutrophication; thus, nitrogen and phosphorus over-enrichment occurs. This decrease in silica often limits the growth of diatoms or causes a shift from heavily silicified to less silicified diatoms (Rabalais et al. 1996). Given these changes in the cycling of nitrogen, phosphorous, and silica, it is no surprise that the phytoplankton community composition is altered by nutrient enrichment (Jorgensen and Richardson 1996). The consequences of changes in phytoplankton species composition on grazers and predators can be great, but in general are poorly studied.

Setting specific criteria for chlorophyll a is encumbered by the fact that there are no clear linkages between phytoplankton biomass and nutrient over-enrichment, and because multiple factors contribute to the amount of phytoplankton biomass present in a system. For example, despite high concentrations of nitrogen and phosphorus, the San Francisco Bay estuary has a relatively small phytoplankton biomass (Cloern 1982, 1996). The suspected reasons for the low chlorophyll a concentrations are grazing by benthic filter feeders (Cloern 1982) and light limitation due to high sediment loads (Cloern 1996, 1999). Thus, in order to set thresholds for chlorophyll a for California estuaries, more data would need to be collected to establish relationships between nutrient loading, chlorophyll a, and other indicators including water clarity, benthic community composition, and competition with macroalgal biomass.

3.2.6 Harmful Algal Blooms

Harmful algal bloom (HAB) phenomena take a variety of forms. In estuarine or marine environments, one major category of impact occurs when toxic phytoplankton are filtered from the water as food by shellfish that then accumulate the algal toxins to levels that can be lethal to humans or other consumers. Phytoplankton blooms consisting of toxic species of the diatom genus *Pseudo-nitzschia*, which are a common occurrence along the western US coast, fall into this category (Villac et al. 1993, Fryxell et al. 1997). Members of this genus are known producers of the neurological toxin domoic acid (DA) which, when accumulated through trophic activities, has lead to sickness or mortality in sea mammals, seabirds and humans (Amnesic Shellfish Poisoning (ASP); (Bates et al. 1989, Scholin et al. 2000). Other poisoning syndromes have been given the names paralytic, diarrhetic, and neurotoxic shellfish poisoning (PSP, DSP, and NSP). Whales, porpoises, seabirds, and other animals can be victims as well, receiving toxins through the food web from contaminated zooplankton or fish. At least 1,500 km^2 along the southern California coastline were affected by a toxic event in May/June of 2003 when some of the highest particulate DA concentrations reported for US coastal waters were measured inside the Los Angeles harbor. Overall, DA poisoning was implicated in greater than 1,400 mammal stranding incidents within the Southern California Bight during 2003 and 2004. These events do not adequately document the scale of toxic HAB impacts, as adverse effects on viability, growth, fecundity, and recruitment can occur within different trophic levels, either through toxin transmitted directly from the algae to the affected organism or indirectly through food web transfer.

Harmful algal blooms are not only a problem in marine systems. Recent research has linked health problems and ecological problems to blue-green algae (also known as Cyanobacteria) blooms that occur in fresh – brackish water environments, such as lakes, nontidal lagoons, and the tidal freshwater portions of estuaries. Blue-green algae blooms are common in the U.S. and are most frequently associated with eutrophication and nutrient enrichment from sewage treatment plants and agricultural runoff. Most forms of blue-green algae float at the surface and are most prevalent during the warmest times of the year. As a result they are a very common source of complaints from boating, fishing, and swimming enthusiasts, and are considered a nuisance form of algae. Also, they are frequently associated with taste and odor problems at water treatment plants. Cyanobacteria can also produce toxins that, in high concentrations, have caused deaths in South America and Asia; in the US, they have been associated with waterfowl kills and health problems in people and animals that have come in contact with them.

Considerable research has been conducted in an effort to understand the environmental factors that promote toxic blooms of various harmful algal species. Through these studies, coastal upwelling and river runoff have been implicated as factors that may create physical and chemical conditions (e.g., high nutrient concentrations) that are conducive to promoting phytoplankton blooms (Bates et al. 1999, Trainer et al. 2002). However, linking these processes to blooms of *Pseudo-nitzschia* species and to toxin production has been problematic. Not all *Pseudo-nitzschia* species are capable of producing DA, and toxic species do not produce DA constitutively. Laboratory studies have demonstrated that toxin production in some species of *Pseudo-nitzschia* may increase under silicate or phosphate limitation (Bates et al. 1991, Fehling et al. 2004). In addition, DA can chelate iron and copper, and thus the molecule may affect trace metal acquisition or metal detoxification by phytoplankton (Rue and Bruland 2001, Wells et al. 2005). Thus, the scenario(s) under which *Pseudo-nitzschia* blooms and DA is produced in nature may be varied and complicated, making it difficult to develop a strategy to mitigate the occurrence of these events.

At present there are no established thresholds for HABs in California estuaries outside of their impacts on water column and benthic DO levels. Further studies are required to understand the conditions under which HABs will generate toxins and the linkage to nutrient loading.

3.2.7 Submerged Aquatic Vegetation

Worldwide, seagrasses -- otherwise known as submerged aquatic vegetation (SAV) -- provide important habitats for pelagic and epibenthic species in coastal ecosystems. SAV is often degraded or destroyed by cultural eutrophication (Cardoso et al. 2004, Lee et al. 2004). The distribution and abundance of SAV is largely controlled by light availability (Backman and Barilotti 1976, Zimmerman et al. 1995a, Zimmerman et al. 1995b). Four consequences of eutrophication can result in the decline of SAV: 1) increase in nuisance macroalgae biomass (covered in previous section), 2) phytoplankton bloom, 3) increase of the cover of epiphytes, and 4) increases in toxic metals and sulfide in the sediments from excessive organic matter loading (Cummins et al. 2004).

Phytoplankton blooms increase the turbidity of the water column, thus reducing the light penetration through the water column to benthic SAV communities. Once patches of bare sediment are exposed, turbidity in the water column can increase as the action of tidal currents, wind stress, and storms continuously resuspend sediment (Robblee et al. 1991). This can further exacerbate a decline in SAV communities.

Epiphytes are algae that grow on the surfaces of plants or other objects. Epiphytic macroalgal biomass increases in response to nutrient availability and becomes more abundant on seagrass leaves in eutrophic waters. Subsequently, this leads to light attenuation at the leaf surface, as well as reduced gas and nutrient exchange (San-Jensen 1977, Tomasko and Lapointe 1991). This can cause losses of submerged aquatic vegetation by encrusting leaf surfaces and thereby reducing the light available to the plant leaves.

Macroalgae, epiphytes, and phytoplankton blooms increase organic matter delivery to sediments, often settling in large aggregations over SAV. As this organic matter decomposes, it creates low DO environments around the SAV roots, resulting in sediment hypoxia (Robblee et al. 1991, Cummins et al. 2004). Lowered oxygen concentrations may increase energy costs for SAV associated with oxygen translocation from aboveground biomass to roots (Pregnall et al. 1984). Foliar or root nutrient uptake may be affected due to altered concentrations of nutrients in the water column and at the sediment-water interface when macroalgae are present (Pregnall et al. 1984, Bierzychudek et al. 1993). High concentrations of ammonium and sulfide even in water around SAV have been shown to be toxic (Koch et al. 1990, vanKatwijk et al. 1997, Hauxwell et al. 2001). Experimentally, elevated sediment sulfide associated with excessive organic matter accumulation has been shown to reduce both light-limited and light-saturated photosynthesis of SAV, as well as to increase the minimum light requirements for survival (Goodman et al. 1995).

A number of factors have been outlined in the literature as leading to declines in SAV habitat related to cultural eutrophication and associated light limitation and bottom water anoxia. Critical conditions that lead to the decline in SAV habitat in response to eutrophication are listed in Table 3-5. These critical conditions can be considered when establishing DO and macroalgal biomass thresholds, though more research is required on specific threshold values in California lagoons before specific numeric endpoints can be set. Because the quantitative determination of SAV coverage and health is costly and time intensive we do not recommend using this as a biological response variable. However, periodic assessment of SAV habitat in lagoons can be used as a qualitative measure of improvement in water quality in California coastal lagoons. As water quality improves in coastal lagoons and the system become less eutrophic, the aerial extent and habitat quality of seagrass beds should improve as well. SAV habitat regeneration can thus serve as a qualitative indicator of the success of restoration efforts.

Alternatively, water clarity could serve as a response variable that, when applied at appropriate thresholds, would be protective of seagrass beds. As noted above, increased turbidity from sediment loads and increased water column productivity (chlorophyll a) can result in light stress of underwater seagrass beds. Water clarity can be used as a proxy for light availability. Sunlight hits the water surface and is partially reflected back at the water surface. The rest of that light is attenuated within the water column above SAV habitat by particulate matter (chlorophyll a, and total suspended solids), by dissolved organic matter, and by the water itself. The light that actually makes it to the SAV is further attenuated by epiphytes living on the seagrass leaves (USEPA 2003). Thresholds for light attenuation from the literature are presented in Table 3-5, and these values can be translated into a water column light attenuation coefficient (or Secchi depth) for criteria development.

It should be noted that the above discussion is limited to beneficial SAV habitat; other SAV can grow to levels that can impair beneficial uses in an estuary, particularly in seasonally tidal lagoons. Such species are mostly brackish (e.g., *Ruppia maritima*) and can increase in abundance under nutrient enrichment to dominate other seagrass communities (Johnson et al. 2003, Sutula et al. 2004). As biomass from nuisance SAV decays it will ultimately result in low DO conditions in bottom waters in the same way as macroalgal blooms (Sutula et al. 2004). As salinity regimes change seasonally, die-offs of nuisance SAV can cause a catastrophic hypoxic event. Biomass of nuisance SAV could serve as an additional indicator of impairment; however, more research will need to be conducted to determine at what biomass quantity nuisance SAV begins to impair beneficial uses (macroinvertebrate, fish, bird, and seagrass beds).

Table 3-5. Literature-based thresholds leading to declines in SAV habitat.

SAV Species	Factor Leading to Decline in SAV Habitat	Thresholds	Source
Halodule wrightii *Zostera marina*	Light Limitation	>8 - 18 % surface light >10 - 20% surface light >5 - 6 hours of light saturated irradiance	Dunton (1994) Duarte (1991) Zimmerman et al. (1994, 1995b)
Vallisneria americana	Light Limitation	Low Salinity: 9% surface light High Salinity: 14% surface light	Dobberfuhl (2007)
Halodule wrightii *Zostera marina*	Light Limitation	Minimum growing season light requirement: 20% surface light	Steward et al. (2005)
Seagrass meadow: *Thalassia testudinum* (turtle grass), *Halodule wrightii* (shoal grass), *Syringodium filiforme* (manatee grass)	Macroalgal Biomass Cover	Macroalgae patches >0.25 m diameter	Holmquist (1997)
Zostera marina	Macroalgal Canopy Height	Canopy height > 9-12 cm	Hauxwell et al. (2001)
Zostera noltii	Ammonia Toxicity	16 um in 16 days instantaneous 200 um	Brun et al. (2002)

SAV Species	Factor Leading to Decline in SAV Habitat	Thresholds	Source
Zostera marina	Ammonia Toxicity	25 um in 5 weeks 125 um in 2 weeks	van Katwijk et al. (1997)
Zostera marina	Sulfide Toxicity/ Dissolved Oxygen Concentrations	water column DO < 7.3 kPa (30 - 35% saturation)	Pedersen et al. (2004)
P. pectinatus	Sulfide Toxicity	0.48 - 1.27 mg/g sediment sulfide	Van Wijck et al. (1992)
Zostera marina	Sulfide Toxicity	< 0.4 mM sediment sulfide	Goodman et al. (1995)
Zostera marina	Sulfide Toxicity	<2 mM sediment sulfide	Pregnall et al. (1984); Smith et al. (1988)
SAV habitat	Sediment Grain Size	<20% silt and clay	Koch (2001)
SAV habitat	Sediment Organic Carbon	<5 %	Koch (2001)

3.2.8 Summary of Existing Science and Recommendations Actions to Develop NNEs

For the particular estuarine subclasses of interest, protected embayments, perennially tidal, and seasonally tidal lagoons, some biological response indicators may be better suited than others. Thus, initial research efforts should be aimed at developing these “high priority” indicators. Additional indicators can be added to the suite as time and funding allow. Table 3-6 summarizes the status of the science for each of the potential biological response variables, recommended action, and the priority that should be given to each.

Table 3-6. Summary of status of science on biological response variables and recommended course of action with respect to Phase I implementation.

Variable	Status of Science	Recommended Action
Surface Water Dissolved Oxygen	Physiological impacts of low DO on pelagic and benthic species is well documented, though not necessarily for resident species of California estuaries. Need to interpret existing data for relevance to resident species and identify data gaps.	Assemble expert panel to recommend key indicator species and review literature to assess applicability of existing effects data on indicator species. Set preliminary thresholds and define critical data gaps for refinement of thresholds.
Macroalgal Biomass	Current science does not provide clear linkage to impairments of beneficial uses. Need to investigate impacts of macroalgal biomass on benthic infaunal communities and SAV, with ultimate linkage to food and habitat for estuarine birds and fish.	Create research plans to study macroalgal impacts on benthic infauna and SAV. Initiate research after approval of study plans by stakeholder group.
Harmful Algal Blooms	Environmental conditions for toxin production are poorly understood. Impacts of toxins are not adequately documented, such as adverse effects on viability, growth, fecundity, etc. Need better understanding of linkage with nutrient loading. No clear linkage has been established with management controls.	Assemble a statewide group of experts to identify major data gaps and identify implementation plan needed to address them.
Aesthetics	No documentation of linkage of macroalgal biomass, surface water hypoxia, or sediment sulfide production with foul odor, bad taste, or unsightliness	Develop study plan to establish, via focus group, levels of odor or extent of macroalgal blooms associated with poor aesthetics.
Water Clarity	Thresholds exist for east coast estuaries. Need to determine thresholds that are appropriate for SAV communities in California estuaries.	Develop a study plan to address identified data gaps with respect to light requirements of West Coast SAV species.
Surface Water Chl a	No consensus on appropriate thresholds in literature. This indicator is particularly relevant for San Francisco Bay.	Assemble panel of experts to review literature and existing monitoring data to recommend course of action.
Nuisance SAV Biomass[1]	Studies of coastal lagoons have noted overproduction SAV in oligohaline areas.	Review existing monitoring data to understand prevalence of nuisance SAV in lagoons. If prevalence is great, prioritize indicator as high course of action

[1]This indicator applies specifically to overproduction of SAV (e.g. *Ruppia* sp.) in oligohaline lagoons. Impacts to seagrass (e.g. *Zostera* spp.) are assessed through macroalgal biomass and water clarity.

3.3 Status of Science to Develop Nutrient TMDL Tools

3.3.1 Linking Biological Response Indicators to Watershed Nutrient Loads: The Importance of TMDL Tools

Management of eutrophication typically occurs through: 1) Reduction of watershed nutrient loads to the estuary, 2) management of some of the factors that control how the biological response of an estuary to nutrient loads (tidal prism, freshwater residence time, estuary bathymetry and habitat distribution, etc.; Table 3-7). Although eutrophication can sometimes be alleviated by manipulation of these endogenous factors, reduction of nutrient loading is the primary management strategy to reduce eutrophication. Use of biological response indicators as numeric endpoints assumes that tools are available to provide linkage with these major strategies to manage eutrophication.

Table 3-7. Summary of major factors controlling estuarine biological response to nutrient loads adapted from NAS 2000.

Factor	Summary of How Factor Controls Estuarine Biological Response
Physiographic Setting	The physiographic setting of an estuary describes its general landform, landscape context and hydrology (e.g., inverted continental shelf estuary like the Mississippi River plume, coastal embayment, and drowned river valley). Physiographic setting largely determines the primary production base.
Primary Production Base	The term primary production base refers to various primary producer communities that have unique temperature, substrate, light, and nutrient requirements and thus respond differently to nutrient loading. Susceptibility to eutrophication will vary across estuaries with different primary production bases. Examples of major types of primary producer communities include: emergent marshes and swamps, attached intertidal algae, benthic microalgae, drifting macroalgae, seagrasses, phytoplankton, and coral.
Dilution	Dilution of watershed-derived nutrients occurs due to a variety of mixing processes upon entry into an estuary. The extent to which nutrient loads entering the estuary are diluted will determine to a great extent the susceptibility of the system to eutrophication.
Water Residence Time, and Flushing	The hydraulic residence time of an estuary is the time required to replace the equivalent amount of fresh water in the estuary by fresh-water inputs. In short, it is the time that a molecule of water or nutrient spends in the estuary. Residence time is an important controlling factor on the susceptibility of an estuary to eutrophication (Malone 1977, Cloern et al. 1983, Vallino and Hopkinson 1998, Howarth et al. 2000). Estuaries with short residence times are more able to flush out nitrogen from groundwater, watershed input, etc. Phytoplankton blooms can occur only when the plankton turnover time is shorter than the water residence time. Alternatively, if the residence time is much greater than phytoplankton turnover time, phytoplankton can double several times over prior to being exported, thus producing a bloom.
Stratification	Stratification is an important physical process affecting eutrophication. Stratification can isolate deeper waters from reaeration and maintain phytoplankton in the nutrient rich, photic zone (Malone 1977, Howarth et al. 2000). Most hydrodynamic classifications include a measure of stratification intensity (Hansen and Rattray 1966).
Hypsography	Hypsography is the relative areal extent of land surface elevation. Knowledge of the relationship between estuarine area and elevation/depth will indicate the percentage of area potentially colonizable by emergent marsh, intertidal flats, submerged aquatic vegetation, phytoplankton, macroalgae, etc. Overlaid with measures of water turbidity and stratification, it might be possible to illustrate the spatial extent of sites potentially susceptible to a variety of eutrophication symptoms.
Grazing of Secondary Consumers on Primary Producers	Grazing by benthic filter feeders acts to clear particles from the water column, and can limit the accumulation of algal biomass (Cloern 1982). Alpine and Cloern (1992) showed that filter feeding benthos in San Francisco Bay decreased the response to nutrient loading via phytoplankton production. The California Horn Snail (*Cerethidia sp).* can greatly reduce the concentration of benthic algal biomass, thus reducing the availability of organic matter.
Suspended Materials Load and Light Extinction	Suspended load and light are two important factors that control estuarine response to nutrient loading. Light is a primary factor controlling primary production. (e.g., Cloern 1987, 1991, 1996, 1999). In northern San Francisco Bay, high turbidity from watershed sediment erosion reduces light levels to such an extent that primary production is light-limited year round.
Denitrification	Denitrification converts nitrate to gaseous nitrogen and N_2O, and as such represents a process by which nitrogen is permanently lost from an estuary. Knowledge of the magnitude of denitrification can help predict the eutrophication response of an estuary because nitrogen that is denitrified is largely unavailable to support primary production. Similarly, nitrogen that has been stored as organic N in algal biomass is no longer available to be denitrified; thus bloom events in estuaries can result in the retention of nitrogen in an estuary.
External Inputs of Organic Matter	External loads of organic matter contributes directly to eutrophication. Since organic matter contains nutrients, this input is considered part of the nutrient load. However, the relative magnitude of dissolved versus particulate organic matter loads influences residence time of inputs and how quickly they are processed by the system.

Generally, there are two types of tools needed to link biological response indicators to watershed nutrient loads and other controlling factors: 1) watershed-loading models, and 2) models that relate load to estuarine biological response (a.k.a. estuarine load-response models). The purpose of this section is to highlight the data gaps and critical for the development of these tools and to propose an implementation plan to address these gaps.

3.3.2 Watershed-Loading Models

Watershed-loading models are a critical component of a TMDL toolkit because they are the mechanism used to predict loads to the estuary during any given hydrologic year or portion of that year (e.g., during wet-weather events). In turn, these models serve as an input to the estuarine load-response models. Watershed-loading models are also used to assess pollutant sources and evaluate the impact of proposed source reduction strategies and best management practices (BMPs).

Most of the hydraulic and contaminant loading to estuaries occurs during wet-weather events. Because of this, a stochastic model is generally used to empirically estimate loads during dry weather, while modeling of wet-weather loads can either be done with simple spreadsheet models or dynamic-simulation models. Simple spreadsheet models can be used when only predictions of annual load estimates are required. Such models do not provide any temporal resolution on when the loading occurs (early storm, late storm), andare generally less precise than dynamic-simulation models, although the difference in precision has never been well-quantified. Dynamic-simulation models, which are more data intensive and more expensive to develop, provide more information about loading at various temporal scales because runoff events are simulated. As such, these dynamic-simulation models are better equipped to answer questions on potential utility of BMPs to reduce wet-weather nutrient loads.

While use of dynamic-simulation models to predict watershed loads is clearly preferred for TMDL development, development of dynamic models to predict loading for a large number of watersheds is cost-prohibitive. The use of simple spreadsheet tools to predict annual watershed nutrient loads offers a cost-effective alternative to dynamic-simulation models for this work. In order to employ this tool, additional research is needed to understand the accuracy and precision of simple spreadsheet tools in predicting annual loads.

Either type of model is developed and calibrated using data on the flow-weighted mean concentrations of contaminants (e.g., nutrients) from various types of land uses during storm events. The models are then validated by comparing predicted loads versus measured loads at the bottom of the watershed. Thus, development of good watershed-loading models are critically dependent on the availability of representative data on wet-weather loads from various land uses.

Currently, limited data exist on wet-weather loads from urban and agricultural land uses in California coastal watersheds. The one data set that was collected for this purpose was limited with respect to nutrients and, since it focused on the Los Angeles basin, was not regionally representative of coastal watersheds in the Southern California Bight. No appropriate data sets of wet-weather loading are known to exist in the Central Coast and North Coast. This is a critical data gap for regional implementation of nutrient TMDLs in California. Without these data, watershed-loading models can be built, but there is no way to justify modification of land use loading coefficients if the watershed mass loads are not found to adequately predict measured loads at the bottom of the watershed.

Optimally, an efficient and cost-effective means of developing watershed loading tools for nutrient TMDLS would be to develop a set of models that are calibrated regionally for different types of watersheds. This approach has been taken in southern California for metals TMDLs in the Los Angeles River, San Gabriel River and Ballona Creek watersheds. This approach would be possible with a reliable regional dataset on wet-weather loads from a range of land uses.

3.3.3 Load –Response Models

Development of load-response models is critical for the implementation of the NNE framework in California estuaries because the numeric endpoints are based on biological response (i.e., algal biomass, surface water DO) rather than nutrient concentrations or loads. Load-response models can either take the form of statistical models of nutrient loads and biological response or dynamic-simulation models of estuarine water quality. Both types of load-response models have an important role to play in facilitating the implementation of the California NNE framework for estuaries.

Statistical load-response models establish the relationship between nutrient load (as the independent variable) and biological response (as the dependent variable) over a gradient of disturbance (Figure 3-2). These data might be derived from long-term monitoring in a single estuary, or from synoptically collected monitoring data conducted in multiple estuaries of the same class. The fit of this relationship (i.e., correlation coefficient) is a measure of the degree to which the regression model explains the variability in estuarine biological responses to nutrient loads.

Regression models have an advantage as a tool for TMDL development in that, once established for a class of estuaries, they may be used in an estuary of the same class without extensive data collection. Thus, they are a cost-efficient means to understand the principle factors controlling eutrophication in a class of estuaries and can be used to suggest management measures, including targeted reductions in nutrient loads, without the need for the expensive data collection and modeling required for dynamic-simulation models. These models are also helpful to validate the relationship between nutrient loads and biological response predicted by dynamic-simulation models One major disadvantage of using a simple regression model to establish TMDL allocations is the inability to use the tool to test management scenarios that might arise in altered hydrology, and salinity. Another possible disadvantage is that, when the regression relationship is established based on synoptically collected monitoring data in multiple estuaries during a single year, the overall trend may be highly significant but the precision of the prediction is likely to be poor. This is because of differences between systems, even within estuaries of the same class, and estuarine response to interannual differences in loadings may not be linear.

Several studies have documented the statistical relationships between nitrogen loading and biological response, usually of phytoplankton biomass (chlorophyll a; (Boynton et al. 1982, Boynton et al. 1996, Boynton and Kemp 2000, Harding et al. 2002). Some studies have found that the correlation of these regression models can be improved if key estuarine characteristics and processes are accounted for in the model (e.g., denitrification rates, freshwater residence time, estuarine volume; Dettman 2001). Relationships between phosphorus loads and biological response have also been documented, albeit less commonly (Harding et al. 2002). With the exception of one study, which included San Francisco Bay, all of these studies were conducted in estuaries found in the eastern or gulf coast of the United States. Preliminary work done in California to establish the relationship between macroalgal biomass and total nitrogen (TN) concentrations in major tributaries to estuaries in southern California revealed no statistically significant relationship; however, this study did not estimate TN loads, nor did it take into account major variables which have been documented to control estuarine response (freshwater residence time, estuarine volume; NAS (2000), Dettman 2001). Thus, in order to address this critical data gap, collecting information on nutrient loads and biological response in several estuaries over a gradient of disturbance is recommended. These data can be used to construct simple load response models and as supplemental information to support the development of numeric endpoints.

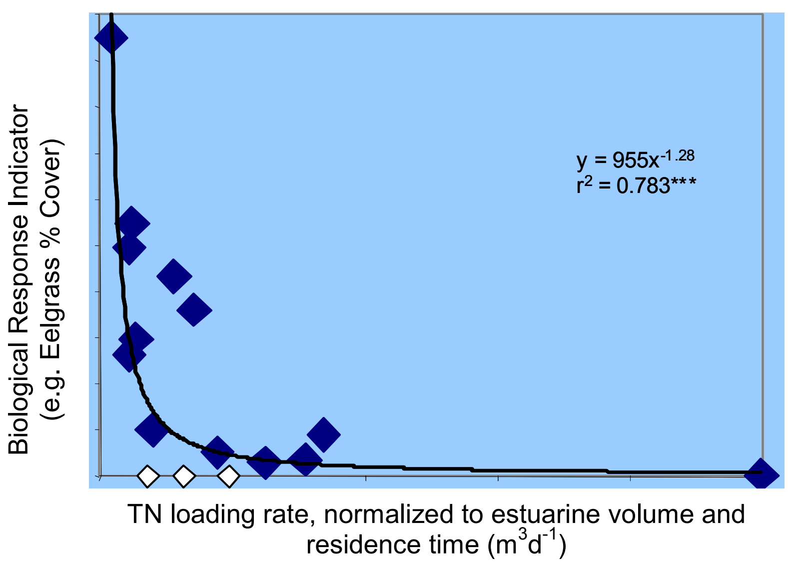

Figure 3-2. Example of a load-response regression model for eelgrass percent cover versus total nitrogen (TN) loading, normalized to estuarine volume and residence time.

Dynamic-simulation models of estuarine water quality have been an important tool in development of water quality standards and allocation of TMDL. These models, once established and calibrated, provide many advantages. These models can be used to assess current condition, assess pollutant sources, evaluate the impact of load reduction strategies and BMPs, and identify important data gaps. Although they are simulations of natural systems, properly calibrated models provide excellent tools for analyses of future projections and alternative planning scenarios to help managers identify and make cost-effective decisions. Both stakeholders and regulators have used estuarine water quality in southern California watersheds to support watershed management decisions (Ackerman and Schiff 2003, TetraTech 2003, 2004, LARWQCB 2005b, c, a, SDRWQCB 2005).

One of the disadvantages of dynamic-simulation models is that they require extensive data collection to calibrate and validate for each estuary. This information typically includes data on the physical forces that move water and sediment through the estuary, data on timing, frequency and magnitude of nutrient inputs to the estuary from various sources, as well as data on the rates of transformation for key biogeochemical processes in estuaries. While these processes controlling nutrient cycling are common to most estuaries, site-specific factors such as climate, hydrology, land use, and the dominant biological communities greatly affect the relative importance and rates of these processes. The current lack of site-specific data on nutrient sources, sinks, and rates of transformation in California estuaries greatly affects the applicability of existing conceptual and dynamic-simulation models to these systems. A comprehensive literature review and identification of data gaps is needed to better prioritize funding of research.

Another disadvantage of dynamic-simulation models is the uncertainty associated with the predicted model output. Model parameters are often adjusted so that predicted outcomes match calibration data; but the models are rarely validated with independent data and may have not accurately modeled the underlying mechanisms that control biological response. For this reason, managers must be careful about how much weight they place on the predicted outcomes of the model. Statistical load-response models can provide an independent means of validating these dynamic-simulation models and improving confidence in the predictions produced.

Because of the great utility of watershed loading and estuarine load-response models to TMDL development, it is worthwhile to consider how to streamline the process for model development and calibration in order to reduce costs and improve model quality. This study demonstrates the importance of developing nutrient TMDL tools for ecoregions in California and establish an internet-based clearinghouse for applicable conceptual models, watershed loading and estuarine water quality models, and supporting studies by estuarine class. Funding for the development of watershed loading and estuarine water quality models should be contingent on the creation of these models in open source code, such that the modeling approaches can be improved over time by the process of collaboration and data sharing. Table 3-8 summarizes the status of existing science and recommendations for development of TMDLs tools for California estuaries.

Table 3-8. Summary of status of science and recommended actions for TMDL tool development.

Tool/Data	Status of Science	Recommended Action
Watershed-loading Models (applies to both dynamic simulation and simple spreadsheet model)	Development of watershed-loading models hampered by lack of wet-weather loading data from a variety of land uses.	Two actions are recommended: 1) acquire data characterizing wet-weather loads of nitrogen and phosphorus from a variety of urban, agricultural and undeveloped land uses. This dataset will be used to create watershed-loading models and statistical load-response models for TMDL development; and 2) develop regionally calibrated dynamic-simulation models of nutrient wet-weather loading.
Simple Spreadsheet Models (to predict annual watershed nutrient loads)	Simple spreadsheet tool to predict annual nutrient loads can provide cost-effective means to improve understanding of harmful algal blooms and estuarine eutrophication. Accuracy and precision in predicting annual loads needs to be better understood.	Compare precision of simple spreadsheet model and dynamic-simulation model for predicting annual loads to estuaries.
Statistical load–response Models	Statistical load-response models exist for some east coast estuaries; these models are not relevant for California because of major differences in dominant primary producer communities and hydrology. Need to investigate better investigate major variables controlling biological response to loads in California estuaries. Monitoring data that relates total nutrient loads to estuarine biological response are only known to be currently available for 2 estuaries (SF Bay and Upper Newport Bay). Data are currently being generated for 5 more lagoons. A more comprehensive data set is required to develop load-response models	Acquire a datasets that relates nutrient loads to estuarine biological response. Two approaches are suggested: 1) develop a statistical load-response model for an existing long-term data set and 2) collect data on an estuarine biological response through a probability-based sample of estuaries. Both approaches will be used to understand the spatial and temporal variability in estuarine load-response
Dynamic-simulation Models (estuarine water quality)	Data are generally lacking on the major processes responsible for uptake, transformation, and release of nutrients in California estuaries.	Conduct a comprehensive literature review and survey of existing research in California to identify key processes that need to be modeled, locate existing and relevant data, and identify major data gaps. This report can be used to identify priorities for future funding by entities such as Sea Grant, USEPA, etc.

4 REVIEW OF INDICATORS AND APPROACHES TO SETTING NUTRIENT NUMERIC ENDPOINTS: PERSPECTIVES FROM EXISTING STATE PROGRAMS

Development of NNEs to address eutrophication in California estuaries can be accelerated by learning from the experiences of other states that have developed or are developing NNE criteria. The purpose of this section is to provide a review of existing state programs with respect to indicators chosen and the technical approach taken to derive criteria, and how this can inform California's approach.

4.1 INDICATORS SELECTED BY EXISTING STATE PROGRAMS

A number of states and programs within the US are in the process of developing nutrient numeric criteria for estuaries. Most of these are still in the developmental phase. Table 4-1 summarizes the indicators that are proposed by each program to assess eutrophication in estuarine environments.

Table 4-1. Summary of proposed or established indicators by program.

Program	DO	Chlorophyll a	Water Clarity	Nutrients	SAV	Benthic Communities
Virginia Province (Cape Cod – Cape Hatteras)	X					
Chesapeake Bay	X	X	X			
Georgia Basin-Puget Sound	X			X	X	
Long Island Sound	X	X		X		
Gulf of Maine	X					
State of Maryland	X		X			
State of Delaware		X	X	X		
State of Texas	X	X	X	X		
State of North Carolina		X		X		
State of New Jersey	X		X		X	X
State of Massachusetts	X	X	X		X	X
State of Florida		X				
State of Virginia	X	X	X			

Some of these programs have advanced to setting thresholds for specific biological response indicators; of which, DO criteria are most common. Table 4-2 lists established thresholds for DO developed for the Chesapeake Bay Program, the Virginia Province, and the Gulf of Maine; and Table 4-3 lists established thresholds for water clarity developed for the Chesapeake Bay Program. Many programs have established narrative criteria for biological response variables and are in the process of collecting monitoring data that would support the development of numeric values that are protective for specific estuaries (e.g., Maryland, Maine, and Chesapeake Bay for chlorophyll a).

Table 4-2. Established thresholds for dissolved oxygen (DO).

Program	End Point	Persistent Exposure (≥ 24-hour low DO conditions)	Episodic and Cyclic Exposure (≤ 24-hour low DO conditions)
Chesapeake Bay	Migratory Fish Spawning and Nursery Use	7-day mean >6 mg/L apply February 1 - May 31	Instantaneous minimum >5 mg/L apply February 1 - May 31
		Open-water fish and shellfish designated use criteria apply June 1 - January 31	
	Shallow-water Bay Grass Use	Open-water fish and shellfish designated use criteria apply year-round	
	Open-water Fish and Shellfish Use	30-day mean >5.5 mg/L (0-0.5 ppt salinity) apply year round	Instantaneous minimum >3.2 mg/L apply year round
		30-day mean >5 mg/L (>0.5 ppt salinity) apply year round	
		7-day mean >4 mg/L apply year round	
	Deep-water Seasonal Fish and Shellfish Use	30-day mean >3 mg/L apply June 1- September 30	Instantaneous minimum >1.7 mg/L apply June 1 - September 30
		1-day mean >2.3 mg/L—apply year round	
		Open-water fish and shellfish designated use criteria apply October 1 - May 31	
	Deep-channel Seasonal Refuge Use	--	Instantaneous minimum >1 mg/L apply June 1 - September 30
		Open-water fish and shellfish designated use criteria apply October 1 - May 31	
Virginia Province Salt water Dissolved Oxygen Criteria	Juvenile and Adult Survival (minimum allowable conditions for survival)	Limit for continuous exposure: DO = 2.3 mg/L	Limit based on the hourly duration of exposure: DO = 0.370*ln(t) + 1.095 where: DO = minimum concentration t = exposure time (hours)
	Growth Effects (maximum conditions required for growth)	Limit for continuous exposure: DO = 4.8 mg/L	Limit based on the intensity and hourly duration of exposure: $\sum_{i}^{n} \frac{t_i * Gred_i}{24} < 25\%$ and $Gred_i = -23.1 * DO_i + 138.1$ where: $Gred_i$ = growth reduction DO_i = allowable concentration (mg/L) t_i = exposure interval duration (hours) I = exposure interval
	Larval Recruitment Effects (specific allowable conditions for recruitment)	Limit based on the number of days a continuous exposure can occur: $\sum \frac{t_i(acutal)}{t_i(allowed)} < 1.0$ and $DO_i = \frac{13.0}{(2.80 + 1.84e^{-0.10ti})}$ where: DO_i = allowable concentration (mg/L) T_i = exposure interval duration (hours) I = exposure interval	Limit based on the number of days an intensity and hourly duration pattern of exposure can occur: Maximum daily cohort mortality for any hourly duration interval of low DO must not exceed a corresponding number of allowable days. Allowable number of days is a function of maximum daily cohort mortality, which is a function of the DO minimum of an interval and the duration of the interval
Gulf of Maine	Protect survival, growth and reproduction of fish populations	Monthly average of 6.5 ppm when the temperature is 24 deg C or less	Instantaneous minimum of 5.0 ppm

Table 4-3. Established thresholds for water clarity.

Program	End-Point	Salinity Regime	Water Clarity Criteria as % Light-through-Water	Water Clarity Criteria as Secchi Depth								Temporal Application
				Water Clarity Criteria Application Depths (m)								
				0.25	0.5	0.75	1	1.25	1.5	1.75	2	
				Secchi Depth (m) for above criteria application depth								
Chesapeake Bay	Preservation of Seagrass Habitat	Tidal Fresh	13%	0.2	0.4	0.5	0.7	0.9	1.1	1.2	1.4	April 1- October 31
		Oligohaline	13%	0.2	0.4	0.5	0.7	0.9	1.1	1.2	1.4	April 1- October 31
		Mesohaline	22%	0.2	0.5	0.7	1	1.2	1.4	1.7	1.9	April 1- October 31
		Polyhaline	22%	0.2	0.5	0.7	1	1.2	1.4	1.7	1.9	March 1- May 31, September 1 - November 30

4.2 TECHNICAL APPROACHES TO SETTING NUMERIC CRITERIA

Technical approaches to developing numeric criteria would be expected to vary widely among states depending on the availability of existing science and monitoring data, the type and size of estuaries and the availability of reference systems, and the predominant sources of nutrient loading to the system. In reviewing the strategies for numeric criteria development in other states, three factors were considered: 1) whether effects-based or percentile based criteria were developed, 2) whether numeric criteria are based on a single indicator versus multiple indicators, and 3) whether the criteria are either site-specific versus regional in scope.

Comparison of effects-based versus percentile approach (USEPA 2001) to setting numeric criteria deserves additional explanation. In brief, the percentile approach consists of a statistical methodology in which endpoints are selected by setting the 75th percentile of reference systems or the 25th percentile of data derived from an ambient survey of indicators in population of estuaries. The effects-based approach relies on scientific data, which links impacts to estuarine beneficial uses (e.g., mortality, reduced reproduction) to certain thresholds in the biological response variable (e.g., hypoxia).

Of the programs reviewed, the Chesapeake Bay Program (USEPA 2003) had the most developed and integrated approach to nutrient criteria (Table 4-8). The three Chesapeake Bay criteria – DO, water clarity, and chlorophyll a – are intended to be viewed as a multimetric set of criteria, largely based on data for the effects of eutrophication on the biological communities in the Bay. The established thresholds were designed to protect species and communities in designated uses during specific time periods. In defining attainment conditions for the nutrient criteria, stressor magnitude, duration, return frequency, spatial extent, and temporal assessment period were accounted for via a monitoring program. The bay was monitored at the spatial scale of 78 bay segments using the most recent three consecutive years of monitoring data. The method used for assessing criteria attainment estimates the frequency with which any given biological response indicator or a combination of indicators (DO, water clarity, and chlorophyll a) is exceeded, as a function of the area or volume affected at a given place and over a defined period of time. These results are then compared to biologically based reference curves, which integrate the acceptable levels of the three criteria, to determine if a given segment is impaired.

The Virginian Province criteria, applied from Cape Cod to Cape Hatteras (USEPA 2000), were generated from a single indicator effects-based approach in which a series of scientific studies were used to derive the lower limits of DO necessary to protect coastal and estuarine animals. This approach combines features of traditional water quality criteria with a new biological framework that integrates time (replacing the concept of an averaging period) and establishes separate criteria for different life stages (larvae versus juveniles and adults). The recommended criteria is then applied to both continuous (persistent) and cyclic (diel, tidal, or episodic) hypoxia. Using this approach, a site is considered to meet objectives for protection if DO exceeds the chronic protective value for growth (4.8 mg/L). If the DO is below the limit for juvenile and adult survival (2.3 mg/L) the site does not meet objectives. When DO is between these values, the site requires evaluation of duration and intensity of hypoxia to determine the suitability of habitat for the larval recruitment objective. Figure 4-1 shows the DO concentration and exposure times required to sustain specific life stages using the Virginian Province criteria. The Chesapeake Bay program used techniques developed for this program when setting its DO criteria.

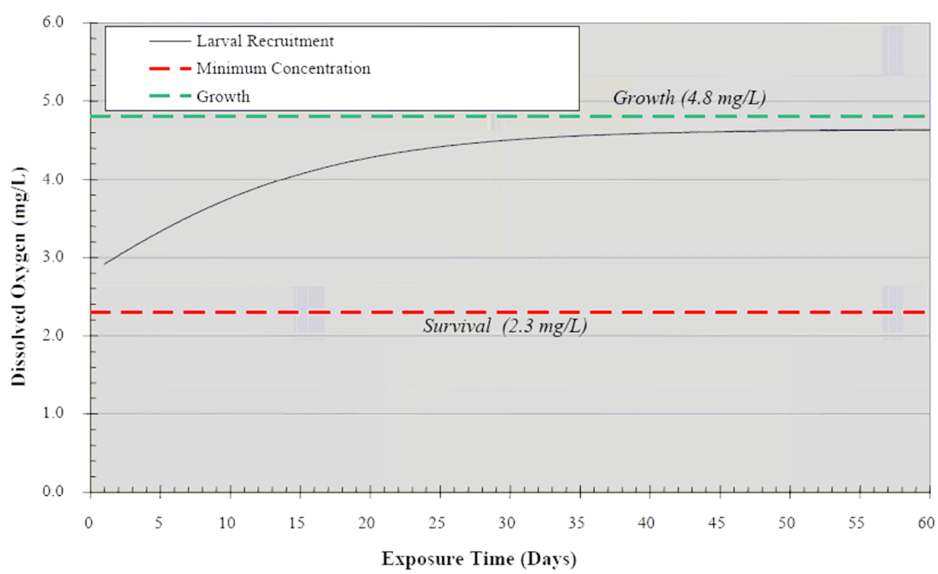

Figure 4-1. Virginian Province dissolved oxygen continuous exposure criteria

The Massachusetts Estuarine Project is approaching the development of criteria on a site-specific basis, using an effects-based, multimetric approach. Instead of generating criteria that would be applied across all estuaries, which would seek to protect the most sensitive systems at the cost of "over-managing" other systems, this program would set site-specific thresholds for each estuary (Howes et al. 2003). The specific parameters would include DO, organic and inorganic nitrogen, water clarity, phytoplankton (as chlorophyll a pigments), and temperature. Other indicators of ecological health under consideration include eelgrass distribution, macroalgal distribution, and benthic infaunal populations. These thresholds will be developed on the basis of specific basin configuration, source water quality, and watershed spatial features for each embayment.

Development of numeric criteria for the Gulf of Maine is being approached in a tiered fashion because the available monitoring data that could be used to set thresholds for biological response indicators are limited (Mills 2003). Their general conceptual approach will yield multimetric, effects-based multimetric criteria. First, they plan to identify a suite of indicators to represent a range of ecosystem functions. These indicators would then be categorized into a tiered system according to the priority of indicators or data availability. Using tiers, the program could start collecting and reporting on indicators with current datasets while working to get monitoring programs in place for future indictor data collection. Similar to California, the Gulf of Maine program plans to classify estuaries into types based on physical and ecological characteristics that would describe susceptibility to eutrophication. The program will establish baseline information and reference conditions to compare changes in indicators, use a literature review of historical data, model conditions for "best possible conditions" given the natural environmental parameters in the area, and/or use local reference sites with the most pristine conditions.

4.3 RELEVANCE OF OTHER STATES' STRATEGIES TO CALIFORNIA'S TECHNICAL APPROACH TO SETTING NNEs

The conceptual framework proposed for development of NNEs in California estuaries has a effects-based multimetric approach. Under the proposed implementation plan, the endpoints, derived as BURC I/II and II/III thresholds, would be regionally derived but could be adjusted to accommodate site-specific differences in the individual Regional Water Quality Control Board basin plans. In general, the approaches taken by other states tend to support the validity of California's conceptual approach.

Effect-based endpoints provide direct linkages to estuarine management endpoints. Thus, whenever effects-based criteria are met, beneficial uses should be sustained. However, in order to apply effect-based thresholds, credible scientific evidence must exist to support the selection of thresholds. For DO, sufficient physiological effects data are available, particularly with respect to the effects of hypoxia on pelagic and benthic species. Even though some of this data is not directly applicable, it may be possible to extrapolate preliminary DO thresholds for California species using organisms within the same genus or family for which data exists. For other biological response variables (macroalgal biomass, water clarity, aesthetics, harmful algal blooms, chlorophyll a, and SAV biomass) much less effects-based literature is available from which thresholds can be derived. Thus, it would require additional time (3-5 years) and funding to conduct the research required to implement this approach.

The time and funding required to fully implement an effects-based approach to endpoint development must be contrasted with the advantages and disadvantages of the percentile approach. Particularly in the case where pristine reference systems are available, use of the percentile approach is arguably cheaper. However, endpoints derived from the percentile approach are not directly linked to beneficial uses; thus endpoints derived from such a method have less value to determine whether beneficial uses are being sustained. For this reason, the percentile approach is not the preferred method for criteria development. Appendix A presents a comparison of effects-based versus percentile approaches to setting DO criteria, using existing data from California perennially tidal estuaries. This analysis illustrates the superiority of effects-based criteria to percentile criteria 1) because strong scientific foundation and linkage to impaired beneficial uses exists, and 2) data do not currently exist in California to implement the percentile approach.

Many programs attempted the 25th percentile approach outlined by the USEPA to guide the development of numeric criteria (USEPA 2001). These programs found that the methodology yields results that differ substantially depending on the study area, and provides no evidence of a technical tie between the thresholds established by the percentile approach and the established beneficial uses for the waters (e.g., Delaware and Texas). Consequently, these programs planned to develop monitoring programs to establish effects-based criteria that support specific beneficial uses (PBS&J 2002, Santoro 2004). In cases where programs will use the percentile approach (e.g., Delaware River Estuary), effects-based endpoints represent a baseline from which managers could determine whether the system is improving or deteriorating rather than serving as water quality criteria per se.

Finally, most programs have rejected setting single criteria that would be applied to all aquatic systems, instead favoring a multimetric approach. In addition, most programs favor development of site-specific criteria that would account for each system's unique physical configuration. The practicality of this approach is facilitated by the general wealth of monitoring data and scientific studies of East Coast estuarine ecosystem relative to the West Coast. Given the number of coastal drainages in California and the relative lack of scientific and monitoring data to support endpoint development, a purely site-specific approach is not cost-effective. California's conceptual framework, in which BURC I/II and II/III endpoints would be regionally derived but adjustable to accommodate site-specific differences, is a reasonable approach given these challenges.

5 IMPLEMENTATION PLAN FOR DEVELOPMENT OF NUTRIENT NUMERIC ENDPOINTS AND TMDL TOOLS

5.1 INTRODUCTION

The overarching goal of the NNE effort is to produce water quality criteria and tools for the effective and cost-efficient reduction of eutrophication in California estuaries. Two objectives will aid the State in achieving this goal: 1) development of numeric endpoints that serve as guidance for RWQCBs to establish water quality criteria, and 2) development of tools for TMDL implementation that link biological response variables to nutrient loading and other factors controlling eutrophication. The USEPA (2007) provided a conceptual framework and a list of research recommendations and data gaps that should be addressed to develop numeric endpoints and TMDL tools for California estuaries. These recommendations were not prioritized, nor was any attempt made to craft them into an implementation plan that could be used as an outline to guide development of numeric endpoints.

The purpose of this section is to present an implementation plan outlining the research and monitoring required for development of numeric endpoints and TMDL tools in California estuarine classes.

As outlined in Tables 3-6 and 3-8, available science supports endpoint development for each of the candidate biological-response variables. In summary:

- Sufficient science appears to exist on the physiological impacts of DO to proceed with NNE development for this biological response variable.
- Additional research and monitoring is required to provide suggested numeric endpoints for the additional biological response variables.
- Critical data gaps to support the development of watershed-loading models include wet-weather loading data from a variety of urban, agricultural and undeveloped land uses.
- The development of statistical estuarine load-response models by estuarine classes, which would greatly aid TMDL implementation, requires a regional dataset of watershed nutrient loads and estuarine biological response variables.
- Cost-effective development of estuarine water quality model requires data on the major processes responsible for transformation, uptake and release of nutrients in estuaries. A literature review that summarizes data relevant to California estuaries is needed to prioritize funding for research.

The plan for numeric endpoint development will be guided by an explicit set of assumptions that must be vetted through the Regional Water Quality Control Boards and the stakeholder group assembled for endpoint development. These assumptions are presented below.

Assumptions Governing Process and Geographic Focus

- Development of numeric endpoints should proceed through a process guided by a group of stakeholders that represent the regulators, the regulated community and scientific experts in the field of eutrophication impacts on estuaries.

- NNE development can proceed most efficiently via a coordinated statewide process.
- Numeric endpoints could be regionalized if warranted by regional differences in estuarine ecology.
- TMDL tool development can proceed more efficiently through a regionalized effort.
- Work to develop numeric endpoints and TDML tools can proceed in phases, with guidance developed where existing science may support it (e.g., for DO) while research and monitoring are being conducted on additional biological response variables and for TMDL tool development.

To efficiently move through the process of numeric endpoint development and TMDL tool development, we recommend four phases, which are outlined in detail below.

5.2 Work Plan Phases, Objectives , and Elements

The general approach proposed for NNE and TMDL tool development in California is to establish a statewide process to develop numeric endpoints, advised by a representative group of stakeholders and an independent panel of scientific experts. Numeric endpoints could be regionalized if warranted by differences in estuarine ecology. TMDL tool development would proceed with an overarching scientific framework, but would be regionalized in order to provide a mechanism for coordinated tool development and cost sharing at the regional level.

Figure 5-1 gives the management structure for this approach, modeled loosely after the approach to develop Sediment Quality Objectives for the State of California. The major organizational units which would serve to advise the SWRCB on NNE development are the technical team (TT), a stakeholder advisory group (SAG), and a Scientific Advisory Board (SAB).

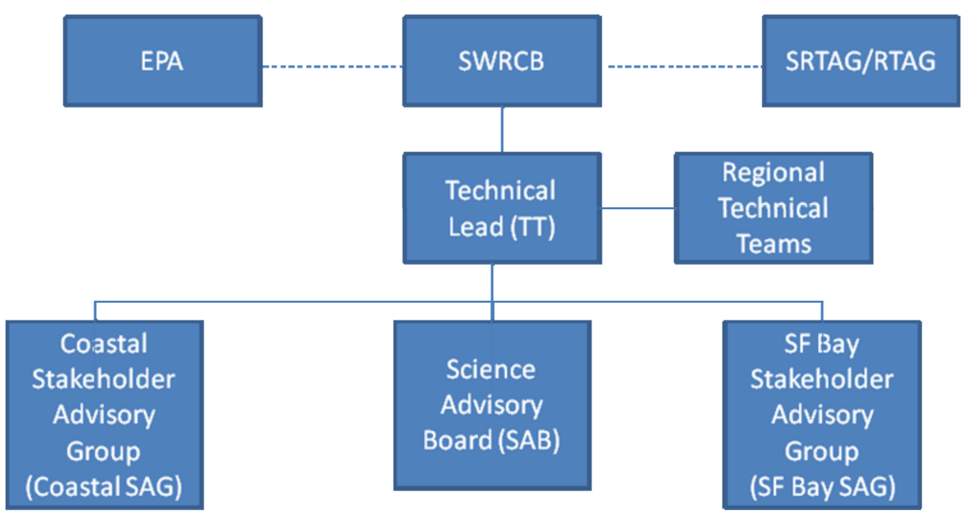

Figure 5-1. Management structure required for statewide approach to NNE development.

The TT would consist of technical lead for the program and regional technical experts in key areas of estuarine ecology relevant to NNE development in California estuaries. The TT's role would be to conduct the scientific studies and analysis of existing data that will support the NNE development process. The TT would be the mechanism to incorporate regional differences in climate, geomorphology, and biological communities into consideration when determining NNE.

The SAG would consist of the regulated community and the interested public (municipalities, stormwater agencies, water districts, environmental groups, etc.) and the RWQCBs. The role of the SAG would be to provide feedback and advice on the conceptual approach, process, and endpoints under consideration. To make this process more efficient, it is suggested that a San Francisco Bay SAG (SF Bay) be created independent of the SAG representing estuaries located on the outer coastline of California (Coastal SAG) because of the size and complexity of San Francisco Bay science and management issues.

The SAB would consist of a small group of national experts in the field of eutrophication in estuaries. The SAB's role would be to provide peer review and advice with respect to science supporting NNE development.

Four phases have been identified to implement this approach. The objectives of these phases are as follows:

Phase 1: Initiate statewide outreach and review of the conceptual framework for estuarine NNE development

Phase 2: Initiate stakeholder driven process to advise the development of an DO numeric endpoint.

Phase 3: Conduct studies to support the development of numeric endpoints for additional biological response indicators and develop numeric endpoints for these indicators.

Phase 4: Develop Nutrient TMDL tools

Phase 1 and 2 activities are well defined with clear endpoints and products. These activities are presented as work elements 1 - 5 below. Phase 3 and Phase 4 studies are presented in summary form, with the intention that these studies will be refined and prioritized through discussions with the regional technical advisory committee (TAC). Detailed study plans will be developed for Phase 3 and Phase 4 studies that are prioritized. Note that work elements in Phases 2 through 4 can be conducted in parallel.

Phase 1: Initiate statewide outreach and coordination on estuarine NNE development

*Work Element 1. Initiate statewide outreac*h. The USEPA Region IX Regional Technical Advisory Group (RTAG) and the California State Regional Board Technical Group (STRTAG) to date have been the principle stakeholders involved in reviewing and refining the NNE framework for lakes and streams. The RTAG is comprised of government, academia, private consultants, and environmental groups in California and Arizona. The STRTAG is comprised of members of the State Water Resources Control Board (SWRCB) and the Regional Water Quality Control Boards (RWQCBs). The purpose of this task is to conduct outreach and get feedback from the RTAG/ STRTAG on the conceptual framework and implementation plan for NNE and TMDL tool development in estuaries. During the initial phase, the existing SWRCB definition of estuary will be interpreted with respect to how it will be applied to select the target population of estuaries for NNE and TMDL tool development..

Work products for this task include: 1) revised conceptual approach and implementation plan for NNE and TMDL tool development and 2) a finalized definition of estuary that will be used to guide decisions on target population of California estuaries.

Work Element 2. Assemble Technical Team (TT), Coastal and SF Bay Stakeholder Advisory Groups (SAG) and Scientific Advisory Board (SAB. The roles of each of these component teams are described in the introduction above. The purpose of this work element is to assemble these teams. The work product for this element is a list of team member agencies and/or individuals representing them for each.

Work Element 3. Compile selected existing data on California estuaries. Decisions on selection of targeted biological response indicators and selection of bioregional boundaries should be driven by existing data on these estuaries. The purpose of this task is to correlate existing data from each estuary in the target population to a baseline of information from which more accurate decisions on indicator selection, estuarine classification, and prioritization of indicator endpoint studies (Phase 3) can be made. These existing data consist of, but are not limited to: 1) estuarine beneficial uses designated for each estuary, 2) identification of the primary producer communities and fish species within each estuary by season and dominant, threatened, or endangered rate, 3) seasonality of ocean inlet closure, if this occurs, and 4) compilation of bathymetry and flow data, where it exists.

The products of this task include a technical memo summarizing dominant primary producer communities, fish species, estuarine volume and flow where available for each estuary, and a database containing the compiled information.

Work Element 4. Propose class and bioregion groups for California estuaries. The regionalized process for NNE and TMDL tool development should be consistent with a statewide scientific framework. The purpose of this task is to provide such consistency by proposing *class and bioregion groups for* California estuaries that can be clustered based on estuarine classification, commonality of key biological response variables identified in Element 3, hydrogeomorphic controls on biological response to nutrient loads, and other bioregional differences in estuarine ecology. Products from Work Element 3 will be used by the TT to make recommendations on a purely technical basis. These recommendations will then be presented to the STRTAG for review with the assumption that refinements suggested will reflect geopolitical considerations.

The work products for this task include: 1) a technical memo that proposes biological response indicators by estuarine class, groups the California systems by estuarine class and bioregion, and provides a technical rationale for these groupings, and 2) a revised memo summarizing the input of the STRTAG and the final proposed estuarine class and bioregion groupings.

Work Element 5. Conduct expert review of existing data on impacts of DO on selected fish indicator species. The purpose of this element is to assemble a group of experts on benthic and pelagic species in California to 1) determine applicability of existing literature and data on physiological impacts of hypoxia on selected indicator species for each estuarine class, 2) suggest BURC thresholds for indicator species for DO by estuarine class, and 3) define additional data gaps to be addressed in Phase 3.

Work Products should include a report summarizing findings of the workgroup.

Work Element 6. Convene SAG and SAB meetings to review the conceptual approach to NNE development and prioritize Phase 3 studies. The purpose of this task is to convene the Coastal and SF Bay SAGs and SAB to review the conceptual approach to NNE development and prioritize Phase 3 studies for additional biological indicators. Table 3-6 summarizes the recommended course of action with respect to studies for biological response variables. Additional studies may be identified by the SAGs or SAB. Work Element 3 provides additional data to inform stakeholders which biological response indicators (in addition to DO) are common to most estuaries. Detailed study plans will be developed for selected Phase 3 studies. Work products should include a revised conceptual framework document and detailed study plans for selected priority studies.

Phase 2: Initiate a statewide stakeholder-driven process to advise development of DO numeric endpoints

Work Element 7. Set NNEs for DO. The purpose of this element is to guide the SAGs and the SAB through a process of setting numeric endpoints for surface water DO for each estuarine beneficial use, based on scientific literature and expert review conducted in Work Element 2.

Phase 3: Address data gaps for additional biological response indicators

The work elements that correspond to critical data gaps in Phase 2 would be expected to differ among bioregions. As an example, the following are suggested elements to support the development of numeric endpoints for macroalgae and nuisance SAV relevant to shallow perennially tidal and seasonally tidal lagoons. Specific studies would be identified and prioritized for funding based on discussions with the SAB and the SAGs in Work Element 6. Thus, the final work elements in the Phase 3 will change as Phase 1 work elements are completed and reviewed by the SAGs and the SAB.

Work Element 8. Impact of macroalgae on SAV. The purpose of this element is to determine the relationship between macroalgal biomass or percent cover and loss of SAV habitats. Loss of SAV habitat is directly linked to impairment of estuarine habitat for fish and benthic species. This task would be conducted via mesocosm and field-based experiments to determine the amount and duration of macroalgae blooms that result in loss of seagrass habitat. Consideration of the spatial scale of impact (extent of macroalgal bloom), depth of habitat, as well as seagrass habitat recovery time will be given in defining the thresholds.

Work Element 9. Impact of macroalgae on benthic infauna. The purpose of this element is to determine the relationship between macroalgal biomass and benthic infauna community composition and biomass. Reduction in benthic algal biomass causes impairment in estuarine habitat, specifically impacting food sources for fish and birds, including migratory and threatened and endangered species. This task would be conducted via mesocosm and field-based experiments and field surveys to determine the amount and duration of macroalgal blooms that results in significant changes in benthic algal biomass. Consideration of the confounding factors such as hydrodrologic residence time, tidal elevation, sediment oxygen demand, etc. will be given to provide results that can be interpreted to select critical thresholds.

Work Element 10. Levels of macroalgal cover associated with poor aesthetics. The purpose of this element is to determine the levels of macroalgal cover associated with poor aesthetics or recreational experiences. The assumption is that estuaries with extensive macroalgal blooms are impaired for REC2 beneficial uses. This task would utilize focus groups to determine levels of macroalgae cover that provide a poor aesthetic or recreational experience.

Phase 4: TDML Tool Development

The conceptual basis for effective TMDL development holds that a regionalize approach can provide an economy of scale and improve technical tools available to manage nutrients associated with eutrophication. The following five work elements were judged to be common to all regions in the development of TMDL tools. Work elements would be refined, customized, and prioritized within each region. **The assumption is that the majority of funding would come from regional stakeholders, but that an overarching scientific framework could guide this effort with supplemental funding from the State.**

Work Element 11. Acquire wet-weather nutrient loading data from a variety of land uses. The purpose of this task is to develop a regional data set documenting wet-weather loads of nitrogen and phosphorus from a variety of land uses in coastal watersheds within each region. Mass loads and event mean concentrations would be characterized for approximately 100 wetweather events per region from a variety of land uses such residential, commercial, industrial, low and high intensity agricultural, and open space. Distribution of sites would maximize the applicability of the set to the coastal watersheds within the region.

Work Element 12. Compare the precision of simple spreadsheet models to dynamic-simulation models for predicting watershed loads. Simple spreadsheet tools to predict annual nutrient loads can provide cost-effective means to improve understanding the dynamics of harmful algal blooms and estuarine eutrophication. Accuracy and precision of simple spreadsheet tools in predicting annual loads needs to be better understood. Dynamic-simulation models are currently being created for nutrient TMDLs in five San Diego county lagoons. The purpose of this work element is to compare precision of simple spreadsheet model versus dynamic-simulation models in predicting annual loads from these lagoons. This work will greatly aid efforts to understand how watershed nutrient loading is controlling the occurrence and extent of HABs in coastal waters.

Work Element 13. Acquire data characterizing watershed nutrient loading and estuarine biological response. Statistical estuarine load-response models are needed to better investigate major variables controlling biological response to loads in California estuaries and to provide a cost-effective means to estimate load reductions required to achieve numeric endpoints. Currently, monitoring data that relates total nutrient loads to estuarine biological response are available for only two estuaries: San Francisco Bay and Upper Newport Bay; data are being generated for five more lagoons. A more comprehensive data set is required to develop statistical load-response models. This work element has two objectives: 1) develop a load-response model based on existing historical data from Upper Newport Bay (macroalgal biomass, total nitrogen loads), and 2) acquire data on nutrient loads and estuarine biological response from a probability-based sample of estuaries in each region. For those segments of estuaries that are selected, data will be collected monthly over a one-year period on macroalgal biomass, sediment and surface water chlorophyll a, and SAV biomass. Data will be collected on surface water DO, turbidity, salinity, and other physiochemical variables continuously over a four-month period in the dry season and a three-month period in the wet season. Data on annual nutrient loads will be predicted from simple watershed-loading spreadsheet models. The data will serve to develop statistical load-response models for each regional class of estuaries (Work Element 16). It can be used to make predictions on the load reductions required to achieve the targeted numeric endpoints and to validate dynamic-simulation models. Finally, the data can also be used to generate a 305(b) report for eutrophication in estuaries for each region.

Work Element 14. Develop statistical load-response models. Statistical load-response models exist for some East Coast estuaries, but these models are not relevant for California because of major differences in dominant primary producer communities and hydrology. There is a need to better investigate major variables controlling biological response to loads in California estuaries and develop statistical load-response models. The purpose of this work element is to utilize the data set generated in Work Element 15 to develop a statistical load–response model. Major differences in key estuarine characteristics and processes will be accounted for in the model (e.g., freshwater residence time, denitrification rates, etc.) and differences in the biological response to nutrient loads among estuarine class will be explored.

Work Element 15. Literature review of existing data on rates of key processes impacting nutrient cycling in California estuaries. Dynamic-simulation models of estuarine water quality require data on the rates of transformation for key biogeochemical processes in estuaries. While the processes controlling nutrient cycling are common to most estuaries, site-specific factors such as climate, hydrology, land use, and the dominant biological communities greatly affect the relative importance and rates of these processes. The current lack of site-specific data on nutrient sources, sinks, and rates of transformation in California estuaries greatly affects the applicability of existing conceptual and dynamic-simulation models to these systems. The purpose of this literature review is to identify the major processes that should be modeled, review the literature and existing studies of California estuaries to compile rates for these studies, and identify the major data gaps for which research needs to be conducted. The results of this task can be used to identify priorities for future funding by entities such as Sea Grant, USEPA, and other entities.

5.3 TIMELINE FOR IMPLEMENTATION

Table 5-1 gives a general timeframe for Phase 1 of the implementation plan. Phase 3 and 4 activities can begin after Element 8 (prioritization of Phase 3 and 4 studies) has been completed. These phases are of unspecified length because of lack of detail of the studies that will be selected for each region.

Table 5-1. Suggested schedule for implementation of workplan elements for NNE development.

Workplan Element	*Y1-Q1*	*Y1-Q2*	*Y1-Q3*	*Y1-Q4*	*Y2-Q1*	*Y2-Q2*	*Y2-Q3*	*Y2-Q4*	*Y3-Q1*	*Y3-Q2*	*Y3-Q4*	*Y4*	*Y4*	*Y6*
Phase 1: Stakeholder Outreach and Coordination on estuarine NNE Development														
1. Initiate statewide outreach														
2. Assemble TT, SAG, and SAB														
3. Compile existing data														
4. Propose groups of estuaries														
5. Expert review of DO data														
6. Initiate SAG and SAB review														
Phase 2: Initiate Regional NNE development														
7. Set NNEs for DO														
Phase 3: Develop NNEs for other indicators														
Phase 4: TMDL tool development														

6 CONCLUSIONS

This report builds on the conceptual framework for NNE development in estuaries by: 1) undertaking a review of existing literature to determine available science to support numeric thresholds for a suite of biological response indicators, 2) summarizing the state of science with respect to TMDL tool development, 3) reviewing the technical approaches of other state's programs in developing numeric criteria to address eutrophication, and 4) outlining an implementation plan to develop NNEs and TMDL tools in California estuaries.

The review found that, of the seven candidate indicators, DO is the variable for which the most data are available to support numeric endpoint development, particularly with respect to the physiological effects of hypoxia on pelagic and benthic species. However, most of these data are derived from studies conducted for East Coast estuaries, and little data exist on the response of California fish and benthic invertebrate species to low DO environments. Despite these existing data gaps, it is possible to develop criteria for DO based on existing data in the near term by extrapolating preliminary DO thresholds for California species using organisms within the same genus or family for which data exists. For other biological-response variables (macroalgal biomass, water clarity, aesthetics, harmful algal blooms, chlorophyll a, and SAV biomass) much less literature is available from which effects-based thresholds can be derived. Furthermore, little or no monitoring data was found for California estuaries to support the development of effects-based criteria. These data gaps should be addressed in order to create a multi-metric approach to setting numeric endpoints in California estuaries.

An effects-based approach is the most scientifically credible means to establish NNE for California estuaries has direct linkages to beneficial uses, and when effects-based criteria are met, beneficial uses will be sustained. In contrast, the endpoints established using the percentile approach do not have a solid scientific foundation. This approach is even further hampered by the lack of pristine, reference systems in California. Thus, efforts to establish numeric endpoints for California estuaries should be effects-based, focusing on research which establishes scientific links to beneficial uses and outlines critical thresholds which, when met, will sustain estuarine ecosystems.

A summary of existing science to support TMDL tool development shows that: 1) critical data gaps to support the development of watershed-loading models include wet-weather loading data from a variety of urban, agricultural, and undeveloped land uses; 2) the development of statistical estuarine load-response models by estuarine classes requires a regional dataset of watershed nutrient-loading and biological-response variables in a representative sample of estuaries; and 3) cost-effective development of estuarine water-quality model requires data on the major processes responsible for transformation, uptake, and release of nutrients in estuaries; and 4) a literature review that summarizes data relevant to California estuaries is needed to prioritize funding for research.

This report presents an implementation plan to develop numeric endpoints and TMDL tools in a selected set of estuarine classes. To efficiently move through the process of numeric-endpoint development, a phased approach is recommended. This phased approach would allow for the development of endpoints for DO to move forward while data gaps for the additional biological response variables and TMDL tool development are addressed. Phase 1 and 2 activities are well-defined with clear endpoints and products. Phase 3 and 4 studies are presented in summary form. Detailed study plans will be developed for studies that are prioritized.

7 REFERENCES

Ackerman, D. and K. Schiff. 2003. Modeling stormwater mass emissions to the southern California bight. *Journal of Environmental Engineering* 129:308-317.

Alpine, A.E., and J.E. Cloern. 1992. Trophic interactions and direct physocial effects control phytoplankton biomass and production in an estuary. *Limnology and Oceanography* 37:946-955.

Arrigo, K.R., D.H. Robinson, D.L. Worthen, R.B. Dunbar, G.R. DiTullio, M. VanWoert and M. P. Lizotte. 1999. Phytoplankton community structure and the drawdown of nutrients and CO_2 in the Southern Ocean. *Science* 283:365-367.

Atkinson, M.J. and S.V. Smith. 1983. C:N:P ratios of benthic marine plants. *Limnology and Oceanography* 28:568-574.

Backman, T.W. and D.C. Barilotti. 1976. Irradiance reduction effects on standing crops of the eelgrass *Zostera marina* in a coastal lagoon. *Marine Biology* 34:33-40.

Bates, S., C. J. Bird, A. S. W. de Freitas, R. Foxall, M. Gilgan, L. A. Hanic, G. R. Johnson, A. W. McCulloch, and P. Odense. 1989. Pennate diatom Nitzschia pungens as the primary source of domoic acid, a toxin in shellfish from eastern Prince Edward Island, Canada. Pages 1203-1215.

Bates, S., C.A. Scholin, M. Ferguson and C. Leger. 1999. Application of ribosomal RNA-targeted probes to detect *Pseudo-nitzchia* multiseries and *P. pungens* in Atlantic Canadian waters. p. 2261 *in:* J.L. Martin and K. Haya, editors, Proceedings of the Sixth Canadian Workshop on Harmful Marine Algae. Canadian Technical Report on Fish and Aquatic Science.

Bates, S.S., A.S.W. DeFreitas, J.E. Milley, R. Pocklington, M.A. Quilliam, J.C. Smith and J. Worms. 1991. Controls on domoic acid production by the diatom Nitzschia pungens f. multiseries in culture: nutrients and irradiance. *Canadian Journal of Fisheries and Aquatic Sciences* 48:1136-1144.

Beitinger, T. L., and M. J. Pettit. 1984. Comparision of low oxygen avoidance in a bimodal breather, *Erpetoichthys calabaricua* and an obligated water breather, *Percina caprodes*. *Environmental Biology of Fishes* 11:235-240.

Beloff, B.R. and E.R. Beaver. 2000. Evaluation of societal costs: odors and eutrophication. BRIDGES to Sustainability, Houston, TX.

Bierzychudek, A., C. D'Avanzo, and I. Valiela. 1993. Effects of macroalgae, night and day, on ammonium profiles in Waquoit Bay. *Biological Bulletin* 185:330-331.

Borsuk, M.E., C.A. Stow, J.R.A. Luettich, H.W. Paerl and J.L. Pinckney. 2001. Modelling oxygen dynamics in an intermittently stratified sstuary: Estimation of process rates using field data. *Esturine, Coastal and Shelf Science* 52:33-49.

Boynton, W.R., J.D. Hagy, L. Murray, C. Stokes and W.M. Kemp. 1996. A comparative analysis of eutrophicaiton pattersn in a temperate coastal lagoon. *Estuaries* 19:408-421.

Boynton, W.R. and W.M. Kemp. 2000. Influence of river flow and nutrient loads on selected ecosystem processes: a synthesis of Chesapeake Bay data. Pages 269-298 *in*: J.E. Hobbie, editor, Estuarine Science: A Synthetic Appraoch to Research and Practice. Island Press. Washington, DC.

Boynton, W.R., W.M. Kemp and C.W. Keefe. 1982. A comparative analysis of nutrients and other factors influenceing estuarine phytoplankton production. pp. 69-90. *in:* V.S. Kennedy, editor, Esuarine Comparisons. Academic Press. San Diego, CA.

Brandt, S.B., E. Demers, J.A. Tyler and M.A. Gerken. 1998. Fish Bioenergetics Modeling: Chesapeake Bay Ecosystem Modeling Program (1993-1998), Report to the Chesapeake Bay Program. US Environmental Protection Agency, Chesapeake Bay Program Office. Annapolis, MD.

Breitburg, D. 2002. Effects of hypoxia, and the balance between hypoxia and enrichment, on coastal fishes and fisheries. *Estuaries* 25:767-781.

Breitburg, D.L. 1992. Episodic hypoxia in Chesapeake Bay: Interacting effects of recruitment, behavior and physical disturbance. *Ecological Monographs* 62:525-546.

Breitburg, D.L. 1994. Behavioral response of fish larvae to low dissolved oxygen concentrations in a stratified water column. *Marine Biology* 120:615-625.

Breitburg, D.L., T. Loher, C.A. Pacey and A. Gerstein. 1997. Varying effects of low dissolved oxygen on trophic interactions in an estuarine food web. *Ecological Monographs* 67:489-507.

Breitburg, D.L., L. Pihl and S.E. Kolesar. 2001. Effects of low dissolved oxygen on the behavior, ecology and harvest of fishes: A comparison of the Chesapeake Bay and Baltic-Kattegat systems. pp. 241-267.*in*: N.N. Rabelais and R.E. Turner, editors, Coastal Hypoxia: Consequences for living resoures and ecosystmes. American Geophysical Union. Washington, DC.

Breitburg, D.L., N. Steinberg, S. DuBeau, C.Cooksey and E.D. Houde. 1994. Effects of low dissolved oxygen on predation on estuarine fish larvae. *Marine Ecology Progress Series* 104:235-246.

Bricker, S.B., C.G. Clement, D.E. Pirhalla, S.P. Orlando and D.R.G. Farrow. 1999. National Estuarine Eutrophication Assessment: Effects of Nutrient Enrichment in the Nation's Estuaries. NOAA, National Ocean Service, Speical Projects Office and National Centers for Coastal Ocean Science. Silver Springs, MD.

Bricker, S.B., J.G. Ferreira and T. Simas. 2003. An integrated methodology for assessment of estuarine trophic status. Ecological Modelling 169:39-60.

Brun, F.G., I. Hernandez, J.J. Vergara, G. Peralta and J.L. Perez-Llorens. 2002. Assessing the toxicology of ammonium pulses to the survival and growth of *Zostera noltii*. *Marine Ecology Progress Series* 225:177-187.

Burkholder, J., E. Noga, C. Hobbs and H. Glasgow. 1992. New 'phantom' dinoflagellate is the causative agent of major estuarine fish kills. *Nature* 358:407-410.

Burton, D.T., L.B. Richardson and C.J. Moore. 1980. Effect of oxygen reduction rate and constant low dissolved oxygen concentrations on two estuarine fish. *Transactions of the American Fisheries Society* 109:552-557.

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

Campbell, S.G., R.B. Hanna, M. Flug and J.F. Scott. 2001. Modeling Klamath River system operations for quantity and quality. *Journal of Water Resources Planning and Management* 127:284-294.

Cardoso, P.G., M.A. Pardal, A.I. Lillebo, S.M. Ferreira, D. Raffaelli and J.C. Marques. 2004. Dynamic changes in seagrass assemblages under eutrophication and implications for recovery. *Journal of Experimental Marine Biology and Ecology* 302:233-248.

Chesney, E.J. and E.D. Houde. 1989. Laboratory studies on the effect of hypoxic waters on the survival of eggs and yolk-sac larvae of the bay anchovy, Anchova mitchilli. pp. 184-191 *in*: E. D. Houde, E.J. Chesney, T.A. Newberger, A.V. Vazquez, C.E. Zastrow, L.G. Morin, H.R. Harvey and J.W. Gooch, editors, Population Biology of Bay Anchovy in Mid-Chesapeake Bay- Final Report to Maryland Sea Grant. R/F-56, UMCEES Ref. No. CBL 89-141.

Chetelat, J., F. Pick, A. Morin and P. Hamilton. 1999. Periphyton biomass and community composition in rivers of different nutrient status. *Canadian Journal of Fisheries and Aquatic Sciences* 56:560-569.

Chittenden, Jr., M.E. 1972. Effects of handling and salinity on oxygen requirements of the striped bass Morone saxatilis. *Journal of Fisheries Research Board of Canada* 28:1823-1830.

Christie, R.W., P.T. Walker, A.G. Eversole and T.A. Curtis. 1981. Distribution of spawning blueback herring on the West Branch of Cooper River and Aantee River, South Carolina. Proceeding of the Annual Conference of the Southeastern Association of Fisheries and Wildlife Agencies 35:632-640.

Cloern, J.E. 1982. Does the benthos control phytoplankton biomass in South San Francisco Bay? *Marine Ecology Progress Series* 9:191-202.

Cloern, J.E. 1991. Tidal stirring and phytoplankton bloom dynamics in and estuary. *Journal of Marine Research* 49:203-221.

Cloern, J.E. 1996. Phytoplankton bloom dynamics in coastal ecosystems: A review with some general lessons from sustained investigation of San Francisco Bay, California. *Reviews of Geophysics* 34:127-168.

Cloern, J.E. 1999. The relative importance of light and nutrient limitation of phytoplankton growth: a simple index of coastal ecosystem sensitivity to nutrient enrichment. *Aquatic Ecology* 33:3-16.

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

Cloern, J.E., B.E. Cole, R.L.J. Wong and A.E. Alpine. 1985. Temporal dynamics of estuarine phytoplankton: A case study of San Francisco Bay. *Hydrobiologia* 129:153-176.

Coutant, C.C. 1985. Striped bass, temperature, and dissolved oxygen: A speculative hypothesis for environmental risk. *Transactions of the American Fisheries Society* 114:31-61.

Craig, J.K., C.D. Gray, C.M. McDaniel, T.L. Henwood and J.G. Hanifen. 2001. Ecological effects of hypoxia on fish, seaturtles, and marine mammals in the northwestern Gulf of Mexico. pp. 269-291 *in*: N.N. Rabalais and R.E. Turner, editors, Coastal Hypoxia: Consequenes for Living Resources and Ecosystems. American Geophysical Union, Washington, D.C.

Crocker, C.E. and J.J. Cech. 1997. Effects of environmental hypoxia on oxygen consumption rate and swimming activity in juvenile white sturgeon, *Acipenser transmontanus*, in relation to temperature and life intervals. *Environmental Biology of Fishes* 50:383-389.

Cummins, S.P., D.E. Roberts and K.D. Zimmerman. 2004. Effects of the green macroalga *Enteromorpha intestinalis* on macrobenthic and seagrass assemblages in a shallow coastal estuary. *Marine Ecology-Progress Series* 266:77-87.

D'Avanzo, C. and J.N. Kremer. 1994. Diel oxygen dynamics and anoxic events in an eutrophic estuary of Waquoit Bay, Massachusetts. *Estuaries* 17:131-139.

Dauer, D.M., A. J. Rodi and J.A. Ranasinghe. 1992. Effects of low dissolved oxygen events on the macrobenthos of the Lower Cheasapeake Bay. *Estuaries* 15:384-391.

Dettman, D.L., M.J. Kohn, J. Quade, F.J. Ryerson, T.P. Ojha and S. Hamidullah. 2001. Seasonal stable isotope evidence for a strong Asian monsoon throughout the past 10.7 m.y. *Geology* 29:31-34.

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

Diaz, R.J., R.J. Neubauer, L.C. Schaffner, L. Phil and S.P. Baden. 1992. Continous monitoring of dissolved oxygen in an estuary experience periodic hypoxia and the effects of hypoxia on macrobenthos and fish. *Science of the Total Environment* supplement.

Dobberfuhl, D.R. 2007. Light limiting thresholds for submerged aquatic vegetation in a blackwater river. *Aquatic Botany* 86:346-352.

Dodds, W.K., V.H. Smith and K. Lohman. 2002. Nitrogen and phosphorus relationships to benthic algal biomass in temperate streams. *Canadian Journal of Fisheries and Aquatic Sciences* 59:865-874.

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

Dunton, K.H. 1994. Seasonal growth and biomass of the subtropical seagrass Halodule wrightii in relation to continous measurements of underwater irradiance. *Marine Biology* 120:479-489.

Eby, L.A. and L.B. Crowder. 2002. Hypoxia-based habitat compression in the Neuse River Estuary: context-dependent shifts in behavioral avoidance thresholds. *Canadian Journal of Fisheries and Aquatic Sciences* 59:952-965.

Fehling, J., K. Davidson, C.J. Bolch and S.S. Bates. 2004. Growth and domoic acid production by *Pseudo-nitzschia seriata* (*Bacillariophyceae*) under phosphate and silicate limitation. *Journal of Phycology* 40:674-683.

Fernandez, J.M., M.A.E. Selma, F.R. Aymerich, M.T.P. Saez and M.F.C. Fructuoso. 2005. Aquatic birds as bioindicators of trophic changes and ecosystem deterioration in the Mar Menor lagoon (SE Spain). *Hydrobiologia* 550:221-235.

Fevold, K. 1998. Subsurface controls on the distribution of benthic algae in floodplain back channel habitats of the Queets River. University of Washington, Seattle. Seattle, WA.

Fong, P., K.E. Boyer and J.B. Zedler. 1998. Developing and indicator of nutrient enrichment in coastal estuaries and lagoons using tissue nitrogen content of the opportunistic algal, Enteromorpha intestinalis (L.Link). *Journal of Experimental Marine Biology and Ecology* 231:63-79.

Fong, P. and J.B. Zedler. 2000. Sources, sinks and fluxes of nutrients (N + P) in a small highly modified urban estuary in southern California. *Urban Ecosystems* 4:125-144.

Forbes, T.L. and G.R. Lopez. 1990. The effect of food concentration, body size and environmental oxygen tension on the growth of the deposit deding polychaete, Capitella species. *Limnology and Oceanography* 35:1535-1544.

Fryxell, G., M.C. Villac and L.P. Shapiro. 1997. The occurence of the toxic diatom genus Pseudo-nizschia (Bacillario-phyceae) on the West Coast of the USA: 1920-1996: A review. *Phycologia* 36:419-437.

Geider, R.J. and J. LaRoche. 2002. Redfield revisited: variability of C:N:P in marine microalgae and its biochemical basis. *European Journal of Phycology* 37:1-17.

Geist, D.R., C.S. Abernethy, K.D. Hand, V.I. Cullinan, J.A. Chandler and P.A. Groves. 2006. Survival, development, and growth of fall Chinook salmon embryos, alevins, and fry exposed to variable thermal and dissolved oxygen regimes. *Transactions of the American Fisheries Society* 135:1462-1477.

Gibson, G.R., M.L. Bowman, J. Gerritsen and B.D. Snyder. 2000. Estuarine and Coastal Marine Waters: Bioassessment and Biocriteria Technical Guidance. US Environmental Protection Agency. Washington D.C.

Gillen, M., T. Ketelsen, P. Mack, L. Miles and W. Stavrianou. 2006. Odour as a community problem in the immediate vicinity of the Woodman Point Wastewater Treatment Plant. The University of Western Australia, School of Environmental Systems Engineering Design. Melborn, Australia.

Goodman, J.L., K.A. Moore and W.C. Dennison. 1995. Photosynthetic responses of eelgrass (*Zostera marina*) to light and sediment sulfide in shallow barrie lagoon. *Aquatic Botany* 50:37-48.

Gray, J. S. 1992. Eutrophication in the sea. pp. 3-15 *in:* G. Columbo, I. Ferrari, V. U. Ceccherelli, and R. R., editors. Marine Eutrophication and Population Dynamics. Olsen & Olsen, Fredensborg.

Greening, H., and C. Elfring. 2002. Local, State, Regional, and Federal ROles in Coastal Nutrient Management. *Estuaries.* 25: 838-847.

Hansen, D.V. and M. Rattray. 1966. New dimensions in estuary classification. *Limnology and Oceanography* 11:319-326.

Harding, L.W.J., M.E. Mallonee and E. Perry. 2002. Toward a predictive understanding of primary productivity in a temperate, partially stratified estuary. *Estuarine Coastal and Shelf Science* 55:437-463.

Harlin, M.M. and B. Thorne-Miller. 1981. Nutrient enrichment of seagrass beds in a Rhode Island coastal lagoon. *Marine Biology* 65:221-229.

Hauxwell, J., J. Cebrian, C. Furlong and I. Valiela. 2001. Macroalgal canopies contribute to eelgrass (*Zostera marina*) decline in temperate estuarine ecosystems. *Ecology* 82:1007-1022.

Hawkins, J.N. 1979. Anadromous Fisheries Research Program: Neuse River. North Carolina Deparment of Natural Resources and Community Develoment, Division of Marine Fisheries. Morehead City, NC.

Heiskary, S. and H. Markus. 2001. Establishing relationships among nutrient concentrations, phytoplankton abundance, and biochemical oxygen demand in Minnesota, USA rivers. *Journal of Lake and Resevoir Management* 17:251-262.

Hernandez, I., G. Peralta, J.L. Perez-Llorens and J.J. Vergara. 1997. Biomass and dynamics of growth of *Ulva* species in Palmones River Estuary. *Journal of Phycology* 33:764-772.

Holmquist, J.G. 1997. Disturbance and gap formation in a marine benthic mosaic: influence of shifting macroagal patches on seagrass structure and mobile invertebrates. *Marine Ecology Progress Series* 158:121-130.

Howarth, R.W. 1988. Nutrient limitation of primary production in marine ecosystems. *Annual Review of Ecology and Systematics* 19:89-110.

Howarth, R.W., A. Sharpley and D. Walker. 2002. Sources of nutrient pollution to coastal waters in the United States: Implications for achieving coastal water quality goals. *Estuaries* 25:656-676.

Howarth, R.W., D.P. Swaney, T.J. Butler and R. Marino. 2000. Climatic control on eutrophication of the Hudson River Estuary. *Ecosystems* 3:210-215.

Howell, P. and D. Simpson. 1994. Abundance of marine resources in relation to dissolved oxygen in Long Island Sound. *Estuaries* 17:394-402.

Howes, B.L., R. Samimy and B. Dudley. 2003. Massachusetts Estuaries Project Site Specific Nitrogen Thresholds for Southeastern Massachusetts Embayments: Critical Indicators. Massachusetts Department of Environmental Protection. Boston, MA.

Jenkins, W.E., T.I.J. Smith, L.D. Heyward and D.M. Knott. 1993. Tolerance of shortnose sturgon, Acipenser brevirostrum, juveniles to different salinity and dissolved oxygen concentrations. Proceedings of the Annual Conference of Sotheastern Association of Fish and Wildlife Agencies 47:476-484.

Johnson, M.R., S.L. Williams, C.H. Lieberman and A. Solbak. 2003. Changes in the abundance of the seagrasses *Zostera marina L*. (eelgrass) and *Ruppia maritima L*. (widgeongrass) in San Diego, California, following an El Nino event. *Estuaries* 26:106-115.

Jorgensen, B.B. 1980. Seasonal oxygen depletion in the bottom waters of a Danish Fjod and its effects on the benthic community. *Okios* 34:68-76.

Jorgensen, B. B., and K. Richardson. 1996. Eutrophication in Coastal Marine Systems. American Geophysical Union, Washington, D.C.

Kamer, K., K.A. Boyle and P. Fong. 2001. Macroalgal bloom dynamics in a highly eutrophic southern California estuary. *Estuaries* 24:623-635.

Kamer, K. and E. Stein. 2003. Dissolved oxygen conentration as a potential indicator of water quality in Newport Bay: A review of scientific research, historical data, and criteria development. Southern California Coastal Water Research Project. Westminster, CA.

Kennison, R., K. Kamer and P. Fong. 2003. Nutrient dynamics and macroalgal blooms: A comparison of five southern California estuaries. Southern California Coastal Water Research Project. Westminster, CA.

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

Koch, M.S., I.A. Mendelssohn and K.L. McKee. 1990. Mechanism for the hydrogen sulfide-induced growth limiation in wetland macrophytes. *Limnology and Oceanography* 35:399-408.

Kramer, D.L. 1983. Aquatic surface respiration in the fishes of Panama: distribution in relation to risk hypoxia. *Environmental Biology of Fishes* 8:49-54.

Kramer, D.L. 1987. Dissolved oxygen and fish behavior. Environmental Biology of Fishes 18.

Krause-Jensen, D., P. B. Christensen and S. Rysgaard. 1999. Oxygen and nutrient dynamics within mats of the filamentous acroalga *Chaetomorpha linum*. *Estuaries* 22:31-36.

Krause-Jensen, D., K. McGlathery, S. Rysgaard and P.B. Christensen. 1996. Production within dense mats of *Chaetomorpha linum* in relation to light and nutrient availability. *Marine Ecology Progress Series* 134:207-216.

Krouse, D.L. 1968. Effects of dissolved oxygen, temperature and salinity on survival of young striped bass, *Roccus saxatilis*. University of Maine. Orono, ME.

Los Angeles Regional Water Quality Control Board (LARWQCB). 2005a. Total Maximum Daily Load for Metals-- Los Angeles River and Tributaries. Los Angeles Regional Water Quality Control Board. Los Angeles, CA.

LARWQCB. 2005b. Total Maximum Daily Load for Metals in Ballona Creek. Los Angeles Regional Water Quality Control Board. Los Angles, CA.

LARWQCB. 2005c. Total Maximum Daily Load for Toxic Pollutants in Ballona Creek Estuary. Los Angeles Regional Water Quality Control Board. Los Angeles, CA.

Lee, K.S., F.T. Short and D.M. Burdick. 2004. Development of a nutrient pollution indicator using the seagrass, *Zostera marina*, along nutrient gradients in three New England estuaries. *Aquatic Botany* 78:197-216.

Lewis, L.J. and T.C. Kelly. 2001. A short-term study of the effects of algal mats on the distribution and behavioural ecology of estuarine birds. *Bird Study* 48:354-360.

Llanso, R.J. 1991. Tolerance of low dissolved oxygen and hydrogen sulfide by the polychaete Strepblospio benedicti. *Journal of Experimental Marine Biology and Ecology* 153:165-178.

Llanso, R.J. and R.J. Diaz. 1994. Tolerance to dissolved oxygen by the tubicolous polychaete *Loimia medusa*. *Journal of the Marine Biological Association of the United Kingdom* 74:143-148.

Lopes, R.J., M.A. Pardal and J.C. Marques. 2000. Impact of macroalgal blooms and wader predation on intertidal macroinvertebrates: experimental evidence from the Mondego estuary (Portugal). *Journal of Experimental Marine Biology and Ecology* 249:165-179.

Malone, T.C. 1977. Environmental regulation of phytoplankton productivity in the lower Hudson Estuary. *Estuarine Coastal and Shelf Science* 5:151-171.

Markager, S. and K. Sand-Jensen. 1990. Heterotrophic growth of *Ulva lactuca* (*Chlorohyceae*). *Journal of Phycology* 26:670-673.

Matthews, K.R. and N.H. Berg. 1997. Rainbow trout responses to water temperature and dissolved oxygen stress in two southern California stream pools. *Journal of Fish Biology* 50:50-67.

McGlathery, K. 2001. Macroalgal blooms contribute to the decline of seagrass in nutrient-enriched coastal waters. *Journal of Phycology* 37:453-456.

McGlathery, K.J., D. Krause-Jensen, S. Rysgaard and P. B. Christensen. 1987. Patterns of ammonium uptake within dense mats of the filamentous macroalga *Chaetomorpha linum*. *Aquatic Botany* 59:99-115.

McNatt, R.A. and J.A. Rice. 2004. Hypoxia-induced growth rate reduction in two juvenile estuary-dependent fishes. *Journal of Experimental Marine Biology and Ecology* 311:147-156.

Mills, E. 2003. State of the Gulf Report: Nutrient Indicators. NOAA Office of Ocean and Coastal Resource Management, Gulf of Maine Summit Planning Committee. Silver Springs, MD.

Muli, J.R. and K.M. Mavuti. 2001. The benthic macrofauna community of Kenyan waters of Lake Victoria. *Hydrobiologia* 458:83-90.

Niklitschek, E.J. 2001. Bioeneretics modeling and assessment of suitable habitat for juvenile Atlantic and shortnose sturgeons in Chesapeake Bay. University of Maryland. College Park, MD.

Nimmermark, S. 2004. Odour impact: Odor release, dispersion, and influence on human well-being with specific focus on animal production. Swedish University of Agricultural Sciences. Alnarp.

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

PBS&J. 2002. Nutrient Criteria Study for the Guadalupe River Basin. Austin, Texas.

Peckol, P. and J.S. Rivers. 1995. Contribution by macroalgal mats to primary production of a shallow embayment under high and low nitrogen loading rates. *Estuarine Coastal and Shelf Science* 44:451-465.

Pedersen, O., T. Binzer and J. Borum. 2004. Sulphide intrusion in eelgrass (*Zostera marina L.*). *Plant, Cell and Environment* 27:595-602.
Pihl, L., S.P. Baden, R.J. Diaz and L.C. Schaffer. 1992. Hypoxia-induced structural changes in the diet of bottom-feeding fish and crustacea. *Marine Biology* 112:349-361.
Poucher, S. and L. Coiro. 1997. Effects of low dissolved oxygen on saltwater animals: Morandum to D.C. Miller *in* A.E.D. US Environmental Protection Agency. Narragansett, RI.
Powers, S.P., C.H. Peterson, R.R. Christian, E. Sullivan, M.J. Powers, M.J. Bishop and C.P. Buzzelli. 2005. Effects of eutrophication on bottom habitat and prey resources of demersal fishes. *Marine Ecology-Progress Series* 302:233-243.
Pregnall, A.M., R.D. Smith, T.A. Kursar and R.S. Alberte. 1984. Metabolic adaptation of *Zostera marina* (eelgrass) to diurnal periods of root anoxia. *Marine Biology* 83:141-147.
Rabalais, N.N. and D.Harper, editors. 1992. Studies of benthic biota in areas affected by moderate and severe hypoxia. NOAA Coastal Ocean Program, Texas A&M Sea Grant. College Station, TX.
Rabalais, N.N., R.E. Turner, D. Justic, Q. Dortch, J.W.J. Wiseman and B.K. SenGupta. 1996. Nutrient change in the Mississippi River and system responses on the adjacent continental shelf. *Estuaries* 19:386-407.
Rabalais, N.N., R.E. Turner and J.W.J. Wiseman. 2001. Hypoxia in the Gulf of Mexico. *Journal of Environmental Quality* 30:320-329.
Rabalais, N.N., R.E. Turner and W.J. Wiseman. 2002. Gulf of Mexico hypoxia, A.K.A. "The dead zone". *Annual Review of Ecology and Systematics* 33:235-263.
Raffaelli, D., P. Balls, S. Way, I.J. Patterson, S. Hohmann and N. Corp. 1999. Major long-term changes in the ecology of the Ythan estuary, Aberdeenshire, Scotland; how important are physical factors? *Aquatic Conservation: Marine and Freshwater Ecosystems* 9:219-236.
Robblee, M.B., T.R. Barber, P.R. Carlson, M.J. Durako, J.W. Fourqurean, L.K. Muehlstein, D. Porter, L.A. Yarbro, R.T. Zieman and J. C. Zieman. 1991. Mass mortality of the tropical seagrass *Thalassia-Testudinum* in Florida Bay (USA). *Marine Ecology Progress Series* 71:297-299.
Roman, M., A.L. Gauzens, W.K. Rhinehart and J.R. White. 1993. Effects of low dissolved oxygen water on Chesapeake Bay zooplankton. *Limnology and Oceanography* 38:1603-1614.
Rothschild, B.J. 1990. Development of a sampling expert system: "FISHMAP". Marlyand Department of Natural Resources and US Fish and Wildlife Service Project No. F171-89-008. University of Maryland CEES Ref. No. (UMCEES) CBL 90-090. Chesapeake Biological Laboratory. Solomons, MD.
Rue, E.L. and K.W. Bruland. 2001. Domoic acid binds iron and copper: a possible role for the toxin produced by the marine diatom *Pseudonitzchia*. *Marine Chemistry* 76:127-134.
Sagasti, A., L.C. Schaffner and J. E. Duffy. 2000. Epifaunal communites thrive in an estuary with hypoxic episodes. *Estuaries* 23:474-487.
Saksena, V.P. and E.B. Joseph. 1972. Dissolved oxygen requirements of newly-hatched larvae of the striped blenny (*Chasmodes bosquianus*) the naked goby (*Gobiosoma bosci*) and the skillet fish (*Gobiesox strumosus*). *Chesapeake Science* 13:23-28.
San-Jensen, K. 1977. Effect of epiphytes on eelgrass photosynthesis. *Aquatic Botany* 3:55-63.
Santoro, E.D. 2004. Delaware Estuary Monitoring Report. Delaware River Basin Commission. Philadelphia, PA.

Santos, T.M., J.A. Cabral, R.J. Lopes, M. Pardal, J.C. Marques and J.Goss-Custard. 2005. Competition for feeding in waders: a case study in an estuary of south temperate Europe (Mondego, Portugal). *Hydrobiologia* 544:155-166.

Scholin, C.A., F. Gulland, G.J. Doucette, S. Benson, M. Busman, F.P. Chavez, J. Cordaro, R. DeLong, A. DeVogelaere, J. Harvey, M. Haulena, K. Lefebvre, T. Lipscomb, S. Loscutoff, L.J. Lowenstine III, R.M., P.E. Miller, W.A. McLellan, P.D.R. Moeller, C.L. Powell, T. Rowles, P. Silvagni, M. Silver, T. Spraker, V. Trainer and F.M. VanDolah. 2000. Mortality of sea lions along the central California coast linked to toxic diatom bloom. *Nature* 403:80-84.

San Diego Regional Water Quality Control Board (SDRWQCB). 2005. Total Maximum Daily Loads for Indicator Bacteria Project I-- Beaches in Creeks in San Diego Region-- Technical Report. San Diego Regional Water Quality Control Board. San Diego, CA.

Secor, D.H. and T.E. Gunderson. 1998. Effects of hypoxia and temperature on survival growth and respiration of juvenile Atlantic Sturgeon, *Acipenser oxyrinchus*. *Fisheries Bulletin* 96:603-613.

Secor, D.H. and E.J. Niklitschek. 2004. Sensitivity of sturgeons to environmental hypoxia: Physiological and ecological evidence, *in:* Fish physiology, Toxicology and Water Quality-- Proceedings of the Sixth International Symposium. La Paz, Mexico.

Seymour, N.R., A.G. Miller and D.J.Garbary. 2002. Decline of Canada geese (*Branta canadensis*) and common goldeneye (*Bucehala clandula*) associated with a collapse of eelgrass (*Zostera marina*) in a Nova Scotia estuary. *Helgoland Marine Research* 56:198-202.

Sfriso, A., A. Marcomini and B. Pavoni. 1987. Relationships between macroagal biomass and nutrient concentrations in a hypertrophic area of the Venice Lagoon. *Marine Environmental Research* 22:297-312.

Simpson, D.G. 1995. Cooperative Interagency Resource Assessment. A Study of Marine Recreational Fisheries in Connecticut. Federal Aid to Sport Fish Recreation, F54R. Connecticut Department of Environmental Protection, Bureau of Natural Resources, Fisheries Division.

Smith, R.D., A.M. Pregnall and R.S. Alberte. 1988. Effects of anaerobiosis on root metabolism of *Zostera marina* (eelgrass): Implications for survival in reducing sediments. *Marine Biology* 98:130-141.

Smith, S.V. and J.T. Hollibaugh. 1989. Carbon-controlled nitrogen cycling in a marine 'macrocosm': an ecosystem-scale model for managing cultural eutrophication. *Marine Ecology Progress Series* 52:103-109.

Smith, V.H., G.D. Tilman and J.C. Nekola. 1999. Eutrophication: impacts of excess nutrient inputs on freshwater, marine, and terrestrail ecosystems. *Environmental Pollution* 100:179-196.

Stalder, L.C. and N.H. Marcus. 1997. Zooplankto responses to hypoxia: Behavioral patterns and survival of three species of calanoid copepods. *Marine Biology* 127:599-607.

Stanley, D.W. and S.W. Nixon. 1992. Stratification and bottom-water hypoxia in the Pamlico River Estuary. *Estuaries* 15:270-281.

Steward, J.S., R.W. Virnstein, L.J. Morris and E.F. Lowe. 2005. Setting seagrass depth, coverage, and light targets for the Indian River Lagoon system, Florida. *Estuaries* 28:923-935.

Sutula, M., C. Creager and G. Wortham. 2007. Technical Approach to Develop Nutrient Numeric Endpoints for California Estuaries. Technical Report 516. Southern California Coastal Water Research Project. Costa Mesa, CA.

Sutula, M., K. Kamer and J. Cable. 2004. Sediments as a non-point source of nutrients to Malibu Lagoon, California (USA). Technical Report 441. Southern California Coastal Water Research Project. Westminster, CA.

Swason, C., T. Reid, P.S. Young and J.J. Cech. 2000. Comparative environmental tolerances of threatened delta smelt (*Hypomesus transpacificus*) and introduced wakasagi (*H. nipponensis*) in an altered California estuary. *Oecologia* 123:384-390.

TetraTech. 2003. Lake Elsinore and Canyon Lake Nutrient Source Assessment- Final Report. Santa Ana Water Project Authority (SAWPA). LaFayette, CA.

TetraTech. 2004. San Jacinto Nutrient Management Plan- Final Report. Lake Elsinor & San Jacinto Watersheds Authority (LESJWA). LaFayette, CA.

Tomasko, D.A., and B.E. Lapointe. 1991. Productivity and biomass of *Thalassia-Testudinum* as related to water column nutrient availability and epiphyte levels- field observations and experimental studies. *Marine Ecology Progress Series* 75:9-17.

Trainer, V., B. Hickey and R. Horner. 2002. Biological and physical dynamics of domoic acid production off the Wasington Coast. *Limnology and Oceanography* 47:1438-1446.

Turner, R.E., N. Qureshi, N.N. Rabalais, Q. Dortch, D. Justic, R.F. Shaw and J. Cope. 1998. Fluctuating silicate: nitrate ratios and coastal plankton food webs. *Proceedings of the National Academy of Sciences* 95:13,048-013.

Turner, R.E., W.W. Schroeder and J.W.J. Wiseman. 1987. The role of stratification in the deoxygenation of mobile bay and adjacent shelf bottom Waters. *Estuaries* 10:13-19.

Twilley, R.R. 1985. The exchange of organic carbon in basin mangrove forests in a southwest Florida estuary. *Estuarine Coastal and Shelf Science* 20:543-557.

United States Environmental Protection Agency (USEPA). 1986. Ambient Water Quality Criteria for Dissolved Oxygen (Freshwater). U.A. Environmental Protection Agency, Washinton D.C.

USEPA. 2000. Ambient Aquatic Life Water Quality Criteria for Dissolved Oxygen (Saltwater): Cape Cod to Cape Hatteras. U.S. Environmental Protection Agency, Washington D.C.

USEPA. 2001. Nutrient Criteria Technical Guidance Manual: Estuarine and Coastal Marine Waters. U.S. Environmental Protection Agency, Washington D.C.

USEPA. 2003. Ambient Water Quality Criteria for Dissolved Oxygen, Water Clarity, and Chlorophyll a for Chesapeake Bay and Its Tidal Tributaries. U.S. Environmental Protection Agency, Washington D.C.

USEPA. 2005. Technical Approach to Develop Nutrient Numeric Endpoints for California. Page 115 *in* W. D. C. U.S. Environmental Protection Agency, editor. Tetra Tech, INC., Lafayette, CA.

USEPA. 2007. Technical Approach to Develop Nutrient Numeric Endpoints for California Estuaries. Southern California Coastal Water Research Project, Costa Mesa, CA.

Valiela, I., K. Foreman, M. LaMontagne, D. Hersh, J. Costa, P. Peckol, B. DeMeo-Andreson, C. D'Avanzo, M. Babione, C. Sham, J. Brawley and K. Lajtha. 1992. Couplings of watersheds and coastal waters: Sources and consequences of nutrient enrichment in Waquoit Bay, Massachusetts. *Estuaries* 15:433-457.

Valiela, I., J. McClelland, J. Hauxwell, P. J. Behr, D. Hersh and K. Foreman. 1997. Macroagal blooms in shallow estuaries: Controls and ecophysiological and ecosystem consequences. *Limnology and Oceanography* 42:1105-1118.

Vallino, J.J. and C.S. Hopkinson. 1998. Estimation of dispersion and characeristic mixing times in Plum Island Sound Estuary. *Estuarine Coastal and Shelf Science* 46:333-350.

vanKatwijk, M.M., L.H.T. Vergeer, G.H.W. Schmitz and J.G.M. Roelofs. 1997. Ammonium toxicity in eelgrass *Zostera marina*. *Marine Ecology Progress Series* 157:159-173.

vanWijck, C., C.J. DeGroot and P. Grillas. 1992. The effect of anaerobic sediment on the growth of *Potamogeton pectinatusL*: The role of organic matter, sulfide and ferrous iron. *Aquatic Botany* 44:31-49.

Vargo, S.L. and A.N. Sastry. 1977. Interspecific differences in toleranceof *Eurytemora affinis* and *Acartia tonsa* from an estuarine anoxic vasin to low dissolved oxygen and hydrogen sulfide. pp. 219-226 *in*: D.S. McCluskey and A.J. Berry, editors, Physiology and Behavior of Marine Organisms. 12th European Marine Biology Symposium. Pergamon Press. Oxford, U.K.

Villac, M.C., D.L. Roelke, F.P. Chavez, L.A. Cifuentes and G. Fryxell. 1993. *Pseudonitzschia australis Frenguelli* and related species from the west coast of the U.S.A.: occurrence and domoic acid production. *Journal of Shellfish Research* 12:457-465.

Vismann, B. 1990. Sulfide detoxification and tolerance in Nereis (dediste) diversicolor and Nereis (neanthes) virens (*Annelida: Polychaeta*). *Marine Ecology Progress Series* 59:229-238.

Welch, E.B., R.R. Horner and C.R. Patmont. 1989. Prediction of nuisance periphytic biomass: a management approach. *Water Research* 23:401-405.

Wells, M., C. Trick, W. Cochlan, M. Hughes and V. Taylor. 2005. Domoic acid: The synergy of iron, copper, and the toxicity of diatoms. *Limnology and Oceanography.* 50: 1908-1917.

Zimmerman, R.C., A. Cabello-Pasini and R.S. Alberte. 1994. Modeling daily production of aquatic macrophytes from irradiance measurements: a comparative analysis. *Marine Ecology Progress Series* 114:185-196.

Zimmerman, R.C., D.G. Kohrs, D.L. Steller and R.S. Alberte. 1995a. Carbon partitioning in eelgrass regulation by photosynthesis and the response to daily light-dark cycles. *Plant Physiology* 108:1665-1671.

Zimmerman, R.C., J.L. Reguzzoni and R.S. Alberte. 1995b. Eelgrass (*Zostera marina L.*) Transplants in San Francisco Bay: Role of light availability on metabolism, growth and survival. *Aquatic Botany* 51:67-86.

Zimmermann, C. F., and J. R. Montgomery. 1984. Effects of a decomposing drift algal mat on sediment pore water nutrient concentrations in a Florida seagrass bed. *Marine Ecology Progress Series* 19:299-302.

APPENDIX A: COMPARISON OF APPROACHES TO SETTING NUMERIC ENDPOINTS: AN ANALYSIS OF EXISTING DATA

Introduction

One important consideration in the implementation of the NNE framework is how the endpoints for the various indicators are derived. Currently, two approaches could be taken: 1) an effects-based approach and 2) the percentile approach (USEPA 2001). In brief, the percentile approach consists of a statistical methodology in which endpoints are selected by setting the 75th percentile of reference systems or the 25th percentile of data derived from an ambient survey of indicators in population of estuaries. The effects-based approach relies on scientific data, which links impacts to indicator species or communities (e.g., mortality, reduced reproduction) to certain thresholds in the biological response variable (e.g., hypoxia). The goal of this appendix is to utilize existing data was to compare the different approaches to setting thresholds for numeric endpoints and describe how specific endpoints can be applied to determine the probability that a specific system would exceed a specific threshold of impairment. We utilize dissolved oxygen (DO) for this purpose, as the most data exists on thresholds to physiological impairment of estuarine species and long term monitoring data sets were available for California perennially tidal lagoons. This analysis of data is intended as a demonstration only and is not intended to advocate use of any particular methods for establishing thresholds. Particularly for the percentile approach, there is no data for West Coast estuaries that suggest that the 75th percentile of reference systems would necessarily correspond to the 25th percentile of a probabilistic survey of all systems. Furthermore, a probabilistic survey of estuaries was not available and thus, given the limitations on the data, such thresholds established in this appendix would not be valid.

METHODS

Perennially tidal lagoons were selected for the analysis of existing data because of the number of these systems for which TMDL activities are targeted and because of the availability of several data sets for this estuarine class. The criteria to select data sets from these systems included: 1) data from established monitoring programs with a quality assurance plan in place, 2) continuous monitoring should have included, at minimum, hourly DO concentrations, temperature, and conductivity for minimum of 6 months. Additional data on algal monitoring, nutrient loading, water clarity, chlorophyll a, etc. was also sought; however, most systems only had data on DO, conductivity, and temperature. Because DO was the common variable measured among the lagoons and because it has an extensive history as an indicator for eutrophication in other states, we have chosen to focus our demonstrative data analysis on DO.

We focused the analysis on data obtained for four perennially tidal lagoons: Elkhorn Slough, San Diegito Lagoon, San Elijo Lagoon, and Tijuana Estuary (Table A-1). Many of these lagoons have monitoring stations at several locations, which provide information on the spatial variability of periods of low DO (Table A-2). These data sets were assessed for quality and completeness and analyzed for periods of low DO, and the duration and frequency of these events.

Table A-1. Available data for perennially tidal lagoons.

Estuary	Dissolved Oxygen	SAV Coverage	Water Chlorophyll a	Sediment Chlorophyll a	Macroalgal biomass	TN and TP loading
Elkhorn Slough	X		X			
San Diegito Lagoon	X					
San Elijo Lagoon	X					
Tijuana Estuary	X		X			

Table A-2. Lagoons for which continuous data sets for dissolved oxygen concentration were available and the monitoring duration.

Estuary	Estuarine Class	Monitoring Instrument	Sites	Monitoring Duration
Elkhorn Slough	Perennially Tidal	YSI 6600/YSI 6600EDS datalogger	Azevedo Pond	1995-2005
			South Marsh	1995-2005
			North Marsh	1999-2005
			Vierra Mouth	2003-2005
San Diegito Lagoon	Perennially Tidal	YSI 600XLM data sonde	Grand Avenue Bridge (GAB)	2003-2007
San Elijo Lagoon	Perennially Tidal	YSI 600XLM data sonde	I-5	2003-2007
			Rail Road Trestle	2002-2007
Tijuana Estuary	Perennially Tidal	YSI 6600/YSI 6600EDS datalogger	Model Marsh	2000-2005
			River Channel	2002-2005
			Oneota Slough	1996-2005
			Tidal Linkage	1997-2005

The first method to establish numeric endpoints is the percentile approach (USEPA 2001). The percentile approach consists of development of criteria based statistical analysis and the guidelines suggest two ways to establish criteria. The first is to identify pristine "reference" sites that are relatively undisturbed. The 75th percentile of the frequency distributions of complied data for reference sites of the same estuarine class could be used to set the numeric criteria for that class. In the absence of reference sites, the 25th percentile of the frequency distributions of compiled data sets for a probability-based sample of systems of the same estuarine class could be used to set numeric endpoints.

No reference systems were available for this study, nor were probability-based ambient survey data available for this comparison. Nevertheless, to illustrate how the endpoints would be developed, we therefore applied the 25th percentile methodology to the existing data from the four perennially tidal lagoons. To determine the 25th percentile, all of the DO concentration data was complied into a single data set that was ordered from lowest value to highest value. These data were used to create a cumulative distribution function plot where probabilities of a particular DO concentration occurring in any given lagoon were calculated. The DO numeric criterion was the concentration that occurred at the 25th percentile of the entire data set.

Because effects-based data on the impacts of hypoxia to pelagic and benthic species in California lagoons is limited, we utilized the established DO criteria for the Chesapeake Bay Program to characterize the probability that any of the lagoons suffer DO concentrations that threaten typical estuarine beneficial uses (as determined by the Chesapeake Bay Program). We applied the Chesapeake Bay criteria for migratory fish spawning and nursery use, shallow water bay grass use, and open water fish and shellfish use. The differences in thresholds derived from effect-based data (i.e., the Chesapeake Bay criteria) versus a percentile approach were contrasted using data from these four lagoons.

Results

Dissolved Oxygen Observations

All of the perennially tidal lagoons investigated were subject to low DO events periodically throughout the year. The most extreme hypoxic events occurred during the late spring/early summer (onset in April and extending through October) with some lagoons suffering hypoxic events for short periods during the winter, though hypoxia was always more intense and longer in duration during the summer compared to the winter. The most intensive hypoxia occurred during June and July for all estuaries for which data was available. Dissolved oxygen concentrations varied widely over the course of a single day. Concentrations were highest during the daylight hours, when photosynthetic organisms emit oxygen into surface waters. During the night, DO levels are drawn down by organisms that are respiring organic carbon in the water column (Figures A-1 and A-2). There also appeared to be a tidal component to the diel variability in DO concentration, where spring tides tended to have less day/night variability and neap tides had the greatest difference in daytime/nighttime DO. This may be because increased mixing during spring tides homogenizes the water column muting day/night DO variability.

San Elijo Lagoon had periodic low DO concentrations throughout the year, with the lowest values occurring during the nighttime hours. These were most frequent during the summer, typically lasting for two consecutive months from May through August. During this period, DO is typically less than 4.8 mg/L (continuous limit for growth effects) for the entire two month period, often dipping to less than 2.3 mg/L (continuous limit for survival) for 24 hours or more. The railroad trestle site typically had lower summer time DO concentrations than the I-5 site, and these hypoxic events lasted for a longer duration. There was also a low DO event in the winter months (December-January) during which DO concentrations dropped to less that 4.8 mg/L and 2.3 mg/L for up to three weeks. This was observed at both sites but for different durations (I-5 having longer duration events).

San Diegito Lagoon also suffered periods of low DO throughout the year, with most events occurring during the nighttime hours and recovering during the day. However, a seasonal pattern in continuous low DO stretches was not discernable in San Diegito Lagoon. Continuous low DO events occurred throughout the year, with specific events varying from month to month and year to year. However, low DO events did not typically last as long as San Elijo. Hypoxic events in San Diegito typically lasted 24 - 72 hours during the worst events.

Continuous hypoxia in Elkhorn Slough was not common. While nighttime hypoxia was common in the early spring through the late fall, with DO concentrations dropping to below 2.3 mg/L, hypoxia lasting more than 24 hours was rare, occurring only a few days per year in the summer time. There was considerable variation in DO concentrations between sites in Elkhorn Slough. Locations in the upper part of the slough tended to have more hypoxic events than the lower slough. This may be because of increased flushing in the lower slough and higher nutrient loading in the upper slough, but without algal monitoring and nutrient loading data it is impossible to explain this variability.

Nighttime hypoxic events in Tijuana Estuary occurred throughout the year, but with greater frequency during the summer. Continuous low DO events in Tijuana Estuary occurred occasionally; however, during the summer months of July and August, it was very common that daytime DO concentrations would typically rise to values that were just greater than the 4.8 mg/L threshold for growth and only for a few hours. Summer hypoxic events did occasionally drop below the 2.3 mg/L threshold for survival, but concentrations recovered after 24 - 48 hours. As with Elkhorn Slough, there was considerable variation among sites in the Tijuana Estuary. The river channel had the highest frequency of hypoxic events and Oneota Slough the lowest.

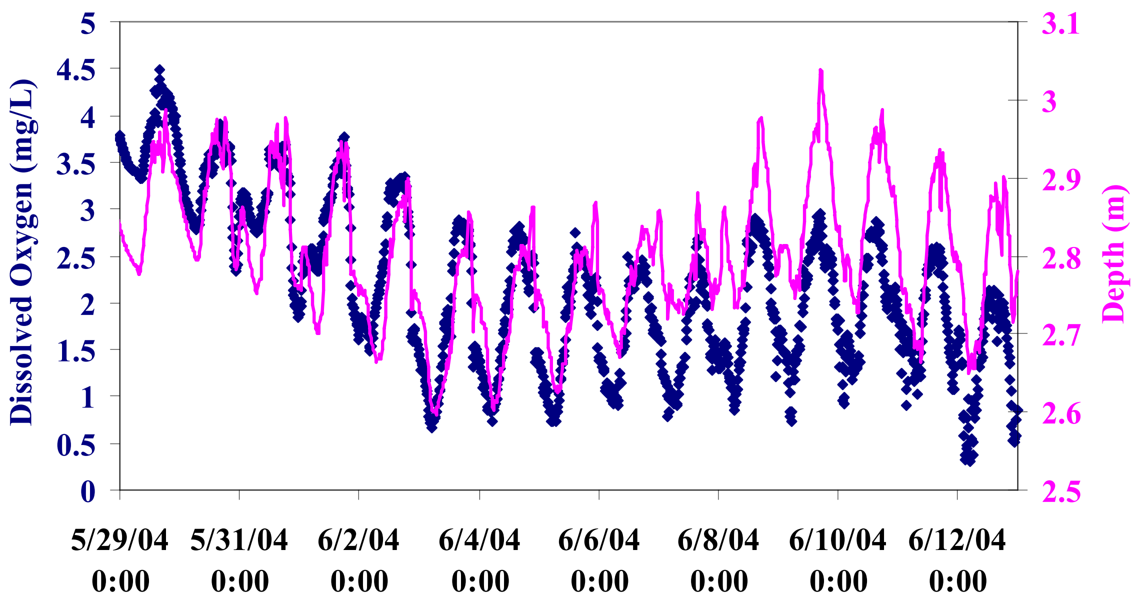

Figure A-1. Fortnightly variations late spring DO (blue diamonds) and depth (magenta line) in San Elijo Lagoon. DO minima occur at night at low tide. DO is lower than the continuous minimum for growth (4.8 mg/L) and is periodically lower than the limit for survival (2.3 mg/L).

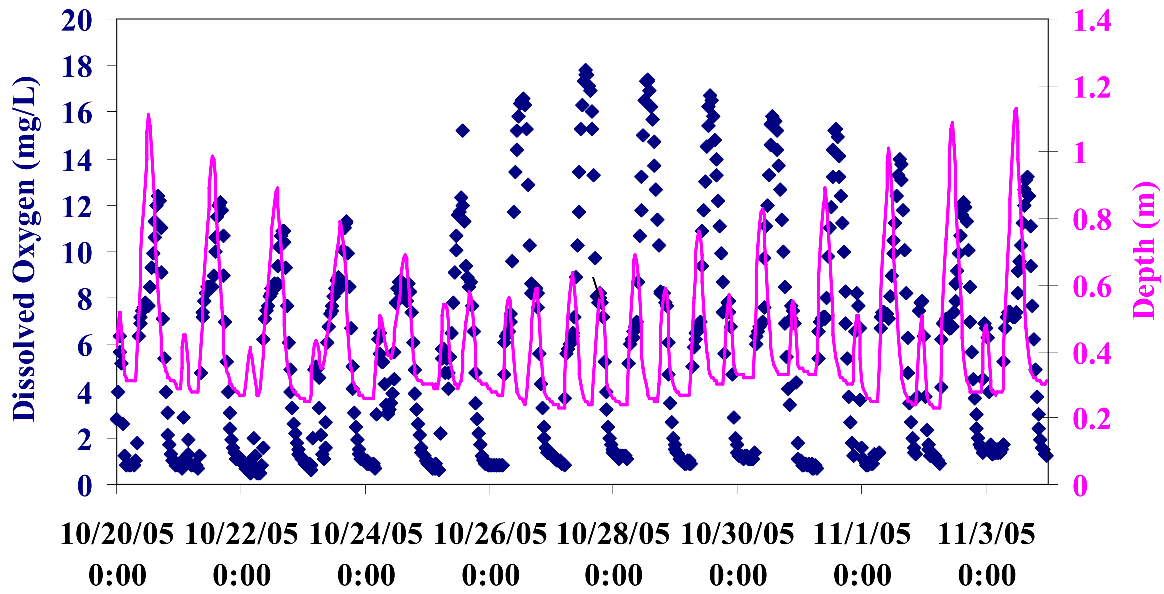

Figure A-2. Fortnightly variations fall dissolved oxygen (blue diamonds) and depth (magenta line) in Elkhorn Slough. Dissolved oxygen has large diel variability with minima occurring at night. There also appears to be a tidal component to the dissolved oxygen signal at where neap tides have greater diel variability than spring tides.

Characterization of Eutrophication Via Effect-Based Versus Percentile-Based Thresholds

We utilized the USEPA methodology of extracting the 25th percentile of the frequency distribution of DO measurements for all perennially tidal lagoons to demonstrate how a threshold DO concentration could be set using this approach (Figure A-3). It should be noted that the 25th percentile strongly depends on the precise dataset utilized to set the endpoint and that such endpoints do not necessarily protect beneficial uses. Figure A-4 is a box and whiskers plot that displays the 25th and 75th percentiles at the bottom and top of the box respectively (ends of the whiskers represent the 10th and 90th percentiles). The variation in the 25th percentile of each lagoon indicates the how variability in ambient data among lagoons can cause significant differences in established endpoints using the percentile method.

Dissolved oxygen concentrations in our data set ranged from 0 – 40 mg/L with values greater than 10 mg/L occurring less than 15% of the time in all lagoons measured. Of the entire suite of DO measurements in our data set, 25% of the instantaneous values were less than 3.7 mg/L (Figure A-3) thus the threshold for an instantaneous minimum value based on the statistical approach would be 3.7 mg/L. Using this numeric criterion, we were able to calculate the probability that each of our monitoring sites would exceed this threshold by establishing frequency distributions of DO data in each lagoon (Table A-3).

The probability that any site in each of the lagoons exceeded this statistical threshold was variable indicating the spatial heterogeneity of biological response in each of the estuaries. Tijuana Estuary had the greatest number of exceedences in the River Channel site (64.2% probability of exceedence). However, sites within the Tijuana Estuary had variable probabilities of exceedence ranging from 27.2% to 64.2%. The probability of DO criterion exceedence in San Elijo was more consistent between the two sites ranging from 44.6 – 50 %. Elkhorn Slough had the lowest probability of exceeding the DO criterion.

Because the DO threshold generated from the percentile approach was only based on a limited data set for perennially tidal lagoons, we compared our value to the established criteria from other programs. Our 3.7 mg/L DO criterion falls in between the continuous DO limit for survival (2.3 mg/L) and growth (4.8 mg/L) as stated in the Virginian Province DO Criteria Report (USEPA 2000). Our threshold was slightly higher than the Chesapeake Bay Program's instantaneous DO criterion for open water habitat (3.2 mg/L). However, it was lower than the Chesapeake Bay program criterion for fish spawning and migration (5 mg/L). Thus, the percentile criteria would be protective of the open water fish and shellfish use as determined by the Chesapeake Bay study, but not protective of migratory fish spawning and nursery use. Thus, these beneficial uses would not be protected by the percentile approach. This highlights the need for effects based criteria that would be protective of all estuarine beneficial uses.

We also calculated the probabilities that each site would exceed the DO criteria established by the Chesapeake Bay Program for thresholds that would potentially translate to California lagoons (Table A-3). Probability of exceedence varied widely among each criterion for any given site. Figures A-5 through A-8 show the cumulative distribution plots for each estuary and how DO measurements in each lagoon compare to established thresholds. Site to site differences in exceedences of threshold values underlies the importance of selecting sites for monitoring in each lagoon that are representative of the habitat.

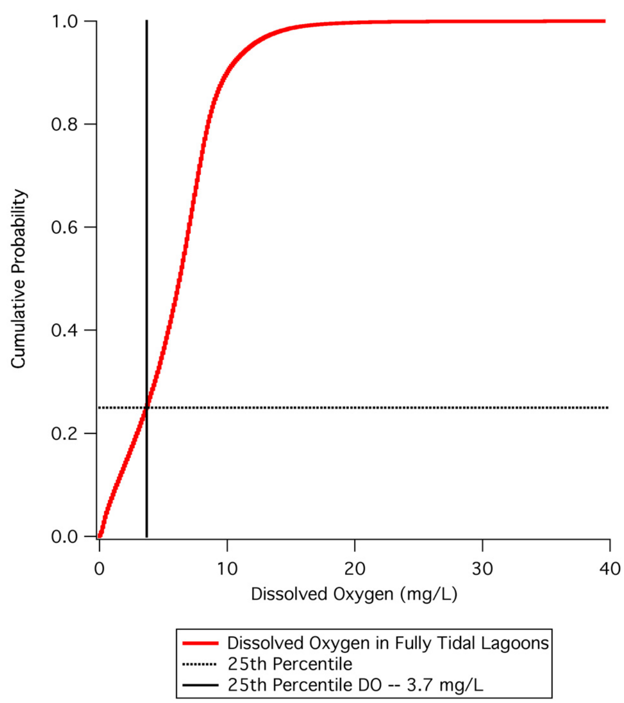

Figure A-3. Proposed USEPA method for determination of dissolved oxygen criteria for estuarine systems using the 25th percentile of all measurements made in perennially tidal coastal lagoons.

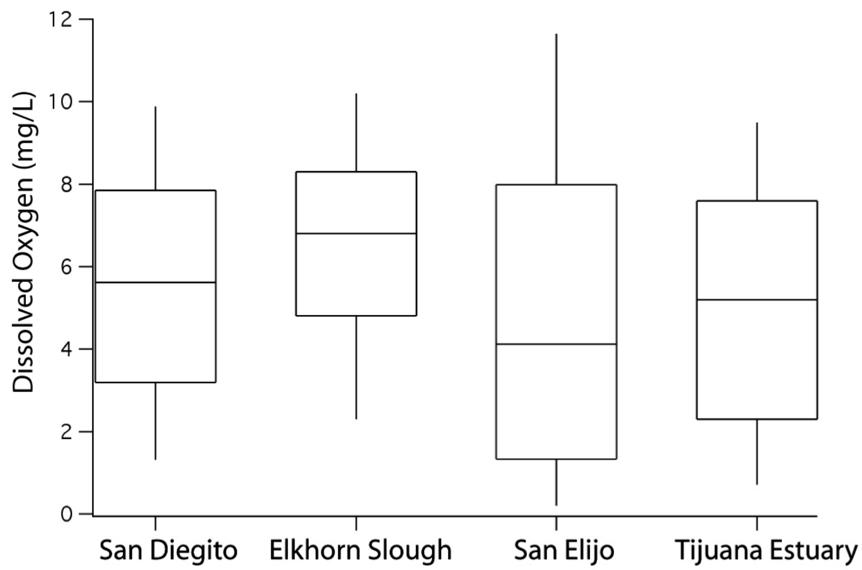

Figure A-4. Box and whiskers plot for all DO data for each of the lagoons. The bottom part of the box represents the 25th percentile for all data in a given lagoon. The variability in those 25th percentiles highlights the problem with utilizing the percentile method for establishing criteria, specifically that the established criteria is strongly dependent on the ambient data used.

Table A-3. Percent probability of threshold exceedence in instantaneous dissolved oxygen (DO) concentration using the criteria established by the USEPA methodology (25th percentile).

Estuary	Location	Probability of Exceedence of Percentile DO Criterion (3.7 mg/L)	Probability of Exceedence of Open Water Fish and Shellfish Use DO criteria (3.2 mg/L)	Probability of Exceedence of Migratory Fish Spawning and Nursery DO Criteria (5 mg/L)
Tijuana Estuary	Model Marsh	32.5	27.7	46.2
	Oneota Slough	27.2	22.7	40
	Tidal Linkage	44.2	40.6	54.2
	River Channel	64.2	60.8	73.7
Elkhorn Slough	Azevedo Pond	26.3	22.3	37.3
	North Marsh	25.8	21.4	38
	South Marsh	7.0	4.1	14.8
	Vierra Mouth	0.2	--	1.8
San Diegito Lagoon	GAB	29.8	25.1	43.0
San Elijo Lagoon	I-5	44.6	40.9	54.7
	Railroad Trestle	50.0	46.2	56.7

Table A-4. Percent probability of threshold exceedence in dissolved oxygen concentration by estuary using Chesapeake Bay criteria for equivalent habitat types.

Estuary	Location	Instantaneous Dissolved Oxygen Minimum		7-Day Dissolved Oxygen Mean		30-Day Dissolved Oxygen Mean	
		3.2* (mg/L)	5 (mg/L)#	4 (mg/L)*	6 (mg/L)#	5 (mg/L)*	5.5* (mg/L)
Tijuana Estuary	Model Marsh	27.7	46.2	19.2	60.8	45.8	51.4
	Oneota Slough	22.7	40	24.2	61.3	48.6	58.5
	Tidal Linkage	40.6	54.2	29.6	61.6	49.6	59.8
	River Channel	60.8	73.7	65.8	93.4	89.5	91.8
Elkhorn Slough	Azevedo Pond	22.3	37.3	6.4	44.7	18.8	38.7
	North Marsh	21.4	38	20	46	17.2	33.0
	South Marsh	4.1	14.8	5.1	26.3	8.9	17.2
	Vierra Mouth	--	1.8	--	3.0	--	--
San Diegito Lagoon	GAB	25.1	43.0	24.5	55.5	3.5	46.4
San Elijo Lagoon	I-5	40.9	54.7	47.0	67.4	59.8	66.6
	Railroad Trestle	46.2	56.7	42.9	55.8	--	--

* Open water fish and shellfish use; Shallow-water bay grass use
# Migratory fish spawning and nursery use

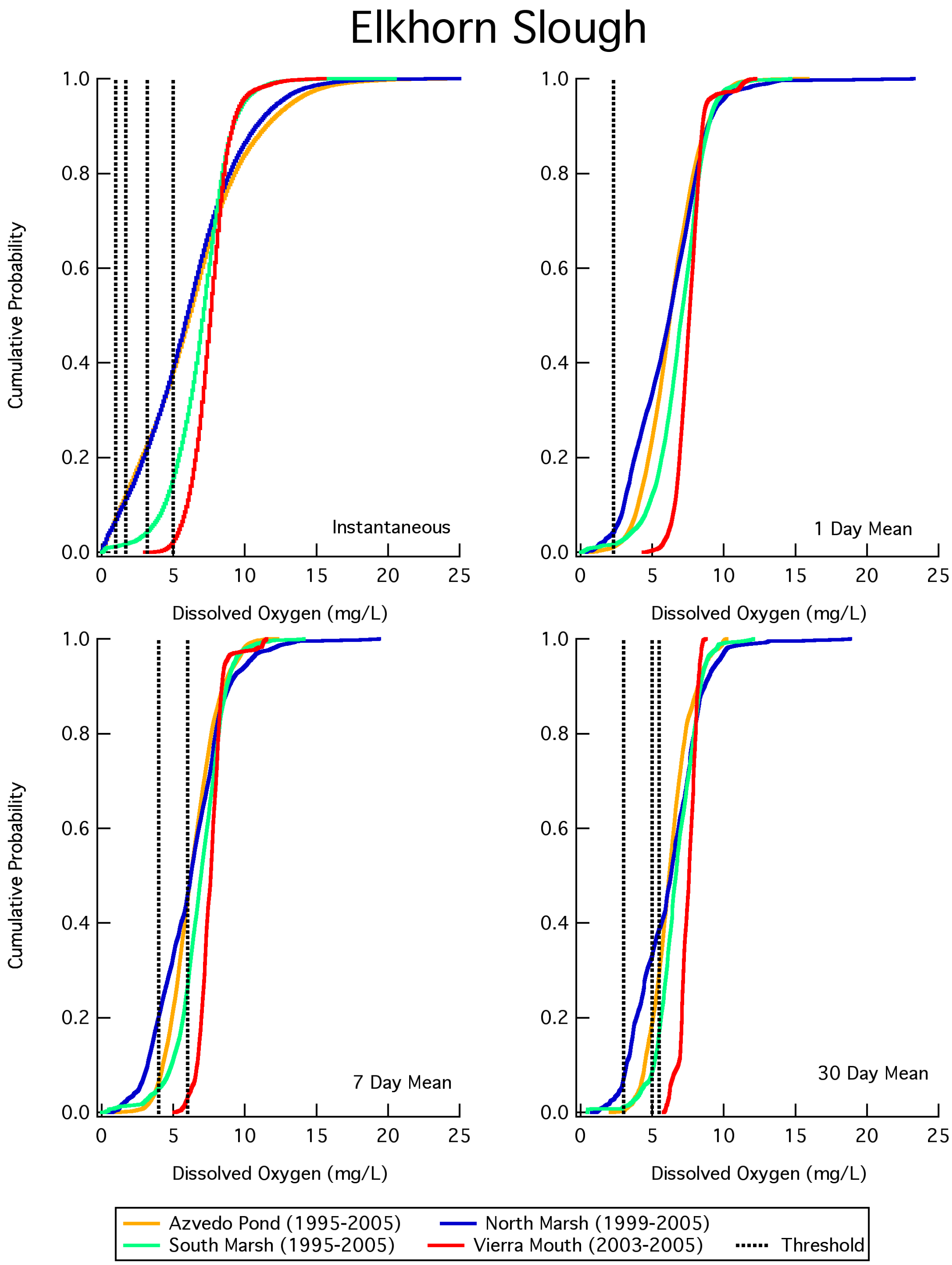

Figure A-5. Cumulative distribution plots for Elkhorn Slough. Panels represent the instantaneous, the 1-day mean, 7-day mean and 30-day mean dissolved oxygen concentrations. Dotted black lines represent the Chesapeake Bay dissolved oxygen criteria. All values to the left of the dotted line represent an exceedence of the threshold.

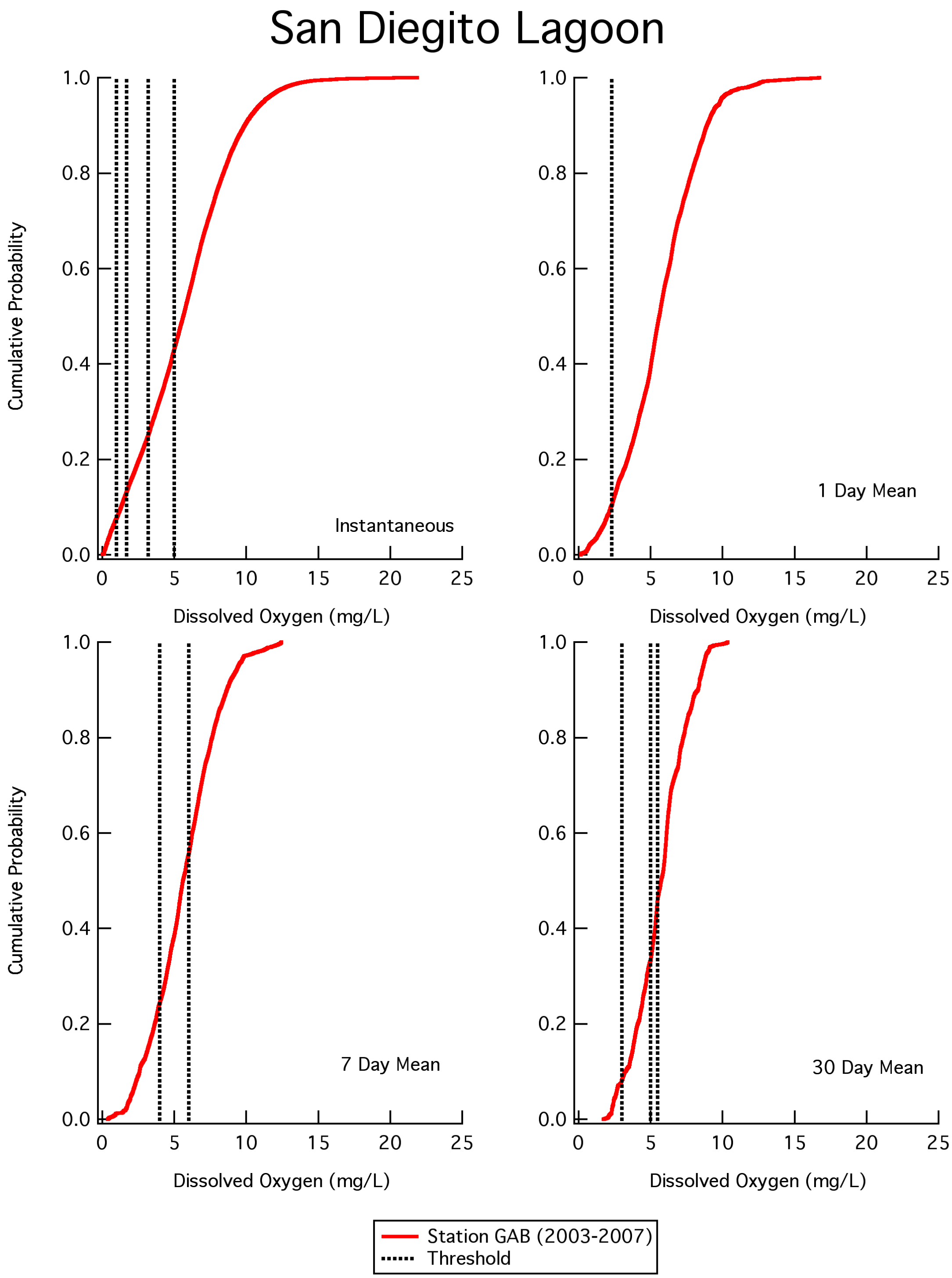

Figure A-6. Cumulative distribution plots for San Diegito Lagoon. Panels represent the instantaneous, the 1-day mean, 7-day mean and 30-day mean dissolved oxygen concentrations. Dotted black lines represent the Chesapeake Bay dissolved oxygen criteria . All values to the left of the dotted line represent an exceedence of the threshold.

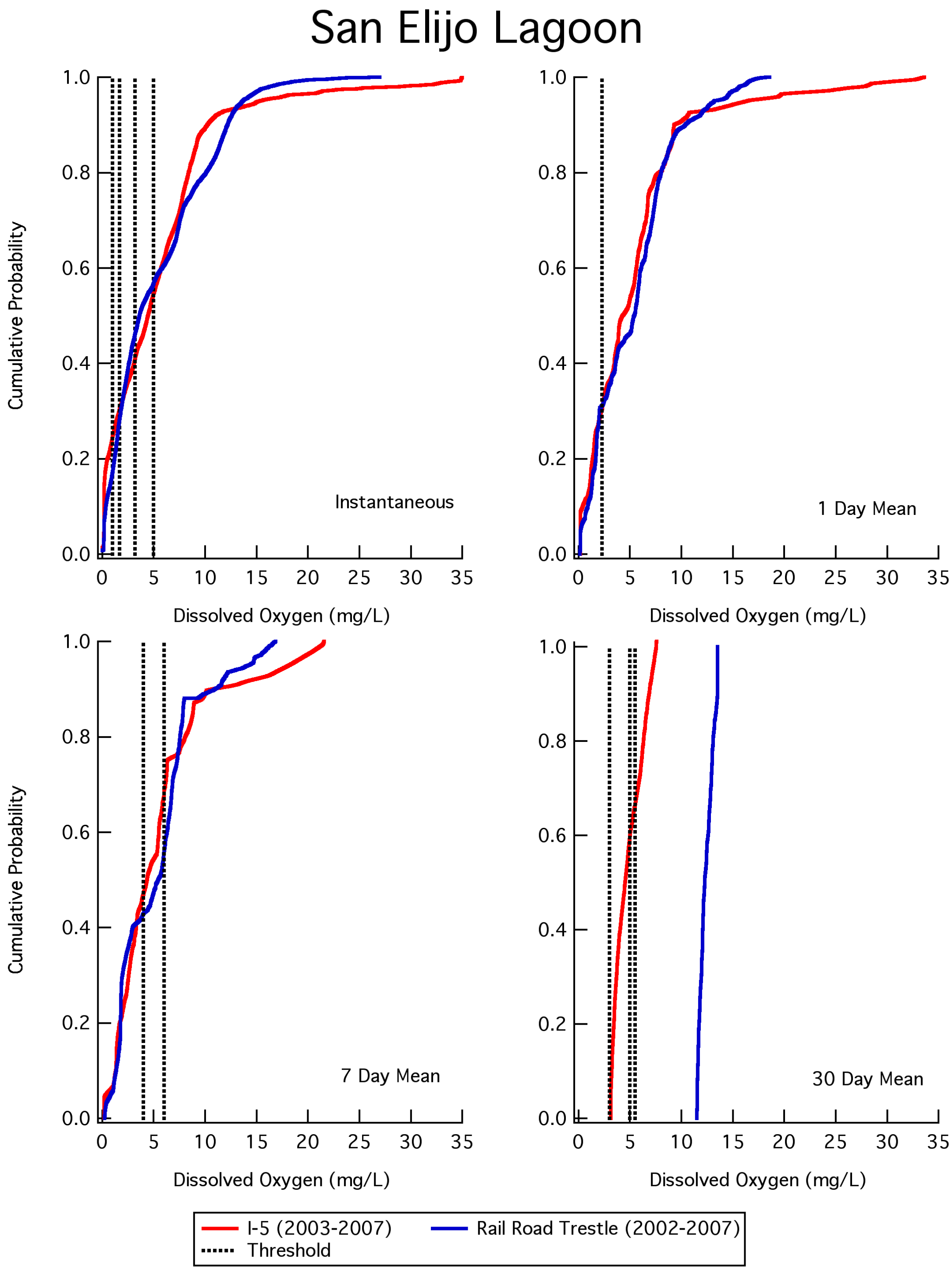

Figure A-7. Cumulative distribution plots for San Elijo Lagoon. Panels represent the instantaneous, the 1-day mean, 7-day mean and 30-day mean dissolved oxygen concentrations. Dotted black lines represent the Chesapeake Bay dissolved oxygen criteria. All values to the left of the dotted line represent an exceedence of the threshold.

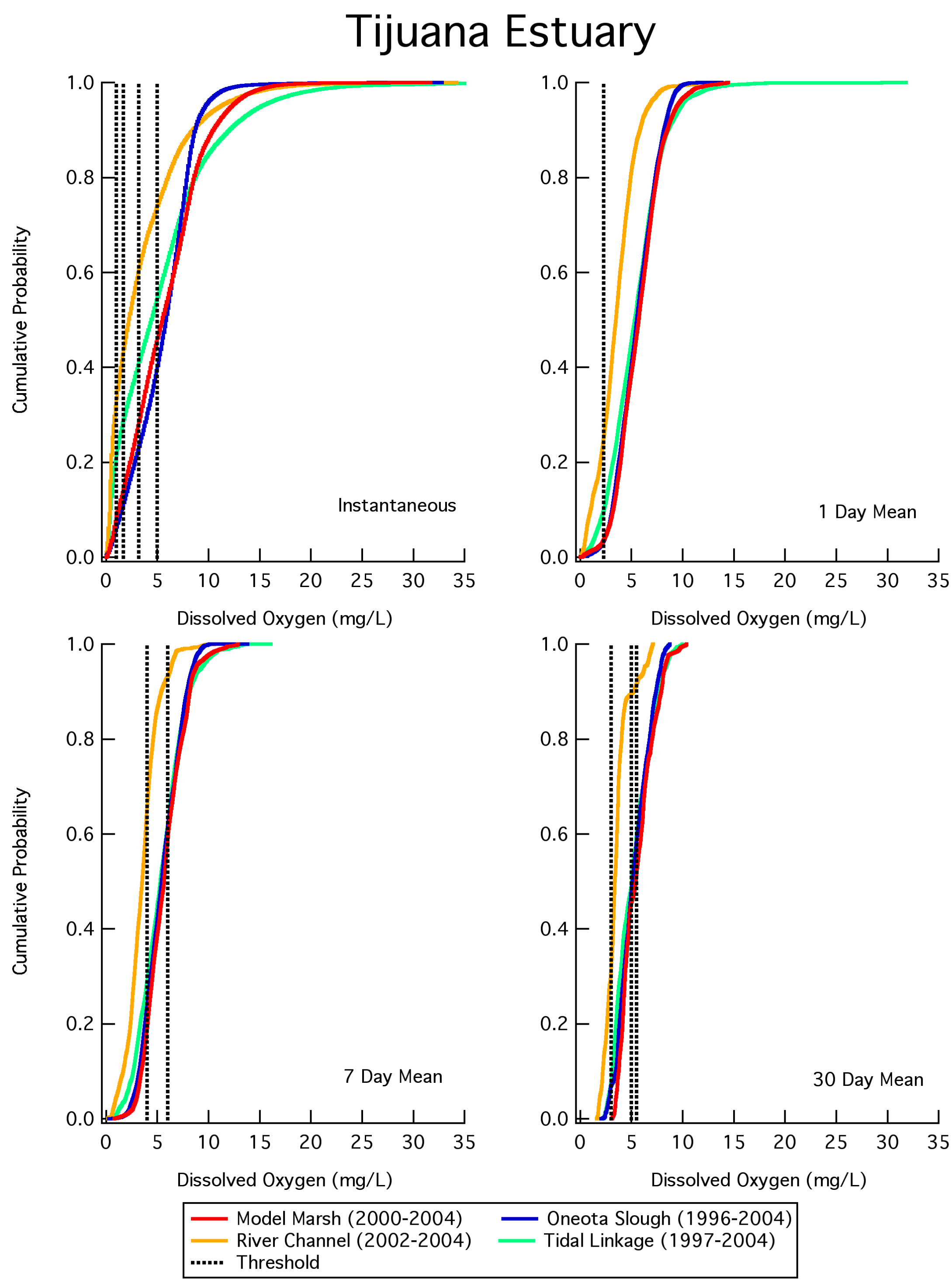

Figure A-8. Cumulative distribution plots for Tijuana River Estuary. Panels represent the instantaneous, the 1-day mean, 7-day mean and 30-day mean dissolved oxygen concentrations. Dotted black lines represent the Chesapeake Bay dissolved oxygen criteria. All values to the left of the dotted line represent an exceedence of the threshold.

Discussion

Our analysis of existing data illustrates that an effect-based approach is preferable to the percentile approach for establishing numeric endpoints. Effect-based endpoints provide direct linkages to estuarine management endpoints. Thus, whenever effects-based criteria are met, beneficial uses should be sustained. For DO, sufficient physiological effects data are available, particularly with respect to the effects of hypoxia on pelagic and benthic species (Tables A-5 and A-6). Even though some of this data is not directly applicable, it may be possible to extrapolate preliminary DO thresholds for California species using organisms within the same genus or family for which data exists. In order to accomplish this, a panel of experts would need to be assembled to select key California indicator species (e.g., fish) and determine the relevance of literature effects data to these species. Table A-7 lists some of the fish species that could be selected as indicator species for response to low DO concentrations. Indicator species should be selected such that a range of responses to low DO are represented. This response range will aid in setting thresholds for habitats where those fish are currently or historically present. Thus endpoints can be generated for DO by extrapolating preliminary DO thresholds for California species using organisms within the same genus or family for which data currently exist. Any data gaps that would need to be addressed could proceed while interim criteria are established. Thus development of effects-based endpoints for DO is possible over the short-term (~2 years).

Table A-5. Dissolved Oxygen thresholds for protection of pelagic species adapted from USEPA 2003.

Criteria Components	Species	Concentration	Duration	Source
Protection Against Growth Effects	resident tidal and freshwater species	> 4.8 mg/L	-	USEPA (2000)
	striped bass	> 3 to 4 mg/L	-	Brandt et al. (1998)
	juvenile striped bass	> 4 mg/L	-	Kramer (1987); Breitburg et al. (1994)
	chinook salmon	>4 mg/L	-	Geist et al. (2006)
	juvenile spot and Atlantic menhaden	>1.5 mg/L	2 weeks	McNatt and Rice (2004)
Egg/Larval Recruitment Effects	resident tidal and freshwater species	>4.6 mg/L; >3.4-3.5 mg/L; >2.7-2.8 mg/L	30 - 40 days 7 days instantaneous minimum	USEPA (2000)
	estuarine species	3 mg/L 1.7 mg/L	30 days instantaneous minimum	Chesney and Houde (1989); Breitburg (1994); USEPA (2000)
	naked goby males abandon nests	0.36 mg/L	-	Breitberg (1992)
Survival	juvenile/adult fish species	>2.3 mg/L	24 hours	USEPA (2000)
	resident tidal and freshwater species	>5.5 mg/L >4 mg/L >3 mg/L	30 days 7 days instantaneous minimum	USEPA (2000)
	striped bass	>5 mg/L	72 hours	Krouse (1968)
	striped bass (preferred concentrations)	>6 mg/L	-	Krouse (1968); Hawkins (1979); Christie et al. (1981); Rothschild (1990)
	juvenile striped bass (preferred concentrations)	> 5 mg/L	-	Kramer (1987); Breitburg et al. (1994)

Criteria Components	Species	Concentration	Duration	Source
Reduced Survival	copepods	<0.86-1.3 mg/L	24 hours	Stalder and Marcus (1997);
	striped bass	> 3 mg/L	72 hours	Krouse (1968)
Protection of Early Life Stages	resident tidal and freshwater species	>6 mg/L >5 mg/L	7-day mean instantaneous minimum	USEPA (1986)
	striped bass	>5 mg/L	-	Krouse (1968)
Protection Against Effects on Threatened/ Endangered Species	shortnose sturgeon	>5 mg/L >3.5 mg/L >3.2 mg/L >4.3 mg/L	30 days 6 hours 2 hours 2 hours	Secor and Niklitschek (2004); Niklitschek (2001); Secor and Gunderson (1998); Jenkins et al. (1993), Campbell and Goodman (2004)
	steelhead, coho, and chinook salmon	>5 mg/L	--	Campbell et al. (2001)
Effects on Total Fish Catch	total fish biomass	< 3.7 mg/L	-	Simpson (1995)
	total fish species richness	< 3.5 mg/L	-	Simpson (1995)
	decline in abundance and diversity of catch	< 2 mg/L	-	Howell and Simpson (1994)
50% Mortality	hogchoaker, northern sea robin	0.5-1 mg/L	24 hours	Reviewed in Breitburg et al. (2001)
	tautog, windowpane flounder adults	>1 mg/L	24 hours	Reviewed in Breitburg et al. (2001); Pihl et al. (1992)
	summer flounder, pipefish, striped bass adults	1.1-1.6 mg/L	24 hours	Reviewed in Breitburg et al. (2001); Pihl et al. (1992); Poucher and Coiro (1997); USEPA (2000)
	skilletfish, naked goby, silverside larvae	1-1.5 mg/L	24 hours	Breitburg (1994); Poucher and Coiro (1997)
	red drum, bay anchovy, striped blenny larvae	1.8-2.5 mg/L	24 hours	Saksena and Joseph (1972); Breitburg (1994); Poucher and Coiro (1997)
	Acartina tonsa and *Eurytemora affinis*	0.36-1.4 mg/L	2 hours	Vargo and Sastry (1977)
	naked goby larvae	>~1.5 mg/L	24 hours	Brietberg (1994)
	sea nettle	>0.7 mg/L	96 hour	Brietberg et al. (1997)
	juvenile stripped bass	>2 mg/L	24 hours	Coutant (1985)
	menhaden	0.70 mg/L 0.88 mg/L 1.04 mg/L	2 hours 24 hours 96 hours	Burton et al. (1980)
	spot	0.49 mg/L 0.67 mg/L 0.70 mg/L	2 hours 24 hours 96 hours	Burton et al. (1980)
100% Mortality	*Acartina tonsa* (copepods)	<1.43 mg/L	24 hours	Stalder and Marcus (1997)
	Acartina tonsa and *Oithona colcarva*	<2 mg/L	24 hours	Roman et al. (1993)
	copepods	<0.71 mg/L	24 hours	Stalder and Marcus (1997)
	naked goby males	<0.24 mg/L	-	Breitberg (1992)
	juvenile striped bass	< 3 mg/L	-	Chittenden (1972); Coutant (1985); Krouse (1968)
	rainbow trout	< 3.1	-	Matthews and Berg (1997)
Food web shift from predation on fish larvae (naked gobi) by juvenile and adult fish (striped bass) to invertebrates (sea nettles)		2.5-3.3	-	Brietberg et al. (1997)
Reduced Activity (swimming)	naked goby	1.5 mg/L	-	Brietberg et al. (1997)
	stripped bass	<3 mg/L	-	Breitberg et al. (1994)
Avoidance Threshold/	weakfish, blue crab, shrimp, croaker and spot	2.1-2.3 mg/L	-	Eby and Crowder (2002)

Criteria Components	Species	Concentration	Duration	Source
Emigration	menhaden and southern flounder	2.6 mg/L	-	Eby and Crowder (2002)
	pinfish and silver perch	4.1-4.2 mg/L	-	Eby and Crowder (2002)
	zooplankton	< 1 mg/L	-	Roman et al. (1993)
	Gulf of Mexico demersal fish	<2 mg/L	-	Craig et al. (2001)
	logfish	2.0 mg/L	-	Beitinger & Pettit (1984)
	cod and whiting	25-40% saturation	-	Gray 1992
	benthic flatfishes	40-50% saturation	-	Gray 1992

Table A-6. Benthic organism response to bottom water dissolved oxygen concentrations adaptedfrom USEPA 2003.

Response	Dissolved Oxygen	Species	Reference
Avoidance			
Infaunal Swimming	1.1	*Paraprionospio pinnata*	Diaz et al. (1992)
	0.5	Nereis succinea	Sagasti et al. (2000)
Epifaunal Off Bottom	0.5	*Neopanope sayi*	Sagasti et al. (2000)
	0.5	*Callinectes sapidus*	Sagasti et al. (2000)
	1	*Stylochus ellipticus*	Sagasti et al. (2000)
	1	*Stylochus ellipticus*	Sagasti et al. (2000)
	0.5	*Dirodella obscura*	Sagasti et al. (2000)
	1	*Cratena kaoruae*	Sagasti et al. (2000)
Fauna, Unable to Leave or Escape, Initiate a Series of Sublethal Responses			
Cessation of Feeding	0.5	*Balanus improvisus*	Sagasti et al. (2000)
	0.6	*Streblospio benedicti*	Llanso (1991)
	1	*Loimia medusa*	Llanso and Diaz (1994)
	1.1	*Capitella sp.*	Forbes and Lopez (1990)
Decreased Activities Not Related to Respiration	0.5	*Balanus improvisus*	Sagasti et al. (2000)
	0.5	*Conopeum tenuissimum*	Sagasti et al. (2000)
	0.5	*Membranipora tenuis*	Sagasti et al. (2000)
	1	*Cratena kaoruae*	Sagasti et al. (2000)
	1	*Stylochus ellipticus*	Sagasti et al. (2000)
	1	*Streblospio benedicti*	Llanso (1991)
Cessation of Burrowing	1.1	*Capitella sp.*	Warren 1977
Emergence from Tubes or Burrows	0.1-1.3	*Ceriathiopis americanus*	Diaz unpublished data
	0.5	*Sabellaria vulgaris*	Sagasti et al. (2000)
	0.5	*Polydora cornuta*	Sagasti et al. (2000)
	0.7	*Micropholis atra*	Diaz et al. (1992)
	1	*Hydroides dianthus*	Sagasti et al. (2000)
	10% saturation	*Nereis diversicolor*	Vismann (1990)
Siphon Stretching into Water Column	0.1-1.0	*Mya arenaria, Abra alba*	Jorgensen (1980)
Siphon or Body Stretching	0.5	*Molgula manhattensis*	Sagasti et al. (2000)
	0.5	*Diadumene leucolena*	Sagasti et al. (2000)
Floating on Surface of Water	0.5	*Diadumene leucolena*	Sagasti et al. (2000)
Formation of Resting Stage	0.5	*Membranipora tenuis*	Sagasti et al. (2000)
	0.5	*Conopeum tenuissimum*	Sagasti et al. (2000)

Table A-7. Fish native to California estuaries that could serve as indicator species for hypoxia.

Type	Species
Small Fish (less able to migrate out of low dissolved oxygen areas)	Intertidal: arrow goby, longjaw mudsucker, tidewater goby (endangered) Subtidal: cheekspot goby, shadow goby, Pacific staghorn sculpins
Poor Swimmers (in SAV or lgae habitat)	Snubnosed pipefish, banded pipefish, bay pipefish, California killifish, California halibut, diamond turbot
Good Swimmers (able to migrate out of low dissolved oxygen areas)	Surf perches, shiner perch, yellowfin croaker, spotfin croaker, topsmelt, jacksmelt, striped mullet, deepbody anchovy, slough anchovy, spotted sand bass, white seabass
Juveniles (in lagoons only)	Barred sand bass, kelp bass
Sharks and Rays	Leopard shark, round stingray, bat rays
Recreational/Commercial Fish	White seabass, California halibut, sand bass, jacksmelt, yelowfin croaker, striped mullet, spotfin croaker
Anadromous Fish/Endangered Species	Steelhead trout

The in order to implement an effect-based approach, credible scientific evidence must exist to support the selection of thresholds. For other biological response variables (macroalgal biomass, water clarity, aesthetics, harmful algal blooms, chlorophyll a, and SAV biomass) much less effects-based literature is available from which thresholds can be derived. Thus it would require additional time (3 - 5 years) and funding to conduct the research required to implement this approach.

Endpoints derived from the percentile approach are not directly linked to beneficial uses. For this reason, the percentile approach is not the preferred method for criteria development. For example, in this study the percentile approach produced a threshold of 3.7 mg/L. This number could be completely different depending on the length of the dataset, source of the data, the location in which the data are taken. In this particular case, the data are clearly not from a representative sample of estuaries; in fact it is highly advisable, in absence of pristine reference sites for the region or state, that ambient survey data from a probability-based survey be used if this approach is to be seriously considered. Adequate data do not currently exist and would have to be developed to support this approach for numeric endpoint development in California estuaries. We were only able to obtain data sets for four lagoons in the perennially tidal coastal lagoon class. Continuous monitoring data for these lagoons is extensive for DO, salinity, conductivity, temperature, water depth, and turbidity. However, there were no continuous monitoring programs for macroalgal biomass, SAV habitat, benthic invertebrates, or harmful algal blooms. Thus, our picture of the eutrophication status for these four lagoons is incomplete. Monitoring should include nutrient loading, DO, algal biomass, and SAV cover at a minimum. Such a data set could be used to develop dynamic ecosystem models for eutrophication in each lagoon subclass and establish links among specific biological response variables and nutrient loading.

Efforts to create a multi-metric approach to setting numeric endpoints in California will be hampered unless these data gaps are addressed. Section 5 of this document lays out an implementation plan intended to address data gaps and move forward with the process of developing NNEs and TMDL tools for California estuaries.

Selection of DO monitoring location within the estuarine system should also be a consideration, as different locations within estuaries will have very different responses to nutrient over-enrichment. Local topography and water column structure will have an impact on the frequency and duration of low DO events (compare probability of exceedence at different sites in the same estuary in Table A-4). Some locations may never suffer low DO events because they are so well mixed (Turner et al. 1987, Borsuk et al. 2001). Thus, it is critical to select monitoring locations within estuaries that are representative of conditions within the entire system. Massachusetts Estuarine Project (Howes et al. 2003) outlined the importance of selecting a sentinel location at which monitoring efforts could be focused. The sentinel location is selected such that the restoration of that one site will necessarily bring the other regions of the system to acceptable habitat quality levels.

Anthropogenic nutrient enrichment does not affect all classes of estuaries equally. For estuaries with similar nutrient loads, some estuaries will exhibit classic symptoms of eutrophication, algal blooms and low DO, while others maintain low algal biomass and primary production (Cloern 1999). This variability in nutrient loading response results from differences among the physical structure of estuarine systems. Thus, it is important that numeric endpoints be developed that are specific to particular estuarine classes is critical so that remediation efforts can be focused on those systems that are most sensitive to eutrophication and beneficial use impairment.

REFERENCES

Beitinger, T. L., and M. J. Pettit. 1984. Comparision of low oxygen avoidance in a bimodal breather, *Erpetoichthys calabaricua* and an obligated water breather, *Percina caprodes*. *Environmental Biology of Fishes* 11:235-240.

Borsuk, M.E., C.A. Stow, J.R.A. Luettich, H.W. Paerl and J.L. Pinckney. 2001. Modelling oxygen dynamics in an intermittently stratified sstuary: Estimation of process rates using field data. *Esturine, Coastal and Shelf Science* 52:33-49.

Brandt, S.B., E. Demers, J.A. Tyler and M.A. Gerken. 1998. Fish Bioenergetics Modeling: Chesapeake Bay Ecosystem Modeling Program (1993-1998), Report to the Chesapeake Bay Program. US Environmental Protection Agency, Chesapeake Bay Program Office. Annapolis, MD.

Breitburg, D.L. 1992. Episodic hypoxia in Chesapeake Bay: Interacting effects of recruitment, behavior and physical disturbance. *Ecological Monographs* 62:525-546.

Breitburg, D.L. 1994. Behavioral response of fish larvae to low dissolved oxygen concentrations in a stratified water column. *Marine Biology* 120:615-625.

Breitburg, D.L., T. Loher, C.A. Pacey and A. Gerstein. 1997. Varying effects of low dissolved oxygen on trophic interactions in an estuarine food web. *Ecological Monographs* 67:489-507.

Breitburg, D.L., L. Pihl and S.E. Kolesar. 2001. Effects of low dissolved oxygen on the behavior, ecology and harvest of fishes: A comparison of the Chesapeake Bay and Baltic-Kattegat systems. pp. 241-267.*in*: N.N. Rabelais and R.E. Turner, editors, Coastal Hypoxia: Consequences for living resoures and ecosystmes. American Geophysical Union. Washington, DC.

Breitburg, D.L., N. Steinberg, S. DuBeau, C.Cooksey and E.D. Houde. 1994. Effects of low dissolved oxygen on predation on estuarine fish larvae. *Marine Ecology Progress Series* 104:235-246.

Burton, D.T., L.B. Richardson and C.J. Moore. 1980. Effect of oxygen reduction rate and constant low dissolved oxygen concentrations on two estuarine fish. *Transactions of the American Fisheries Society* 109:552-557.

Campbell, S.G., R.B. Hanna, M. Flug and J.F. Scott. 2001. Modeling Klamath River system operations for quantity and quality. *Journal of Water Resources Planning and Management* 127:284-294.

Chesney, E.J. and E.D. Houde. 1989. Laboratory studies on the effect of hypoxic waters on the survival of eggs and yolk-sac larvae of the bay anchovy, Anchova mitchilli. pp. 184-191 *in*: E. D. Houde, E.J. Chesney, T.A. Newberger, A.V. Vazquez, C.E. Zastrow, L.G. Morin, H.R. Harvey and J.W. Gooch, editors, Population Biology of Bay Anchovy in Mid-Chesapeake Bay- Final Report to Maryland Sea Grant. R/F-56, UMCEES Ref. No. CBL 89-141.

Christie, R.W., P.T. Walker, A.G. Eversole and T.A. Curtis. 1981. Distribution of spawning blueback herring on the West Branch of Cooper River and Aantee River, South Carolina. Proceeding of the Annual Conference of the Southeastern Association of Fisheries and Wildlife Agencies 35:632-640.

Cloern, J.E. 1999. The relative importance of light and nutrient limitation of phytoplankton growth: a simple index of coastal ecosystem sensitivity to nutrient enrichment. *Aquatic Ecology* 33:3-16.

Coutant, C.C. 1985. Striped bass, temperature, and dissolved oxygen: A speculative hypothesis for environmental risk. *Transactions of the American Fisheries Society* 114:31-61.

Craig, J.K., C.D. Gray, C.M. McDaniel, T.L. Henwood and J.G. Hanifen. 2001. Ecological effects of hypoxia on fish, seaturtles, and marine mammals in the northwestern Gulf of Mexico. pp. 269-291 *in*: N.N. Rabalais and R.E. Turner, editors, Coastal Hypoxia: Consequenes for Living Resources and Ecosystems. American Geophysical Union, Washington, D.C.

Diaz, R.J., R.J. Neubauer, L.C. Schaffner, L. Phil and S.P. Baden. 1992. Continous monitoring of dissolved oxygen in an estuary experience periodic hypoxia and the effects of hypoxia on macrobenthos and fish. *Science of the Total Environment* supplement.

Eby, L.A. and L.B. Crowder. 2002. Hypoxia-based habitat compression in the Neuse River Estuary: context-dependent shifts in behavioral avoidance thresholds. *Canadian Journal of Fisheries and Aquatic Sciences* 59:952-965.

Forbes, T.L. and G.R. Lopez. 1990. The effect of food concentration, body size and environmental oxygen tension on the growth of the deposit deding polychaete, Capitella species. *Limnology and Oceanography* 35:1535-1544.

Geist, D.R., C.S. Abernethy, K.D. Hand, V.I. Cullinan, J.A. Chandler and P.A. Groves. 2006. Survival, development, and growth of fall Chinook salmon embryos, alevins, and fry exposed to variable thermal and dissolved oxygen regimes. *Transactions of the American Fisheries Society* 135:1462-1477.

Gray, J. S. 1992. Eutrophication in the sea. Pages 3-15 *in* G. Columbo, I. Ferrari, V. U. Ceccherelli, and R. R., editors. Marine Eutrophication and Population Dynamics. Olsen & Olsen, Fredensborg.

Hawkins, J.N. 1979. Anadromous Fisheries Research Program: Neuse River. North Carolina Deparment of Natural Resources and Community Develoment, Division of Marine Fisheries. Morehead City, NC.

Howell, P. and D. Simpson. 1994. Abundance of marine resources in relation to dissolved oxygen in Long Island Sound. *Estuaries* 17:394-402.

Howes, B.L., R. Samimy and B. Dudley. 2003. Massachusetts Estuaries Project Site Specific Nitrogen Thresholds for Southeastern Massachusetts Embayments: Critical Indicators. Massachusetts Department of Environmental Protection. Boston, MA.

Jenkins, W.E., T.I.J. Smith, L.D. Heyward and D.M. Knott. 1993. Tolerance of shortnose sturgon, Acipenser brevirostrum, juveniles to different salinity and dissolved oxygen concentrations. Proceedings of the Annual Conference of Sotheastern Association of Fish and Wildlife Agencies 47:476-484.

Jorgensen, B.B. 1980. Seasonal oxygen depletion in the bottom waters of a Danish Fjod and its effects on the benthic community. *Okios* 34:68-76.

Kramer, D.L. 1987. Dissolved oxygen and fish behavior. Environmental Biology of Fishes 18.

Krouse, D.L. 1968. Effects of dissolved oxygen, temperature and salinity on survival of young striped bass, *Roccus saxatilis*. University of Maine. Orono, ME.

Llanso, R.J. 1991. Tolerance of low dissolved oxygen and hydrogen sulfide by the polychaete Strepblospio benedicti. *Journal of Experimental Marine Biology and Ecology* 153:165-178.

Llanso, R.J. and R.J. Diaz. 1994. Tolerance to dissolved oxygen by the tubicolous polychaete *Loimia medusa*. *Journal of the Marine Biological Association of the United Kingdom* 74:143-148.

Matthews, K.R. and N.H. Berg. 1997. Rainbow trout responses to water temperature and dissolved oxygen stress in two southern California stream pools. *Journal of Fish Biology* 50:50-67.

McNatt, R.A. and J.A. Rice. 2004. Hypoxia-induced growth rate reduction in two juvenile estuary-dependent fishes. *Journal of Experimental Marine Biology and Ecology* 311:147-156.

Pihl, L., S.P. Baden, R.J. Diaz and L.C. Schaffer. 1992. Hypoxia-induced structural changes in the diet of bottom-feeding fish and crustacea. *Marine Biology* 112:349-361.

Poucher, S. and L. Coiro. 1997. Effects of low dissolved oxygen on saltwater animals: Morandum to D.C. Miller *in* A.E.D. US Environmental Protection Agency. Narragansett, RI.

Roman, M., A.L. Gauzens, W.K. Rhinehart and J.R. White. 1993. Effects of low dissolved oxygen water on Chesapeake Bay zooplankton. *Limnology and Oceanography* 38:1603-1614.

Rothschild, B.J. 1990. Development of a sampling expert system: "FISHMAP". Marlyand Department of Natural Resources and US Fish and Wildlife Service Project No. F171-89-008. University of Maryland CEES Ref. No. (UMCEES) CBL 90-090. Chesapeake Biological Laboratory. Solomons, MD.

Sagasti, A., L.C. Schaffner and J. E. Duffy. 2000. Epifaunal communites thrive in an estuary with hypoxic episodes. *Estuaries* 23:474-487.

Saksena, V.P. and E.B. Joseph. 1972. Dissolved oxygen requirements of newly-hatched larvae of the striped blenny (*Chasmodes bosquianus*) the naked goby (*Gobiosoma bosci*) and the skillet fish (*Gobiesox strumosus*). *Chesapeake Science* 13:23-28.

Secor, D.H. and T.E. Gunderson. 1998. Effects of hypoxia and temperature on survival growth and respiration of juvenile Atlantic Sturgeon, *Acipenser oxyrinchus*. *Fisheries Bulletin* 96:603-613.

Secor, D.H. and E.J. Niklitschek. 2004. Sensitivity of sturgeons to environmental hypoxia: Physiological and ecological evidence, *in:* Fish physiology, Toxicology and Water Quality-- Proceedings of the Sixth International Symposium. La Paz, Mexico.

Simpson, D.G. 1995. Cooperative Interagency Resource Assessment. A Study of Marine Recreational Fisheries in Connecticut. Federal Aid to Sport Fish Recreation, F54R. Connecticut Department of Environmental Protection, Bureau of Natural Resources, Fisheries Division.

Stalder, L.C. and N.H. Marcus. 1997. Zooplankto responses to hypoxia: Behavioral patterns and survival of three species of calanoid copepods. *Marine Biology* 127:599-607.

Turner, R.E., N. Qureshi, N.N. Rabalais, Q. Dortch, D. Justic, R.F. Shaw and J. Cope. 1998. Fluctuating silicate: nitrate ratios and coastal plankton food webs. *Proceedings of the National Academy of Sciences* 95:13,048-013.

Turner, R.E., W.W. Schroeder and J.W.J. Wiseman. 1987. The role of stratification in the deoxygenation of mobile bay and adjacent shelf bottom Waters. *Estuaries* 10:13-19.

United States Environmental Protection Agency (USEPA). 1986. Ambient Water Quality Criteria for Dissolved Oxygen (Freshwater). U.A. Environmental Protection Agency, Washinton D.C.

USEPA. 2000. Ambient Aquatic Life Water Quality Criteria for Dissolved Oxygen (Saltwater): Cape Cod to Cape Hatteras. U.S. Environmental Protection Agency, Washington D.C.

USEPA. 2001. Nutrient Criteria Technical Guidance Manual: Estuarine and Coastal Marine Waters. U.S. Environmental Protection Agency, Washington D.C.

USEPA. 2003. Ambient Water Quality Criteria for Dissolved Oxygen, Water Clarity, and Chlorophyll a for Chesapeake Bay and Its Tidal Tributaries. U.S. Environmental Protection Agency, Washington D.C.

Vargo, S.L. and A.N. Sastry. 1977. Interspecific differences in toleranceof *Eurytemora affinis* and *Acartia tonsa* from an estuarine anoxic vasin to low dissolved oxygen and hydrogen sulfide. pp. 219-226 *in*: D.S. McCluskey and A.J. Berry, editors, Physiology and Behavior of Marine Organisms. 12th European Marine Biology Symposium. Pergamon Press. Oxford, U.K.

Vismann, B. 1990. Sulfide detoxification and tolerance in Nereis (dediste) diversicolor and Nereis (neanthes) virens (*Annelida: Polychaeta*). *Marine Ecology Progress Series* 59:229-238.