TECHNICAL APPROACH TO DEVELOP NUTRIENT NUMERIC ENDPOINTS FOR CALIFORNIA ESTUARIES

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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 and kelp beds, 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. Without management actions to reduce anthropogenic nutrient loads, symptoms are expected to develop or worsen in the majority of systems, due to projected population increases in coastal areas.

The purpose for this report is to outline a conceptual framework for the development of nutrient numeric endpoints (NNE) for estuaries and to highlight data gaps and research recommendations critical for their development. The purpose of NNEs for California estuaries is to provide 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. The objective is to control excess nutrient loads to levels such that the risk of impairing the designated uses is minimized. If the nutrients present–regardless of actual magnitude–have a low probability of impairing uses, then water quality standards can be considered to be met.

The ultimate goal of this effort is to develop a set of tools that can be used to support the water quality programs of the California State Water Resources Control Board (SWRCB), Regional Water Quality Control Boards and the regulated community. To reach this goal, we envision the development of a set of estuarine NNE tools, including 1) a classification scheme that groups estuaries according to factors that control their biological response to nutrient loading, 2) risk-based indicators of biological response that can provide quantitative measures of the status of beneficial uses relative to nutrient loads; 3) thresholds that define beneficial use risk categories (BURCs), which provides a framework for regulatory decisions based on quantitative assessments of impairment; and 4) modeling tools that link biological response indicators to watershed nutrient loads. To develop a NNE toolkit for estuaries, the first step in this process was to provide a working framework and to identify key data gaps and research recommendations critical for NNE development.

The NNE conceptual framework for estuaries is based on previous work by the SWRCB and US EPA Region IX, which provided guidance for development of NNE in streams and lakes (TetraTech 2005). This framework is founded on the concept that biological response indicators are better suited to evaluate the risk of beneficial use impairment, rather than using pre-defined nutrient limits that may or may not result in mitigation of eutrophication for a particular water body. The advantage of the proposed approach is a more robust link to actual impairment of use, rather than an approach that relies on concentration data alone.

The California NNE framework for estuaries is based on three organizing principals:

- *Biological response indicators provide a more direct risk-based linkage to beneficial uses than nutrient concentrations alone.*
- *A weight of evidence approach with multiple indicators will produce NNE with greater scientific validity.*

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

There is no clear scientific consensus on a target thresholds associated with impairment for many of the biological indicators of eutrophication. To address this problem, the California NNE framework classifies water bodies into the three Beneficial Use Risk Categories (BURC; TetraTech 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: In these waterbodies, 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, the BURC I/II boundary represents a level below which there is general consensus that nutrients will not present a significant risk of impairment. The BURC II/III boundary represents a concentration or load that is sufficiently high that there is consensus 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.

Ultimately, the goal is to propose preliminary NNE targets (i.e. BURC thresholds) for each of the biological response indicators using literature sources, monitoring data, and expert opinion. These values may change from 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 estuaries. Depending on the use, data availability, and economic impact of the decision, other, more detailed and site-specific tools may be appropriate for translating secondary indicator targets to nutrient loading targets.

The creation of a toolkit to support development of NNE can be approached through a set of four discrete steps, each with an inherent set of data requirements for its successful completion:

1. Develop definition and classification scheme
2. Select biological response variables
3. Develop numeric nutrient endpoints
4. Create TMDL tools

There are several data gaps and steps that need to be addressed before thresholds for Beneficial Use Risk Categories for secondary indicators can be established. Below, a list is given of the highest priority data gaps, uncertainties, and other technical and policy issues that were identified during the course of this project. The list is provided for consideration to EPA Region IX, the State Water Resources Control Board and the Regional Water Quality Control Boards with estuarine waters to develop a plan for the next steps forward for the development of nutrient numeric endpoints for California estuaries.

- Adopt, for the purposes of nutrient criteria development, a uniform definition of “estuary” across all regional boards; this definition should be one that easily lends itself to mapping of estuarine classes and the freshwater and oceanic boundaries.
- Generate a comprehensive list of estuaries, using the “uniform” definition of estuary across all regional boards and perform statistical analysis to confirm appropriate classification of each estuary and determine whether ecoregions must be considered for this classification.
- Develop conceptual models of nutrient cycling for each estuarine class, including the sources, sinks, mechanisms for transformation, and links with biological response.
- Collect continuous data sets (2-5 yrs) of nutrient loading and selected biological response indicators (DO, SAV, macroalgae, phytoplankton etc.) in several index systems representing a range of eutrophication for each of the estuarine classes. These data would: 1) assist in defining the “critical condition” for indicator measurement, 2) assist in determination of numeric endpoints by providing a range of reference conditions, and 3) provide a dataset to explore the development of load-response models.
- Conduct research to clarify the relationship between biomass of primary producer communities, sediment oxygen demand, and surface water DO.
- Evaluate the impacts of macroalgal blooms on benthic macroinvertebrates and investigate to what extent any impact may affect food availability to fish and birds
- Investigate mechanisms controlling the production of toxins in harmful algal blooms.
- Investigate the environmental factors that promote toxic harmful algal blooms. This includes: 1) the relative importance of anthropogenic versus natural sources of nutrients (upwelling), 2) the importance of atmospheric deposition and 3) what physical factors (upwelling, river discharge, etc.) create conditions suitable for HAB formation.
- Conduct historical studies that 1) help to establish a range of values of the biological response indicators at a time period when an estuary was unimpacted, and 2) establish connections between historical land use, nutrient loads, and indicators of biological response.
- Explore the developing of regression models of load and response for estuarine classes with existing data. Once established, validate regression models with additional monitoring in index systems. For those classes where adequate data do not exist, collect continuous data on nutrient loads, DO, SAV, macroalgae, phytoplankton and HABs (see above).
- Establish an internet-based clearinghouse for applicable conceptual models, watershed loading and estuarine water quality models, and supporting studies by estuarine class.
- Conduct a literature review to identify ranges in rates for key biogeochemical processes (nitrification and denitrification, benthic nitrogen fixation, sediment nutrient flux, primary producer uptake, storage and transformation of nutrients, etc.) for each estuarine class and identify key data gaps; conduct studies to address data gaps, including studies that establish how rates vary along an eutrophication gradient for each estuarine class.
- Conduct studies to characterize the relative importance of nutrient sources that are typically under-characterized, such as atmospheric deposition or groundwater inputs.
- Develop watershed loading and estuarine water quality models in open source code, such that the modeling approaches can be improved over time by collaboration and data sharing

ACKNOWLEDGEMENTS

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1 NUTRIENT NUMERIC ENDPOINTS FOR CALIFORNIA ESTUARIES: AN INTRODUCTION

1.1 PROBLEM STATEMENT

Cultural eutrophication of estuaries and coastal waters is a global 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), degradation of coral reefs, loss or degradation of seagrass and kelp beds (McGlathery, 2001; Twilley, 1985; Burkholder et al., 1992), 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 (Trainer et al. 2002, Bates et al. 1989a or b). These modifications have significant economic and social costs, some of which can be readily identified and valued, while others are difficult to determine (Turner et al., 1998). According to EPA, eutrophication is one of the top three leading causes of impairments of the nation's waters (along with sedimentation and pathogens, US EPA 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 1996). NOAA's 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 (Figure 1; Bricker, et al., 1999). These estuaries tend to have restricted circulation and high nutrient inputs. In the estuaries of San Francisco Bay, impacts from excess nutrient loads have been attenuated from turbidity due to high sediment loading. However, trends suggest that with declining sediment loads effects of high nutrient concentrations may gradually become more evident (Cloern et al. 1983, Cloern 1982). Without management actions to reduce anthropogenic nutrient loads, symptoms are expected to develop or worsen in the majority of systems, primarily due to projected population increases along the coastal areas.

1.2 PURPOSE OF THIS DOCUMENT

The purpose for this report is to outline a conceptual framework for the development of nutrient numeric endpoints (NNE) for estuaries and to highlight data gaps and research recommendations critical for their development. The ultimate goal of this effort is to develop a set of tools that can be used to support the water quality programs of the California State Water Resources Control Board, Regional Water Quality Control Boards and the regulated community. These programs include setting numeric limits for NPDES permits; developing targets for total maximum daily load allocations (TMDL); and for those Regional Water Quality Control Boards that choose to, developing numeric nutrient criteria.

To reach this goal, we envision the development of an estuarine NNE toolkit. These tools would include: 1) estuary classification categories that group estuaries with common hydrogeomorphology that control their biological response to nutrient loading, 2) risk-based indicators of estuarine biological response that can provide quantitative measures of the status of beneficial uses within the estuaries relative to nutrient loads; 3) beneficial use risk categories (BURCs) that provide a framework for regulatory decisions based on quantitative assessments of impairment; and 4) modeling tools that link biological response indicators to watershed nutrient loads. These modeling tools could be used to support the determination of TMDL allocations of nutrients from watersheds.

This report provides the starting point for creation of this toolkit. Key stakeholders at the local, state, and federal level should review the framework and research needs highlighted in the report before development of the toolkit can continue.

The report is organized in five sections:

- Introduction and purpose of document,
- Proposed approach
- A brief literature review of conceptual models and indicators
- Process for establishing NNEs – data gaps and research recommendations
- Summary findings and research recommendations.

1.3 BACKGROUND FOR DEVELOPMENT OF NUTRIENT CRITERIA IN ESTUARIES

The NNE conceptual framework is based on previous work by the California State Water Resources Control Board and US EPA Region IX, which provided guidance for development of NNE in streams and lakes (TetraTech 2005). This framework is founded on the concept that biological response indicators are better suited to evaluate the risk of beneficial use impairment, rather than using pre-defined nutrient limits that may or may not result in mitigation of eutrophication for a particular water body. This approach deviates from previous national guidance on nutrient criteria development. The background on the evolution of the California NNE framework is presented here.

The process to develop nutrient criteria began with the publication of the National Strategy for the Development of Regional Nutrient Criteria (USEPA, 1998). National criteria recommendations were developed, using a statistical approach to establish thresholds based on the nutrient concentrations in surface waters. Data sets from Legacy STORET, NASQAN, NAWQA, and EPA Region 10 were used to assess nutrient conditions from 1990 to 1998, aggregating by Level III ecoregions. Reference conditions were developed 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). EPA also noted that States and Tribes may "need to identify with greater precision the nutrient levels that protect aquatic life and recreational uses." EPA also encouraged that States and Tribes "critically evaluate this information in light of the specific designated uses that need to be protected." Following this, EPA issued a series of technical guidance manuals for the development of nutrient criteria, including that for estuaries and coastal waters (US EPA 2001).

Several studies have since demonstrated the shortcomings of using ambient nutrient concentrations alone to predict eutrophication, particularly in streams (Heiskary and Markus, 2001; Welch et al., 1989; Chételat, et al., 1999; Dodds et al., 2002; Fevold, 1998) and estuaries (Cloern, 2001; Dettmann, 2001, and Kennison et al. 2003). Use of surface water nutrient concentrations is generally not effective for assessing eutrophication and the subsequent impact on water use because biological response (e.g. algal productivity) depends on several contributing factors such as morphology, light availability, biological community structure, etc. Thus high concentrations are not an obligatory indicator of eutrophication, and low concentrations do not necessarily indicate absence of eutrophication.

In 1999, the US EPA Region IX launched a program to establish nutrient criteria for California. As work on this program has evolved, the California NNE framework emerged. The California NNE framework relies on using selected biological response indicators in addition to nutrient concentrations. Although these biological responses are not always easily measured and are more difficult to predict than concentrations, these indicators appear to be more directly linked to risks of impairing beneficial uses than are nutrient concentrations. Despite the additional data requirement, the advantage of the proposed approach is a more robust link to actual impairment of use, rather than an approach that relies on concentration data alone.

To support the development of nutrient criteria, EPA Region IX convened the Regional Technical Advisory Group (RTAG) and the State Technical Advisory Group (STRTAG) to serve as a forum for collaboration among stakeholders, agencies, and all nine Regional Water Boards (STRTAG). To better support the regulatory priorities of its members, the RTAG and STRTAG focused on the development of nutrient criteria in freshwater systems including streams, rivers, lakes, and reservoirs. The development of nutrient numeric endpoints for fresh waters proceeded prior to estuaries with the caveat that endpoints for streams would consider potential downstream impacts on estuaries. The STRTAG formally adopted the California NNE framework in 2006.

In 2005, US EPA Region IX funded a new initiative to develop nutrient criteria for estuaries and coastal waters. As was done in freshwater systems, the approach incorporates selection of secondary indicators that provide a more direct link to impaired beneficial uses. The first step in this process was to provide a working framework and to identify key data gaps and research recommendations critical for development of NNEs in California estuaries.

As EPA Region IX has been working on the framework for estuarine NNEs, EPA Office of Science and Technology (EPA OST) is currently working on new national guidance, slated for release in 2007. An early product of this guidance is a report by the National Estuaries Expert Workgroup (NEEW) on science critical to develop nutrient criteria for the Nation’s estuaries. The findings of this report are consistent with California NNE framework in that it emphasizes the use of biological response indicators rather than nutrient concentrations alone to manage eutrophication (NEEW, 2007).

2 SETTING NNE IN CALIFORNIA ESTUARIES – A RISK-BASED APPROACH

The purpose of developing numeric nutrient endpoints (NNEs) for California estuaries is to provide 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 excess nutrient loads to levels 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 uses, then water quality standards can be considered to be met.

The basic problem is thus to link specific designated uses to levels of nutrients and other exogenous factors that are likely to impair those uses. The techniques developed for ecological risk assessment (ERA) are highly relevant to establish this connection. This section presents the proposed California approach to develop NNEs for estuaries, defines the designated uses of California estuaries, and describes the conceptual linkage between nutrient loads and risk of use impairment.

2.1 PROPOSED CALIFORNIA APPROACH TO DEVELOP NNES FOR ESTUARIES

The California NNE framework for estuaries is based on three organizing principles:

1. *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, it is biological response to nutrients that impair uses. Instead of setting criteria solely in terms of nutrient concentrations, it is preferable to take into account the risk of impairment of these uses. The NNE framework needs to contain, in addition to nutrient concentrations, targeting information on biological response indicators such as benthic algal biomass, planktonic chlorophyll a, dissolved oxygen, macrophyte cover, and water clarity. These biological response indicators provide a more direct risk-based linkage to beneficial uses than nutrient concentrations alone.

2. *A weight of evidence approach with multiple indicators will produce NNE 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. 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). This “multiple lines of evidence” approach is also being used in the development of the State’s Sediment Quality Objectives

3. *For many of the biological indicators associated with nutrients, no clear 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 is no clear scientific consensus on a target thresholds associated with impairment for many of the biological indicators of eutrophication. To address this problem, the California NNE framework classifies water bodies into the three risk classification categories, referred to as Beneficial Use Risk Categories (BURC) and illustrated in Figure 1-1 (TetraTech 2005). These categories are as follows:

- BURC I water bodies: In these waterbodies, beneficial uses are sustained and are not exhibiting impairment due to nutrients;
- BURC II water bodies: In these waterbodies 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 water bodies: These waterbodies are exhibiting impairment due to nutrients; regulatory action is warranted

For a given beneficial use designation, the BURC I/II boundary represents a level below which there is general consensus that nutrients will not present a significant risk of impairment. This boundary should also be set so that is not less than the expected natural background. Conversely, the BURC II/III boundary represents a concentration or load that is sufficiently high that there is consensus 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. Permitting discharges to waters that remain within BURC II after additional analysis would require some sort of antidegradation or reasonable potential analysis. Ultimately, the goal is to propose preliminary NNE targets (i.e. BURC thresholds) for each of the biological response indicators using literature sources, monitoring data, and expert opinion. These values may change across estuary, classes, and ecoregions within California. We believe this three-tiered approach is superior to a binary meet/does not meet criteria approach.

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 estuaries. Depending on the use, data availability, and economic impact of the decision, other, more detailed and site-specific tools may be appropriate for translating secondary indicator targets to nutrient loading targets. The lessons learned from the experience gained through several years of pilot studies for the development of nutrient criteria suggests that no one approach will be suitable for all the diverse water bodies within California. However, we believe that the proposed California approach will provide solutions to many if not most of the issues that need to be addressed in setting numeric nutrient endpoints for California.

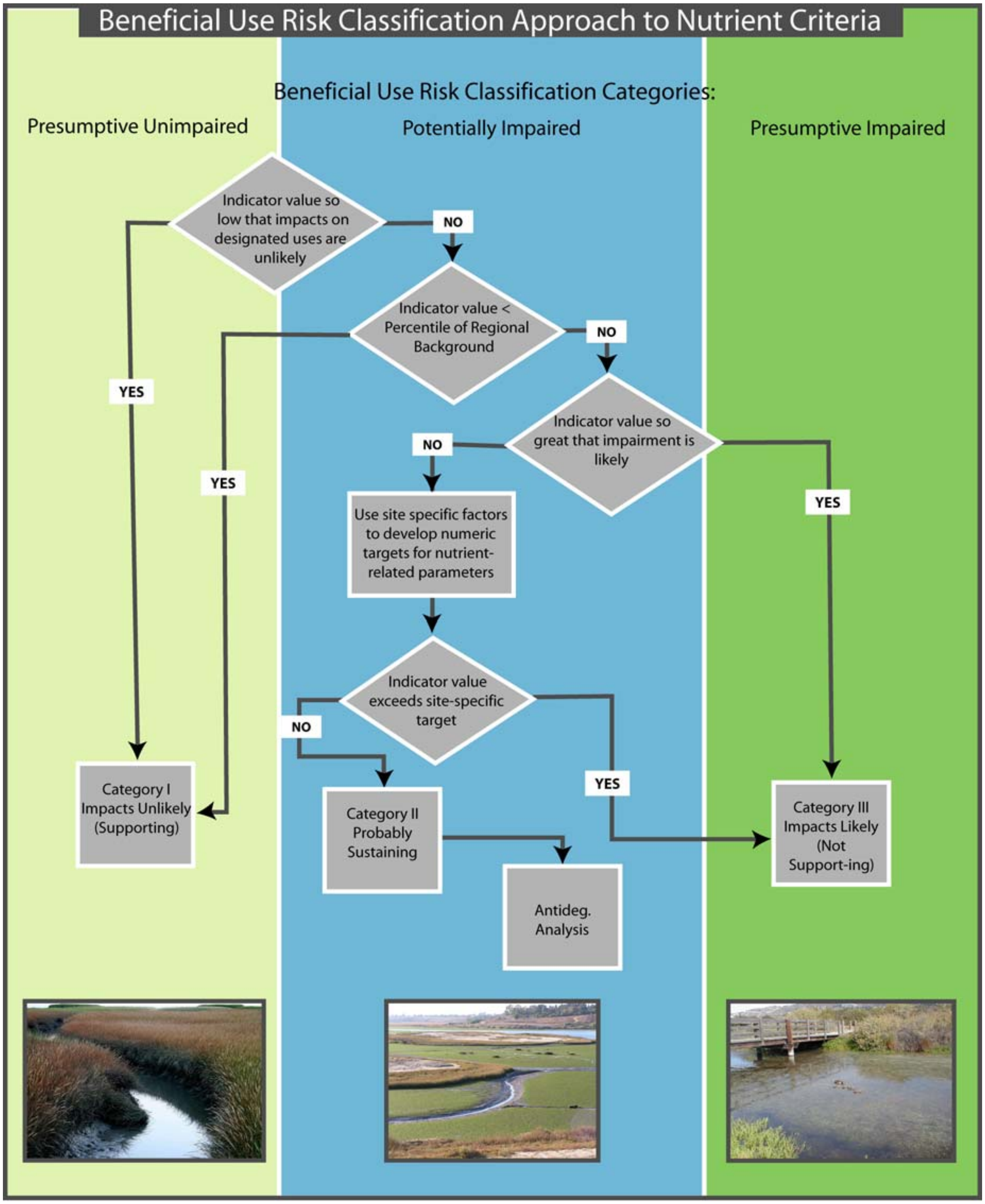


Figure 2.1 Beneficial Use Risk Categories and the nutrient criteria assessment process (adapted from TetraTech (2005).

2.2 DESIGNATED USES FOR ESTUARIES

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. Aquatic ecosystems and underground aquifers provide many different benefits 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. The beneficial uses for California estuaries are illustrated in Figure 2-2 and are described below. The list of designated uses provides a starting point in understanding the relationships between nutrients and use impairment. The following beneficial uses are used throughout California for estuaries. It should be noted that waterbodies are generally assigned multiple beneficial uses.

Areas of Special Biological Significance (ASBS)

ASBS are areas designated by the State Water Resources Control Board. They include marine life refuges, ecological reserves, and designated areas where the preservation and enhancement of natural resources requires special protection. In these areas, alteration of natural water quality is undesirable. The areas that have been designated as ASBS in this region are depicted in. The state Ocean Plan requires wastes to be discharged at a sufficient distance from these areas to assure maintenance of natural water quality conditions.

Ocean, Commercial, and Sport Fishing (COMM)

This category includes the use of water for commercial or recreational collection of fish, shellfish, or other organisms in oceans, bays, and estuaries, including, but not limited to, uses involving organisms intended for human consumption or bait purposes. To maintain ocean fishing, the aquatic life habitats where fish reproduce and seek their food must be protected.

Estuarine Habitat (EST)

This category includes the uses of water that support estuarine ecosystems, they include, but are not limited to, preservation or enhancement of estuarine habitats, vegetation, fish, shellfish, or wildlife (e.g., estuarine mammals, waterfowl, shorebirds), and the propagation, sustenance, and migration of estuarine organisms. Estuarine habitat provides an essential and unique habitat that serves to acclimate anadromous fishes (salmon, striped bass) migrating into fresh or marine water conditions. The protection of estuarine habitat is contingent upon (1) the maintenance of adequate freshwater outflow to provide mixing and salinity control; and (2) provisions to protect wildlife habitat associated with marshlands and the Bay periphery (i.e., prevention of fill activities). Estuarine habitat is generally associated with moderate seasonal fluctuations in dissolved oxygen, pH, and temperature and with a wide range in turbidity.

Industrial Service Supply (IND)

The IND category includes the uses of water for industrial activities that do not depend primarily on water quality, including, but not limited to, mining, cooling water supply, hydraulic conveyance, gravel washing, fire protection, and oil well repressurization. Most industrial service supplies have essentially no water quality limitations except for gross constraints, such as freedom from unusual debris.

Marine Habitat (MAR)

The MAR beneficial use category includes the uses of water that support marine ecosystems, including, but not limited to, preservation or enhancement of marine habitats, vegetation such as kelp, fish, shellfish, or wildlife (e.g., marine mammals, shorebirds). In many cases, the protection of marine habitat will be accomplished by measures that protect wildlife habitat generally, but more stringent criteria may be necessary for waterfowl marshes and other habitats, such as those for shellfish and marine fishes. Some marine habitats, such as important intertidal zones and kelp beds, may require special protection.

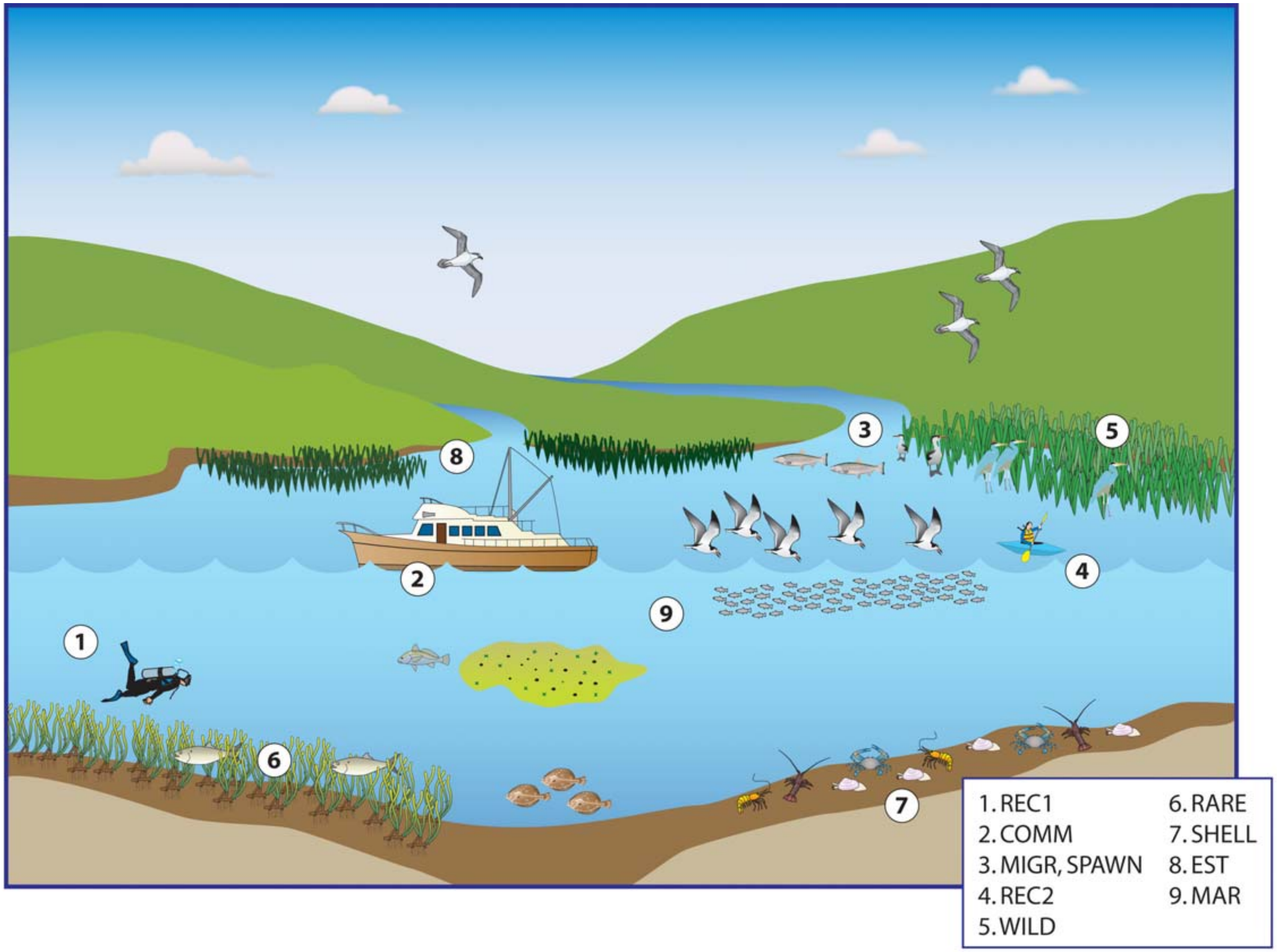


Figure 2-2 Illustration of the typical primary producer communities (macroalgae, submerged aquatic vegetation (SAV), phytoplankton, benthic algae, and macrophytes) in an estuary and related beneficial uses (REC1, COMM, MIGR, SPWN, REC2, WILD, RARE, SHELL, EST). Due to the low probability of impacts from nutrient enrichement the following beneficial uses have not been included: ASBS, IND, NAV, and PROC.

Fish Migration (MIGR)

This category includes the uses of water that support habitats necessary for migration, acclimatization between fresh water and salt water, and protection of aquatic organisms that are temporary inhabitants of waters within the region. The water quality provisions acceptable to cold water fish generally protect anadromous fish as well. However, particular attention must be paid to maintaining zones of passage. Any barrier to migration or free movement of migratory fish is harmful. Natural tidal movement in estuaries and unimpeded river flows are necessary to sustain migratory fish and their offspring. A water quality barrier, whether thermal, physical, or chemical, can destroy the integrity of the migration route and lead to the rapid decline of dependent fisheries. Water quality may vary through a zone of passage as a result of natural or human- induced activities. Fresh water entering estuaries may float on the surface of the denser salt water or hug one shore as a result of density differences related to water temperature, salinity, or suspended matter.

Navigation (NAV)

This category includes the uses of water for shipping, travel, or other transportation by private, military, or commercial vessels.

Industrial Process Supply (PROC)

The PROC beneficial use category includes the uses of water for industrial activities that depend primarily on water quality. Water quality requirements differ widely for the many industrial processes in use today. So many specific industrial processes exist with differing water quality requirements that no meaningful criteria can be established generally for quality of raw water supplies. Fortunately, this is not a serious shortcoming, since current water treatment technology can create desired product waters tailored for specific uses.

Preservation of Rare and Endangered Species (RARE)

This category includes uses of waters that support habitats necessary for the survival and successful maintenance of plant or animal species established under state and/or federal law as rare, threatened, or endangered. The water quality criteria to be achieved that would encourage development and protection of rare and endangered species should be the same as those for protection of fish and wildlife habitats generally. However, where rare or endangered species exist, special control requirements may be necessary to assure attainment and maintenance of particular quality criteria, which may vary slightly with the environmental needs of each particular species. Criteria for species using areas of special biological significance should likewise be derived from the general criteria for the habitat types involved, with special management diligence given where required.

Water Contact Recreation (REC1)

The REC1 category includes uses of water for recreational activities involving body contact with water where ingestion of water is reasonably possible. These uses include, but are not limited to, swimming, wading, water-skiing, skin and scuba diving, surfing, whitewater activities, fishing, and uses of natural hot springs. Water contact implies a risk of waterborne disease transmission and involves human health; accordingly, criteria required to protect this use are more stringent than those for more casual water-oriented recreation. Excessive algal growth has reduced the value of shoreline recreation areas in some cases, particularly for swimming. Where algal growths exist in nuisance proportions, particularly blue green algae, all recreational water uses, including fishing, tend to suffer. One criterion to protect the aesthetic quality of waters used for recreation from excessive algal growth is based on chlorophyll a.

Noncontact Water Recreation (REC2)

The REC2 category includes uses of water for recreational activities involving proximity to water, but not normally involving contact with water where water ingestion is reasonably possible. These uses include, but are not limited to, picnicking, sunbathing, hiking, beachcombing, camping, boating, tide pool and marine life study, hunting, sightseeing, or aesthetic enjoyment in conjunction with the above activities. Water quality considerations relevant to noncontact water recreation, such as hiking, camping, or boating, and those activities related to tide pool or other nature studies require protection of habitats and aesthetic features. In some cases, preservation of a natural wilderness condition is justified, particularly when nature study is a major dedicated use. One criterion to protect the aesthetic quality of waters used for recreation from excessive algal growth is based on chlorophyll a.

Shellfish Harvesting (SHELL)

This category includes uses of water that support habitats suitable for the collection of crustaceans and filter-feeding shellfish (e.g., clams, oysters, and mussels) for human consumption, commercial, or sport purposes. Shellfish harvesting areas require protection and management to preserve the resource and protect public health. The potential for disease transmission and direct poisoning of humans is of considerable concern in shellfish regulation. Toxic metals can accumulate in shellfish. Mercury and cadmium are two metals known to have caused extremely disabling effects in humans who consumed shellfish that concentrated these elements from industrial waste discharges. Other elements, radioactive isotopes, and certain toxins produced by particular plankton species also concentrate in shellfish tissue. Documented cases of paralytic shellfish poisoning are not uncommon in California.

Fish Spawning (SPWN)

The SPWN beneficial use category includes the uses of water that support high quality aquatic habitats suitable for reproduction and early development of fish. Dissolved oxygen levels in spawning areas should ideally approach saturation levels. Free movement of water is essential to maintain well-oxygenated conditions around eggs deposited in sediments. Water temperature, size distribution and organic content of sediments, water depth, and current velocity are also important determinants of spawning area adequacy.

Wildlife Habitat (WILD)

This category includes the uses of waters that support wildlife habitats, including, but not limited to, the preservation and enhancement of vegetation and prey species used by wildlife, such as waterfowl. The two most important types of wildlife habitat are riparian and wetland habitats. These habitats can be threatened by development, erosion, and sedimentation, as well as by poor water quality. The water quality requirements of wildlife pertain to the water directly ingested, the aquatic habitat itself, and the effect of water quality on the production of food materials. Waterfowl habitat is particularly sensitive to changes in water quality. Dissolved oxygen, pH, alkalinity, salinity, turbidity, settleable matter, oil, toxicants, and specific disease organisms are water quality characteristics particularly important to waterfowl habitat. Dissolved oxygen is needed in waterfowl habitats to suppress development of botulism organisms; botulism has killed millions of waterfowl. It is particularly important to maintain adequate circulation and aerobic conditions in shallow fringe areas of ponds or reservoirs where botulism has caused problems.

While all designated uses must be considered, some are unlikely to be impaired by nutrients before other, more sensitive assigned uses covering the basics of the national “fishable, swimmable” goals are also impaired (e.g., industrial service supply, navigation, industrial process supply). Such uses are not likely to be the driving force for nutrient criteria at a site. Areas of Special Biological Significance and Preservation of Rare and Endangered Species would appear to require site-specific management plans. Accordingly, the remainder of this discussion focuses on some of the other designated uses that are both commonly assigned and, to one degree or another, sensitive to impairment by nutrients. These are: Estuarine Habitat (EST), Marine Habitat (MAR), Shellfish Harvesting (SHELL), Fish Migration (MIGR), Water Contact Recreation (REC-1), Noncontact Water Recreation (REC-2), and Commercial and Sport Fishing (COMM), and Fish Spawning (SPWN).

2.3 Conceptual Models Linking Stressors to Impaired Beneficial Uses

Ecological risk assessment (ERA) is a process for evaluating the likelihood that adverse ecological impacts may occur in response to one or more stressors. ERA consists of three phases: planning and problem formulation, risk analysis, and risk characterization. These phases are described in detail in EPA's Guidelines for Ecological Risk Assessment (U.S. EPA, 1998a). The keys to a successful ERA are (1) identifying the pathways by which stressors cause ecological effects and (2) providing informative and representative assessment endpoints. Assessment endpoints are the link between scientifically measurable endpoints and the objectives of stakeholders and resource managers. Endpoints should be ecologically relevant, related to environmental management objectives, and susceptible to stressors (USEPA, 1998b).

A pivotal tool of the ERA process is development and evaluation of a conceptual model, and selection of assessment endpoints. A conceptual model is a graphical and narrative description of the potential physical, chemical and biological stressors within a system, their sources, and the pathways by which they are likely to impact multiple ecological resources. The conceptual model is important because it links exposure characteristics such as water quality parameters (related to water quality standards) with the ecological endpoints important for describing the management goals (related to aquatic life support as designated under the Clean Water Act).

Conceptual model development has been identified as the single most valuable component of EPA's watershed-level ecological risk assessment case studies (Butcher et al., 1998). In each of the five EPA-sponsored case studies, conceptual model development in accordance with the ERA framework was identified as particularly valuable in providing a solid foundation for stakeholder communication, strategic data collection, and priority ranking and targeting. Conceptual models consist of two general components (USEPA, 2001): (1) a description of the hypothesized pathways between human activities (sources of stressors), stressors, and assessment endpoints; and (2) a diagram that illustrates the relationships between human activities, stressors, and direct and indirect ecological effects on assessment endpoints. The conceptual model consolidates available information on ecological resources, stressors, and effects, and describes, in narrative and graphical form, relationships among human activities, stressors, and the effects on valued ecological resources

A conceptual model provides a visual representation for the cases where multiple stressors contribute to water quality problems. The pathways or relationships that are of greatest interest or concern to stakeholders will form the risk hypotheses that are specifically examined in the risk assessment. Thus, the conceptual model will summarize or depict those risk hypotheses. Specific assumptions or hypotheses may be based on theory and logic, empirical data, information from other watersheds, or mathematical models. Thus, they are formulated using a combination of professional judgment and available information on the ecosystem at risk, potential sources of stressors, stressor characteristics, and observed or predicted ecological effects on selected or potential assessment endpoints. With the conceptual model, some attribute or related surrogate (termed an "indicator" in both the watershed approach [USEPA 1995] and the TMDL program) provides a measurable quantity that can be used to evaluate the relationship between pollutant sources and their impact on water quality (USEPA, 1999).

An important role of these statements is to make explicit the rationale for selecting measures or indicators and the choice of modeling or linkage analysis tools. The goal is tied to a stressor by an exposure process. This leads directly to the consideration of measures to manage stressors, and the need for linkage tools that can assess the process of upland sediment generation, loading to the stream, and impact on the substrate. There are many complex ways in which excess nutrient loads can impact one or more designated uses. A generalized conceptual model for the impairment of key uses in estuaries by nutrients is presented in Figure 2-1. Table 2-1 summarizes the beneficial uses that are impacted by the various biological response indicators. The illustrated conceptual models also include major exogenous factors that influence how nutrients are processed within estuaries, and/or have a direct impact on the endpoints. Exogenous factors are included in the California approach because they are critical to the decision-making process to maintain or restore waterbody integrity. These exogenous factors, identified in the conceptual model, also affect the allowable nutrient levels necessary to maintain or protect the desired beneficial uses. Additional linkages may be significant in individual waterbodies; however, most of the major linkage connections are captured in these figures. Section 3 provides a definition of eutrophication and brief review of the major sources of nutrients to estuaries, indicators of biological response, and exogenous factors affecting estuarine biological response to nutrient loads.

Each pathway (from the nutrient load stressor to one of the use impairments) through the conceptual models constitutes a risk hypothesis. Given the complexity of the conceptual models, there are many individual pathways or risk hypotheses to consider. In a place-based watershed ERA, one would typically begin with a full conceptual model (modified as appropriate for the watershed under study), identify the most significant pathways, then proceed with the analysis using these selected pathways as the key risk hypotheses. For generalized nutrient criteria the concept is still relevant; however, there is not the luxury of sifting the many potential risk hypotheses for importance based on site-specific characteristics. Therefore, it is necessary to pare the list to identify, in generic form, those risk hypotheses that are most likely to be important and/or can stand in as surrogates for other, less common risk pathways.

The complex conceptual models may first be reduced to a table showing the relationship of key uses to major stressor response factors that can be key causes of impairment of use, as shown in Figure 2-2. The stressor-response factors primarily relate to problems of excess algal or macrophyte growth, and may be further simplified to generic risk hypotheses. This simplified, generic risk hypotheses is summarized below:

Excess nutrient load results in excess primary producer biomass that may increase turbidity, alter the food chain, create unaesthetic conditions, and alter the DO balance and pH, leading to impairment of uses. The exact format of the risk hypotheses depends on the uses that are designated and characteristics of the waterbody.

These generic risk hypotheses are useful for criteria development because they help focus in on the key points in common site-specific risk hypotheses that control the linkage between indicators of stressors and biological response. These key indicators are discussed in greater detail in the following section.

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

Use	Key Biological Response Indicators							
	Hypoxia	**Unaesthetic algal blooms (macroalgae, phytoplankton, benthic algae)**	**Decreased SAV cover/ density**	**Increased HABs**	**Altered food chain**	**Bad odor or taste**	**Toxic metal, ammonia cycling**	**pH**
ASBS	X		X	X	X	X	X	**X**
COMM	X		X	X	X	X	X	**X**
EST	X		X	X	X		X	**X**
MAR	X		X	X	X		X	**X**
MIGR	X		X	X	X		X	**X**
RARE	X		X	X	X		X	**X**
REC1		X		X		X	X	**X**
REC2		X	X		X	X		
SHELL	X			X	X	X	X	**X**
SPAWN	X		X	X	X		X	**X**
WILD	X		X	X	X		X	**X**

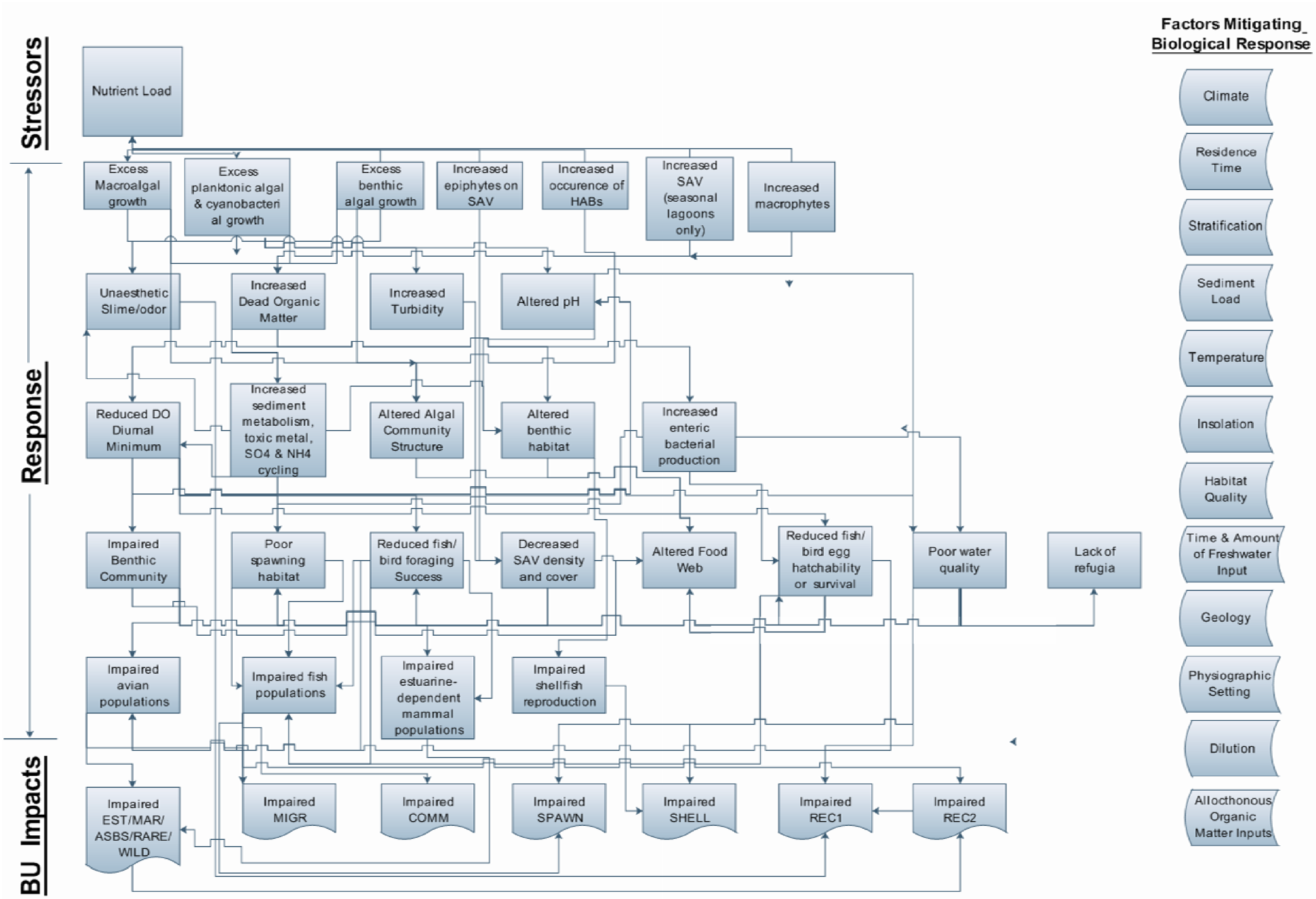


Figure 2-3 Conceptual model depicting relationship between stressors and biological response and linkage to impairment of beneficial uses.

3 EUTROPHICATION IN ESTUARIES: BACKGROUND INFORMATION ON SOURCES, BIOLOGICAL RESPONSES AND MITIGATING FACTORS

In this document, we utilize the definition of eutrophication as given by the NOAA US National Estuarine Eutrophication Assessment (Bricker et al. 2003):

"Eutrophication is a natural process by which productivity of a water body, as measured by organic matter, increases as a result of increasing nutrient inputs. Although these inputs are a result of natural processes, they have been greatly supplemented by various human related activities. Cultural eutrophication, or nutrient overenrichment, is the enhanced accumulation of organic matter, particularly algae, that is caused by human related increases in the amount and composition of nutrients being discharged to the water body. A variety of impacts may result, including nuisance and toxic algal blooms, depleted dissolved oxygen, and loss of submerged aquatic vegetation and benthic fauna. These impacts are interrelated and usually viewed as having a negative effect on water quality, ecosystem health, and human uses. Management concerns should address the human, or cultural, portion of nutrient additions insofar as the additions are detrimental to the environment."

To manage eutrophication, it is important to understand not only the sources of nutrients to estuaries relative to historic levels, but also the processes by which nutrients are transformed and recycled within the estuary, and how they are linked to eutrophication and the impairment of beneficial uses. Among the major direct and indirect ecological impacts of eutrophication cited include: 1) increased incidences of low oxygen events (hypoxia and anoxia), 2) increased productivity of primary producer communities and shifts in the dominance of species within those communities and concomitant reduction in water clarity, and 3) changes in the trophic structure and interactions of phytoplankton, zooplankton, and benthic communities. In this brief review of literature, we attempt to synthesize information on the principal sources, the biological responses of estuaries and coastal waters to elevated nutrient loading and the physical factors that mitigate response of estuaries to this disturbance.

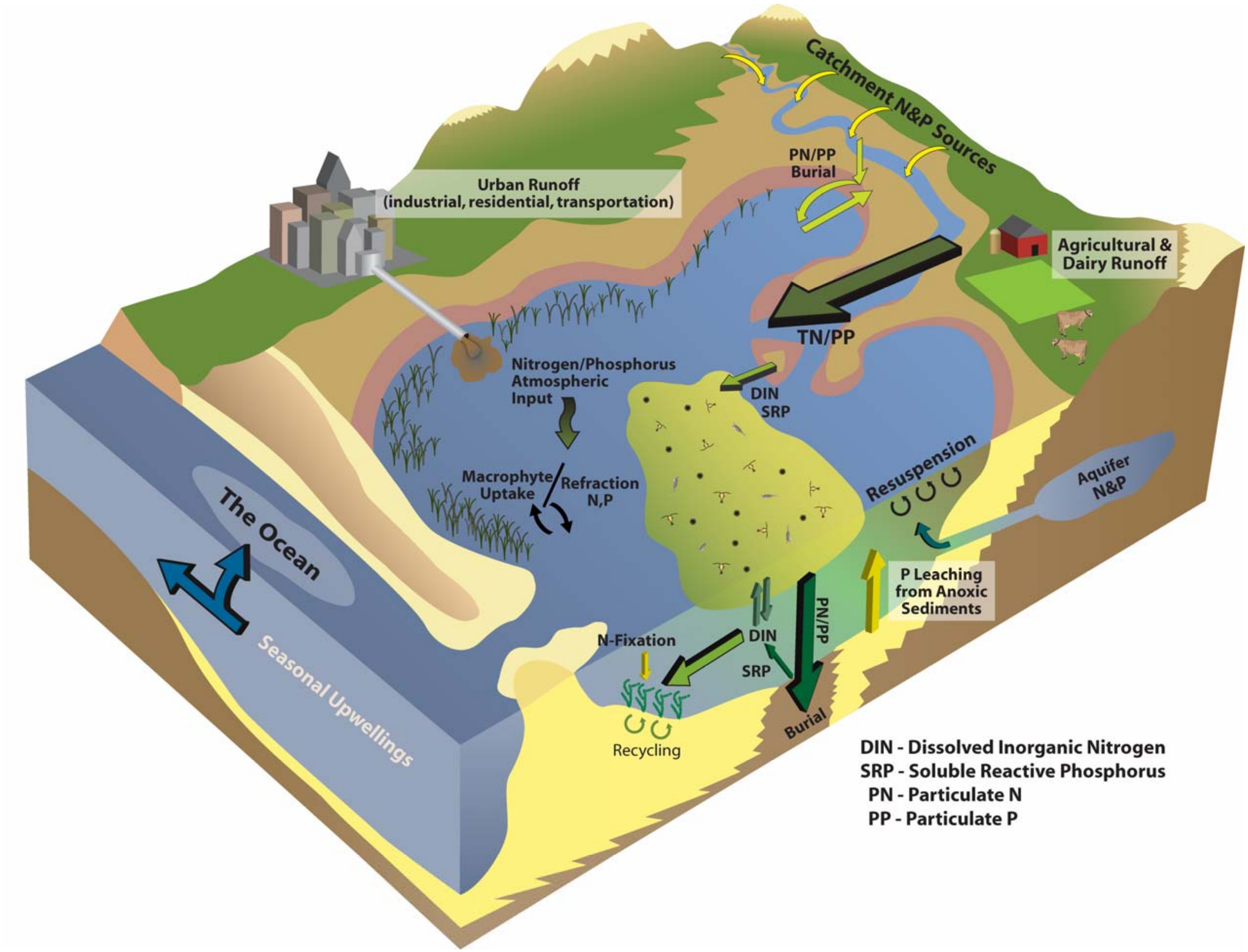
Sources of Nutrients to Estuaries

Human activities on coastal watersheds provide the major sources of nutrients entering shallow coastal ecosystems. Sources include groundwater, atmospheric deposition, municipal wastewater treatment and other point source discharges, urban and agricultural NPS runoff, and upwelling of nutrient-rich waters from the coastal ocean. The relative importance of these sources for each estuary or coastal region depends on factors such as the size of the watershed and its geology and hydrology, the climate and rainfall patterns, population size, degree of urbanization, the mix of land uses, the amount of groundwater input, etc.

Inputs from treated municipal wastewater effluent can sometimes be a major source of nitrogen to an estuary when the watershed is heavily populated and large relative to volume of the estuary itself. (Nixon and Pilson 1983). However, nitrogen (N) and phosphorus (P) from non-point sources exceed point sources in most estuaries. Awareness is also increasing of the importance of atmospheric deposition in the nutrient budgets of coastal and oceanic waters (Duce 1986; Galloway et al. 1996; Paerl 1995). Fischer and Oppenheimer (1991) found that 39% of the "new" N entering the Chesapeake Bay watershed was attributable to atmospheric deposition. Atmospheric deposition is the major N source in 12 northern Florida watersheds (Fu and Winchester 1994; Winchester et al. 1995). Higher rates of atmospheric N and P deposition are associated with proximity to urban or industrial areas (Paerl 1995;

Redfield 1998). Dry deposition of nutrients, which is derived from resuspended agricultural soils, mining activities, urban emissions, and long-range transport of dust, has been estimated to comprise as much as 30-50% of bulk P deposition in Florida (Meyers and Lindberg 1997). In Santa Monica Bay, atmospheric deposition of trace metals comprises 43-99% of the total loading to the Bay (Sabin et al. 2005). The significance of dry deposition in arid areas such as southern California underscores the potential importance of atmospheric deposition to N and P budget for estuaries and coastal waters. This term, however, has not been quantified.

Offshore waters on the continental shelf can receive nutrients from several sources, including deep ocean water, river and sewage inputs from land, and direct deposition from the atmosphere (Nixon et al. 1996; Prospero et al. 1996; Howarth 1998). The relative importance of these sources varies among the coastal waters of the United States, in part because of differences in ocean circulation patterns, particularly upwelling of nutrient-rich waters from the deep ocean onto the continental shelf. For most of the continental shelf area of California, upwelling is assumed to be the dominant nutrient input. However, input from the watershed sources can have a tremendous localized impact on nearshore waters and merits better quantification.



DIN - Dissolved Inorganic Nitrogen
SRP - Soluble Reactive Phosphorus
PN - Particulate N
PP - Particulate P

Figure 3.1 Sources of nutrients into estuarine environments

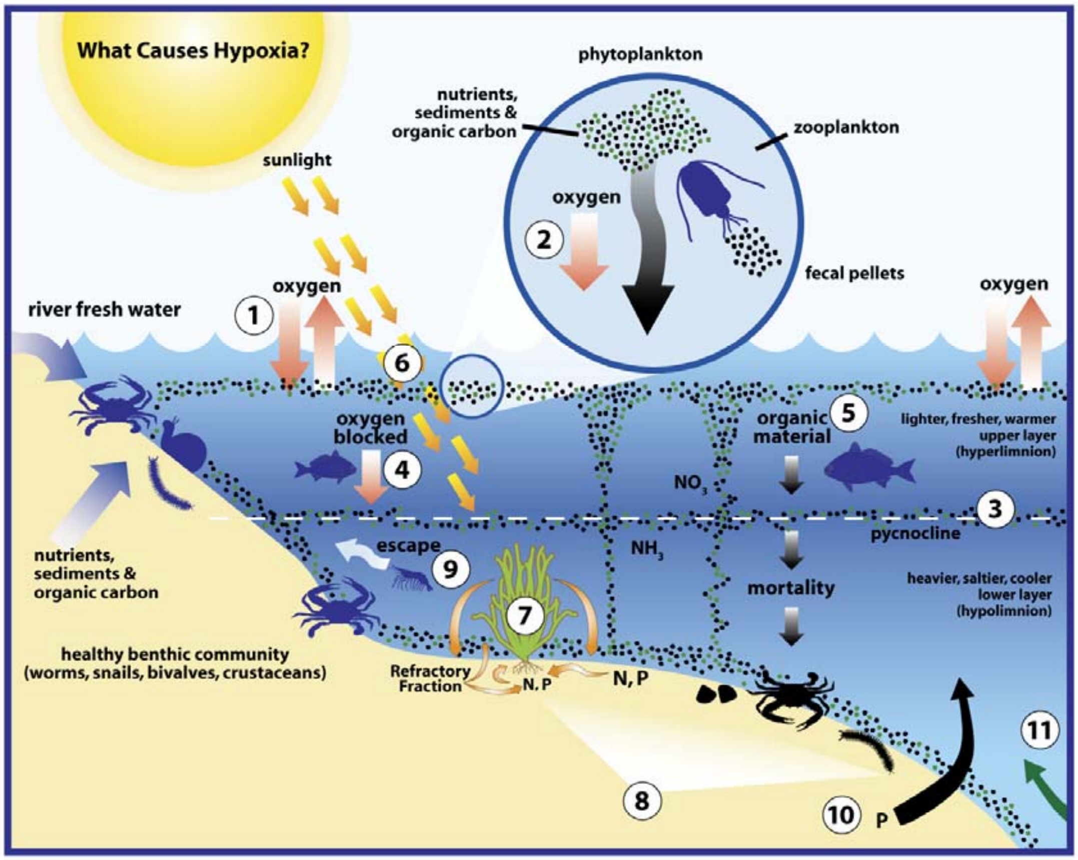
3.1 Biological Response of Estuaries to Nutrient Loading

The biological response of estuaries to nutrient loading is complex, varying greatly as a function of physiographic setting, tidal regime, timing and magnitude of freshwater inputs, etc. The purpose of this section is to summarize some of the major elements of an estuarine ecosystem that are impacted by eutrophication. The intent of this review is a brief summary – not an extensive literature review. Several excellent reviews of this subject matter exist and can be referred to for more information (e.g. NAS 2000)

Hypoxia/Anoxia

The occurrence of hypoxic and anoxic bottom waters, particularly in the coastal zone, has become a major concern in recent years because it appears that the frequency, duration, and spatial coverage of such conditions have been increasing, and this increase is thought to be related to human activities (Diaz and Rosenberg 1995). Zones of reduced oxygen can disrupt the migratory patterns of benthic and demersal species, lead to reduced growth and recruitment of species, and cause large kills of commercially important invertebrates and fish. Such conditions can also lead to an overall reduction in water quality, thereby affecting other coastal zone activities such as swimming and boating. Reports of a "dead zone," an extensive area of reduced oxygen levels covering an expanse originally of some 9,500 km^2 in the Gulf of Mexico (Rabalais et al. 1991), have focused attention on the problem of coastal zone hypoxia.

Hypoxic conditions can develop when there is an overproduction of organic matter, typically algal blooms, resulting in the consumption of oxygen (Figure 3-2). While algal blooms, through photosynthesis, will raise DO saturation during daylight hours, the dense population of a bloom reduces DO saturation during the night. When phytoplankton cells die, they sink towards the bottom waters or sediments and are decomposed by bacteria, a process that further reduces DO in the water column. Hypoxia conditions can be exacerbated or prolonged by physical conditions. The shape of the waterbody and the direction and magnitude of freshwater and tidal flows determine the extent to which stratification occurs. Stratification, or poorly mixed water, can enhance the occurrence and duration of hypoxia. Anthropogenic activities that result in hydrological modification in estuaries can increase residence time and stratification of estuarine waters, thus enhancing conditions in which hypoxia can occur.



1. Atmospheric oxygen equilibrium
2. Photosynthetic input of oxygen by phytoplankton
3. Pycnocline generated as the heavier, cooler, saltier marine layer is overlaid by lighter, warmer and fresher riverine inputs.
4. Pycnocline prevents significant exchange between the oxygenated hyperlimnion and the anoxic hypolimnion.
5. Organic material sinks (less dense material accumulates at pycnocline; more dense material sinks deeper)
6. Organic material accumulated at pycnocline reduces the amount of light that can reach the bottom
7. Macroalgae uptake nutrients during periods of non-stratification and grow. During periods of stratification, low light causes mortality.The subsequent decay af plant tissue recycles nutrients into the sediment and overlying water.
8. Mortality of organisms stimulates bacterial decomposition, which further reduces the oxygen levels
9. Organisms that are mobile can escape the anoxic conditions to more habitable environs.
10. Phosphorus is leached from the anoxic sediments into the hypolimnion.
11. Upwelled nutrients and oxygen.

Figure 3-2 Conceptual model of processes responsible for the development of hypoxia

Phytoplankton Biomass and Community Dynamics (Chlorophyll a/, Turbidity, and Suspended Solids)

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. In some estuaries, nutrients cause dense algal blooms to occur for months at a time, blocking sunlight to submerged aquatic vegetation. Decaying algae from the blooms uses 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 (or 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. Some species are well adapted to low-nutrient conditions where inorganic compounds predominate, whereas others thrive only when major nutrient concentrations are elevated or when 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 and 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 N, P, and silica, it is no surprise that the phytoplankton community composition is altered by nutrient enrichment (Jørgensen and Richardson 1996). The consequences of changes in phytoplankton species composition on grazers and predators can be great, but in general these are poorly studied.

Macroalgal Biomass and Community Dynamics

One consequence of increased nutrient availability in shallow estuaries is macroalgal blooms (Peckol and Rivers 1995, Taylor et al. 1995, Valiela et al. 1992). Although macroalgae are a natural component of these systems, their proliferation due to nutrient enrichment reduces habitat quality. Respiration may reduce dissolved O_2 content of estuarine waters at night (e.g., Peckol and Rivers 1995), while decomposition may cause a large microbial O_2 demand both day and night (Sfriso et al. 1987). Ephemeral benthic macroalgae have light requirements that are significantly less than either seagrasses or perennial macroalgae, and also can shade perennial macrophytes such as seagrasses and contribute to their decline (Markager and Sand-Jensen 1990; Duarte 1995). These nuisance 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, and this can lead to destruction of these submerged aquatic seagrass systems. Massive and persistent macroalgal blooms ultimately displace seagrasses and perennial macroalgae through shading effects (Valiela et al. 1997).

Where mass accumulations of macroalgae occur, their characteristic bloom and die-off cycles influence oxygen dynamics in the entire ecosystem. As a result, eutrophic shallow estuaries and lagoons often experience frequent episodic oxygen depletion throughout the water column rather than the seasonal bottom-water anoxia that occurs in stratified, deeper estuaries (Sfriso et al. 1992; D’Avanzo and Kremer 1994). Benthic macroalgae also uncouple sediment mineralization from water column production by intercepting nutrient fluxes at the sediment-water interface (Thybo-Christesen et al. 1993; McGlathery et al. 1997) and can outcompete phytoplankton for nutrients (Fong et al. 1993). Except during seasonal macroalgal die-off events in these shallow systems, phytoplankton production is typically nutrient-limited and water column chlorophyll concentrations are uncharacteristically low despite high nutrient loading (Sfriso et al. 1992). The nuisance algae also wash up on beaches, creating foul-smelling piles. Macroalgal blooms also impair beneficial recreational uses such as boating, swimming and fishing.

Submerged Aquatic Vegetation (SAV)

Many coastal waters are shallow enough that benthic plant communities can contribute significantly to autotrophic production if sufficient light penetrates the water column to the seafloor. In areas of low nutrient inputs, dense populations of seagrasses and perennial macroalgae (including kelp beds) can attain rates of net primary production that are as high as the most productive terrestrial ecosystems (Charpy-Roubaud and Sournia 1990). These perennial macrophytes are less dependent on water column nutrient levels than phytoplankton and ephemeral macroalgae, and light availability is usually the most important factor controlling their growth (Sand-Jensen and Borum 1991; Dennison et al. 1993; Duarte 1995). As a result, nutrient enrichment rarely stimulates these macrophyte populations, but instead causes a shift to phytoplankton or bloom-forming benthic macroalgae as the main autotrophs. Fast-growing micro- and macroalgae with rapid nutrient uptake potentials can replace seagrasses as the dominant primary producers in enriched systems (Duarte 1995; Hein et al. 1995). The biotic diversity of the community generally decreases with these nutrient-induced changes.

Over the last several decades, nuisance blooms of macroalgae in association with nutrient enrichment have been increasing along many of the world's coastlines (Lapointe and O'Connell 1989). Phytoplankton biomass and total suspended particles increase in nutrient enriched waters and reduce light penetration through the water column to benthic plant communities.

Decreased photosynthetic oxygen production at all light levels also reduces the potential for oxygen translocation and release to the rhizosphere, and creates a positive feedback that reduces sulfide oxidation around the roots, further elevating sediment sulfide levels. In Florida, chronic sediment hypoxia and high sediment sulfide concentrations have been associated with the decline of the tropical seagrass Thalassia testudinum (Robblee et al. 1991). Elevated sediment sulfide, associated with excessive organic matter accumulation, has been shown experimentally 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).

Epiphytes are algae that grow on the surfaces of plants or other objects. Epiphytic microalgae become more abundant on seagrass leaves in eutrophic waters and contribute to light attenuation at the leaf surface, as well as to reduced gas and nutrient exchange (Tomasko and Lapointe 1991; Sand-Jensen 1977). Thus they can cause losses of submerged aquatic vegetation by encrusting leaf surfaces and thereby reducing the light available to the plant leaves.

Declines in seagrass distribution caused by decreased light penetration in deeper waters or changes in community composition prompted by the proliferation of benthic macroalgae in shallower waters will have significant trophic consequences. Seagrass roots and rhizomes stabilize sediments, and their dense leaf canopy promotes sedimentation of fine particles from the water column. Loss of seagrass coverage increases sediment resuspension and causes an efflux of nutrients from the sediment to the overlying water that can promote algal blooms. Low dissolved oxygen (DO) has direct negative impacts on fish and can lead to mortality (Coon 1998). Extended periods of low oxygen can lead to changes in overall species composition, shifts in community structure, and loss of biodiversity (Raffaelli et al. 1991, Edgar et al. 2000).Seagrasses also provide food and shelter for a rich and diverse fauna, and reduced seagrass depth distribution or replacement by macroalgal blooms will result in marked changes in the associated fauna (Thayer et al. 1975; Norko and Bonsdorff 1996).

Harmful Algal Blooms (HAB)

Harmful algal blooms are microscopic algae at the base of the marine ecosystem that can produce potent toxins or that cause harm to humans, fisheries resources, and coastal ecosystems. The impacts of HABs include fish kills, human intoxications or even death from contaminated shellfish or fish, alterations of marine trophic structure through adverse effects on larvae and other life history stages of commercial fisheries species, and death of marine mammals, seabirds, and other animals. The production of toxins by HAB species occurs by physical, chemical, and biological mechanisms that are poorly understood; a HAB may occur without the production of toxins.

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., 1989a or b; 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 more than1,400 mammal stranding incidents within the SCB 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. They are also 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 U.S. 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.

Shifts in Benthic Communities

The occurrence of hypoxic and anoxic bottom waters may also lead to shifts in benthic and pelagic community structure due to the mortality of less mobile or more sensitive taxa, reduction of suitable habitat, and shifts in predator-prey interactions (Diaz and Rosenberg 1995). Hypoxia plays a major role in the structuring of benthic communities because species differ in the sensitivity to oxygen reduction (Diaz and Rosenberg 1995). The response of species to reduced oxygen availability also depends on the frequency and duration of these events. With short bouts of hypoxia, some large or very motile species are able to adjust to or move away from the stress.

Hypoxia tends to shift the benthic community from being dominated by large long-lived species to being dominated by smaller opportunistic short-lived species (Pearson and Rosenberg 1978). In addition, recurring hypoxia may limit successional development to colonizing communities. In such systems more organic matter is available for remineralization by the microbial community. This can decrease the amount of energy available for benthic recruitment when hypoxia and anoxia disappears. Zooplankton that normally vertically migrate into bottom waters during the day may be more susceptible to fish predation if they are forced to restrict their activity to the oxic surficial waters. Roman et al. (1993) concluded that the vertical distribution of copepods in the Chesapeake Bay was altered by the presence of hypoxic bottom waters. Moreover, an hypoxic or anoxic bottom layer may constitute a barrier that de-couples the life cycle of pelagic species (e.g., diatoms, dinoflagellates, and copepods) that have benthic resting stages (Marcus and Boero 1998).

In a controlled eutrophication experiment (Doering et al. 1989), the structure of the zooplankton community was affected by the presence or absence of an intact benthic community. In the absence of an intact benthic community, holoplanktonic forms, especially higher level predators, dominated, whereas meroplanktonic forms were more evident in the presence of an intact benthic community. Although the data did not identify the mechanism behind these shifts, the differences likely reflected alterations in the coupling of the benthic and pelagic environments (nutrient as well as life cycle linkages) (Marcus and Boero 1998).

Shifts in Fish Communities

Changes in predator-prey interactions in the water column can also lead to shifts in energy flow. If the duration and severity of the hypoxia is not sufficient to cause mortality of the macrobenthos, the increased supply of organic matter to the benthic system could fuel the growth of benthic fauna and demersal fish populations at the expense of pelagic fisheries. On the other hand, extended hypoxic and anoxic events could lead to the demise of the macrobenthos and the flourishing of bacterial mats. The loss of burrowing benthic organisms that irrigate the sediments and the presence of an extensive bacterial community may alter geochemical cycling and energy flow between the benthic and pelagic systems (Diaz and Rosenberg 1995). For example, the flux of nitrogen out of the sediments is affected by the rates of nitrification and denitrification, and these processes depend on the naturally oxic and anoxic character of the sediments.

3.2 Factors Affecting Estuarine Biological Response to Nutrient Loads

Certain key characteristics appear to be of primary importance in determining estuarine response to nutrient enrichment. These factors are (NAS 2000):

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.

Nutrient Load

Nutrient load is the total mass of various nutrients contributed by the upstream landscape and atmosphere. Nutrient load should be distinguished from ambient nutrient concentration, which refers to the amount of nitrogen or phosphorus in a defined volume of estuarine surface waters (such as milligrams of nitrogen per liter of water).

There has been a great deal of discussion among scientists and managers about whether ambient nutrient concentrations or nutrient loading to the estuary is more relevant as an indicator of eutrophication. Many managers prefer ambient nutrient concentrations, because of the ease, precision, and comparatively low cost of measurement. Nutrient loads represent the total source of nutrients to the system during a given time period, and thus is a measure of the magnitude of ecosystem-level primary producer response from a mass balance perspective. In contrast, ambient nutrient concentrations are a measure what primary producers (e.g. phytoplankton, macroalgae, vascular plants, etc.) can uptake within short time scales (Boynton and Kemp 2000). Macroalgae are known to rapidly uptake dissolved inorganic nitrogen that within timescales of hours to days can bring surface water concentrations down to near detection limits. Thus, depending on the time scale of nutrient delivery to the estuary versus biological response, ambient nutrient concentrations often reflect the remaining inventory of nutrients that are left over or that which has been are recycled into organic forms. For this reason, ambient nutrient concentrations are often not significantly correlated to biological response indicators (e.g. Kennison et al. 2003).

The significance of the relationship between nutrient load and biological response has also been variable, in part because of the range in lag time between the load and response variables as well as other factors that control biological response. (Kemp and Boynton 1984, Malone et al. 1988). Nixon et al. (1996) and Dettman (2001) have demonstrated that the regression relationship between load and response improves by taking in to account factors such as freshwater residence time, estuarine volume, denitrification rate, etc. While it is generally recognized that nutrient loads themselves are not suitable as criteria, there appears to be consensus within the scientific community that monitoring of nutrient loads is key to identifying sources and setting maximum loads and is therefore more useful than limiting the ambient concentration of a given nutrient in the estuary.

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, TR, 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 spends in the estuary. Estimates of residence time are useful for calculating the movements and concentrations of dissolved substances, such as nutrients, in the estuary. The residence time within a particular estuary varies depending on many factors, including fresh-water input, circulation, and bathymetry. 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. In coastal ecosystems, the production of macroalgae, phytoplankton and other primary producers is limited by the availability of nitrogen. Moreover, phytoplankton blooms can occur only when the plankton turnover time is shorter than the water residence time. If both water residence time and phytoplankton turnover time are one day; algae are flushed from the system as fast as they are produced. 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 describes 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 Phytoplankton

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.

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 N2O, and as such represents a process by which nitrogen is permanently lost from an estuary. There are potential indirect effects of eutrophication that limit denitrification. For example, bottom water anoxia limits nitrification and hence denitrification in sediments and bottom waters. High sulfide concentrations, which are also associated with anoxic conditions, inhibit nitrification as well (Joye and Hollibaugh 1995). 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.

Allochthonous Organic Matter Inputs

Organic matter contributes directly to eutrophication. The relative magnitude of inorganic versus organic nitrogen load influences the balance between autotrophic and heterotrophic metabolism (Hopkinson and Vallino 1995). The relative magnitude of dissolved versus particulate organic matter loads influences residence time of inputs, as particles are preferentially trapped by processes operating in the estuarine turbidity maximum and by gravity. The carbon:nitrogen stoichiometry of organic matter remineralized by the benthos and denitrification further influence the balance between autotrophic and heterotrophic processes in estuaries.

Systematic variation in these factors can result in a complexity of estuarine biological responses. Thus, an understanding of how these factors vary among estuarine classes and how these classes differ with respect to load-response can lead to a predictive framework or classification scheme. This classification scheme provides the basis for the development of a robust set of NNEs and TMDL loading-response tools for each class of estuaries. The classification scheme proposed for the California Estuarine NNE framework is provided in Section 4.

4 PROCESS FOR DEVELOPING ESTUARINE NUTRIENT NUMERIC ENDPOINT (NNE) TOOLKIT

The creation of a toolkit to support development of estuarine nutrient numeric endpoints (NNE) can be approached through a set of four discrete steps, each with an inherent set of data requirements for its successful completion:

1. Develop definition and classification scheme
2. Select biological response variables
3. Develop numeric nutrient endpoints
4. Create TMDL tools

This section describes current progress through each of these steps and highlights apparent data gaps and research recommendations.

4.1 DEFINITION AND CLASSIFICATION SCHEME

The first critical steps to develop NNE's for California's estuaries involve: 1) defining what constitutes an "estuary," and 2) determining whether it is necessary to break the group of estuaries into subcategories and, if so, specifying what those categories are. The purpose of this section is to present the proposed definition of "estuary" and subcategories.

Definition of "Estuary"

For the purposes of this project, we propose the following definition of estuary, modified from the U.S. FWS definition (Cowardin et al. 1979):

"Estuaries can consist of subtidal habitats and/or adjacent tidal wetlands that have open, partly obstructed, or sporadic access to the open ocean, and in which ocean water is at least occasionally diluted by freshwater runoff from the land. Some estuaries are semi-enclosed by land. In some cases, the salinity may be periodically increased above that of the open ocean by evaporation. Offshore areas that are impacted by runoff and mixing of freshwater from rivers are also considered to be estuaries"

One of the most difficult aspects to address is the setting of the freshwater and seaward limits of an estuary. Having definition of estuary that can be applied to map or delineate the boundaries of the system is important for application of the California NNE framework to the State's water quality programs. We propose the following as a means to delineating estuarine boundaries.

- "An estuary extends upstream or toward land to the area where salinity from ocean-derived salts (i.e., largely sodium chloride) is less than 0.5 ppt (parts per thousand) during the average annual low flow of freshwater input."
- An alternative definition would be the upstream extent of tidally-influenced water level changes during average annual low flow of freshwater input. While we recognize the importance of these habitats, for the purposes of nutrient criteria we chose to exclude this region of an estuary features a wide range of freshwater primary producer communities, thus complicating development of biocriteria and TMDL tools.
- The seaward extent of an estuary is defined by: 1) an imaginary line that closes the mouth of an estuary within a semi-enclosed area (e.g., a river, bay, or sound) or (2) extending to the seaward limit of estuarine vegetation dominated by emergent marsh, shrubs, or trees,

or 3) the seaward limit of offshore areas continuously diluted by runoff to salinities less than those of the ocean.

The task of developing nutrient criteria toolkit 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). The approach used by the Regional Boards is to classify estuaries according to their beneficial use designation (Table A-1, Appendix A). 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. This lack of consistency will lead to an arbitrary application of nutrient criteria across the state.

Why Classify?

The need for classifying estuaries into subcategories arises because of the inherent differences in how estuaries respond to nutrient loads. Section 3 presented a discussion of a variety of factors which control how an estuary will respond biologically to loading. These factors include: physiographic setting, salinity regime, frequency and timing of freshwater flows, magnitude of tidal forcing, sediment load, stratification, residence time, denitrification, etc. The combination of these factors result in variation of the dominant primary producer communities (ie. phytoplankton, macroalgae, benthic algae, submerged aquatic vegetation, emergent macrophytes) among estuary types and in the pathways that control how nutrients cycle within the estuary.

NNEs are developed based on the selection of key biological response indicators. Subclasses of estuaries are necessary when: 1) the biological response indicators In the process of developing NNE's for estuaries are different, or 2) among estuaries which share a common response indicator, biological interactions greatly differ such that the thresholds for "impairment," are vastly different.

The disadvantage of classification is that it greatly increases the need for data to develop NNEs and TMDL load-response models. Thus development of estuarine subclasses must balance the need to split based on biological response versus the cost implications for criteria development.

Development of a Classification Scheme

Many of the factors that control estuarine biological response to nutrient load group together according to commonly recognized estuarine types (e.g., Madden et al. 2005). The Coastal Marine Ecological Classification Standards (CMECS), a conceptual framework for classifying all North American coastal and marine ecosystems, provides an example of these common estuary types:

Estuarine Type

- River-dominated
- Coastal lagoon
- Coastal embayment
- Fjord
- Coastal bight
- Sound
- Slough
- Open coastline

- Subestuary
- Drowned river valley
- Bar-built estuary
- Barrier island estuary
- Coastal plain estuary
- Deltaic estuary

The CMECS was designed to encompass all aquatic habitats in Coastal and Marine regimes, from wetlands to the abyssal oceans. For the purposes of this project, a reduced set of the CMECS classification is being used that is more appropriate for California's estuarine habitats. As discussed above, the decision to create subclasses was made on: 1) the biological response indicators among the estuary types are different, or 2) among estuary types which share a common response indicator, biological interactions greatly differ such that the thresholds for "impairment," would be expected to be different. Based on this premise, California's proposed classification scheme includes seven types:

- Tidal Lagoon
- Seasonally Tidal Lagoon
- Nontidal Lagoon,
- River-dominated estuary
- Protected Embayment
- Open Embayment/Coastal Estuarine Front.
- San Francisco Bay Estuary

The selection of these classes was based on seven variables that control how these estuaries respond to nutrient loading. Table 4-1 gives a list of the proposed estuary types and a summary of how they differ with respect to these master variables. San Francisco Bay estuary was selected to be in a class by itself because of the unique nature of this estuary: its size, environmental gradients, and number of subestuaries associated with contributing watersheds may lend it to developing subclasses within the bay, as was done with the Chesapeake Bay estuary.

Brief Definitions Of Each Of These Proposed Types And Examples Of Each Are As Follows:

Tidal Lagoon – These estuaries are dominated by shallow subtidal and intertidal habitat and have a long residence time due to restricted width of mouth. The inlet is continuously open to tidal influence year round (Figure 4-1)

Figure 4-1. Examples of a fully tidal lagoon (Bolinas Lagoon, Marin County (A) and Carpinteria Marsh, Santa Barbara County (B)).

Seasonally tidal lagoon/creek mouth estuary – 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 brackish vegetation for part of the year. River or creek mouth estuaries are included in this category when they experience seasonal closure of their mouths due to the longshore drift of sand (Figure 4-2)

Figure 4-2. Examples of a seasonally tidal lagoon (Big Sur Estuary, Monterrey County (A) and Carmel River Estuary, Monterey County (B)).

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 lagoon can be brackish due to input of ocean water during spring tides, storm surges or advective exchange through a sand berm, and thus can support freshwater or brackish vegetation for part of the year (Figure 4-3).

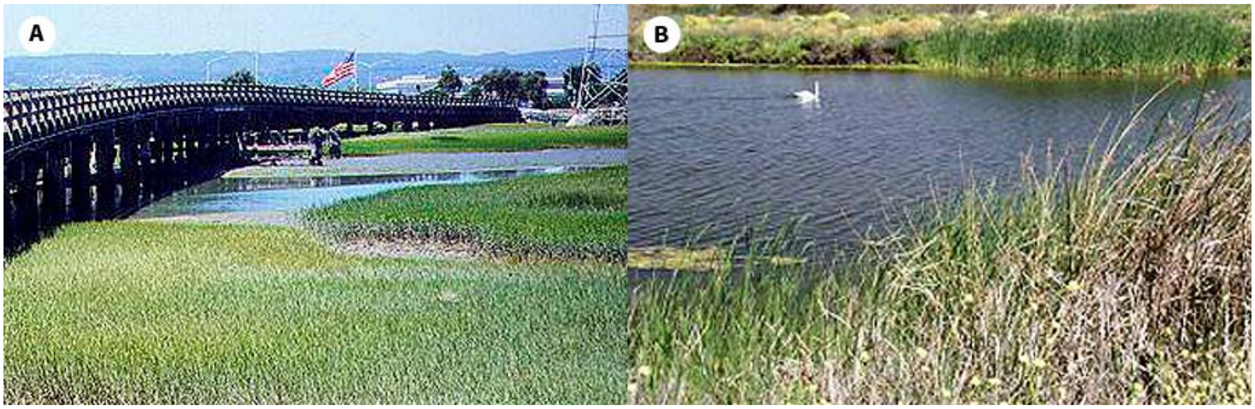


Figure 4-3. Example of a nontidal lagoons (San Mateo Lagoon, San Mateo County (A) and Buena Vista Lagoon, San Diego County (B)).

Protected embayment – This estuary type is typically a semi-enclosed land mass, dominated by subtidal or deepwater habitat. The inlet mouth is not restricted and continuously open to tidal exchange. (Figure 4-4).

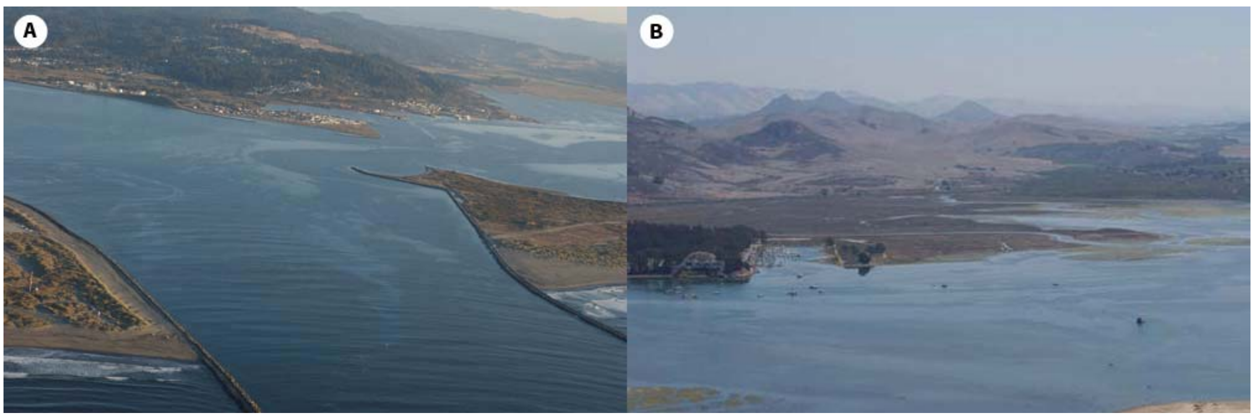


Figure 4-4. Example of protected embayments (Humboldt Bay Humboldt County (A) and Morro Bay Estuary, San Luis Obispo County (B)).

River Dominated Estuary – This class of estuary are found within the confines of high flow, perennial rivers, and is 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. (Figure 4-5).

Figure 4-5. Examples of river dominated estuaries (Big River Estuary, Humboldt County (A) and Klamath River, Klamath County (B))

Open Embayments/Coastal Estuarine Front – These estuaries are found on the continental shelf. The limits of the estuary are defined by the mixing zone of freshwater with salt water. As such boundaries are highly elastic/transient and can vary depending on location and magnitude of river plumes, currents, upwelling, etc. (Figure 4-6).

Figure 4-6. Open Embayment and Coastal Estuarine Front (Santa Monica Bay, Los Angeles County (A) and Drakes Bay, Marin County (B)

San Francisco Bay Estuary and Associated Tidal Creeks - The complexity of the San Francisco Bay/ Estuary makes it unique among west coast estuaries. The estuary contains at least four compartments that are hydrologically distinct from each other. The extreme northern compartment of the estuary receives the largest inflow of fresh water into the estuary. This portion of the estuary could be classified as “deltaic.” The central component of the estuary receives very little freshwater input and is greatly influenced by tidal action and could be classified as a “semi-protected/open embayment”. The lower two compartments include the “south bay” and “extreme south bay.” The extreme south bay encompasses the area between San Jose and the Dumbarton Bridge and is semi-hydrologically distinct and has a slower “flushing rate” than its northern neighbor the “south bay”, which extends north from the Dumbarton Bridge to just south of the Oakland – Bay Bridge. Both compartments receive some seasonal freshwater input and could be classified as “protected embayments.” (Figure 4-7)

Figure 4-7. Picture of the South San Francisco Bay Estuary, Santa Clara County

Table 4-1 Factors affecting biological response to nutrients by California estuarine class.

Variable	Tidal Lagoon	Seasonally Tidal Lagoon	Nontidal lagoon	River Dominated	Protected embayment	Open embayment/ Coastal Estuarine Front	San Francisco Bay Estuary
Physiographic setting	Enclosed water body	Enclosed water body	Enclosed water body	Open fluvial	Semi-enclosed bay	Open oceanic	Semi-enclosed water body
Major Aquatic Primary production base	SAV, macroalgae, benthic algae	SAV, benthic algae, phytoplankton, macroalgae	SAV, benthic algae, phytoplankton	Usually phytoplankton	Macroalgae, phytoplankton, SAV	Phytoplankton, SAV	Phytoplankton
Salinity Regime	0-35 ppt	0-35 ppt	0- 5 ppt	0-35 ppt	0-35 ppt	0-35 ppt	0-35 ppt
Residence Time	Medium	When closed, long	Long	Short	Medium	Short	Medium
Stratification	Well-mixed	Well-mixed	Well-mixed	Stratified	Stratified	Stratified	Stratified
Dominant habitat type	Dominated by shallow-subtidal and intertidal habitat	Dominated by shallow-subtidal and intertidal habitat	Dominated by shallow-subtidal and intertidal habitat	Dominated by deep-water habitat	Dominated by deep-water habitat	Dominated by deep-water habitat	Dominated by deep-water habitat
TSS/Light limitation	Usually only during storm events	Usually only during storm events	Usually only during storm events	High suspended load – light limited	Variable	Usually only within storm plumes	High suspended load – light limited

California's estuaries were preliminarily classified according to this scheme (Table A-2 – Appendix A). Table 4-2 provides a summary of the numbers of estuaries found in each class in California. The waterbodies designated as having EST beneficial uses were used as a starting point for the initial list of estuaries. The additional classification category "Unknown" was added for those waterbodies having an insufficient amount of information to place them into a specific estuarine type. In addition, we suspect that this initial list is not comprehensive. Therefore, we recommend a complete revision of the list of estuarine waterbodies and review of the class to which each is assigned.

In reality, each of these estuaries represents a gradient and the precise category to which a particular estuary belongs will not always be very clear. Since this is a preliminary classification, the individual estuary may be re-classified as the process becomes more refined. The best way to undertake this revision of designated estuarine class is to perform a cluster analysis between watershed nutrient loading and selected response variables (macroalgal biomass, chlorophyll a, etc.). A statistical evaluation of class is an important next step in confirming the assignment of appropriate estuarine classes based on data.

Another key consideration is whether these estuaries must be further segregated by ecoregion. In addition to estuarine classes, it may be appropriate to develop NNEs by ecoregion, since climate and oceanic forcing may exert a strong control on biological response, even within estuaries of the same class. The CMCES classification would group California into two ecoregion (CMECS Region 19 Southern California Pacific and CMECS Region 20 Montereyan Pacific Transition; Figure 4-8). These ecoregions are not completely consistent with local knowledge of climate gradients that affect California estuaries. Based on our understanding, it may be more appropriate to break California into three ecoregions, which cover the southern California Bight (to Point Conception), Central Coast (Point Conception to the Russian River), and North Coast (north of Russian River to the border with Oregon). A determination of whether these ecoregions are important in setting NNEs must be made with sufficient data for each class; these data are not currently available for California.

Table 4-2 Number of estuaries found in each class, according to nutrient criteria classification scheme. The waterbodies designated as having EST beneficial uses were used as a starting point for classification. Open embayments and coastal estuarine fronts were not enumerated.

Estuary Type	***Number Identified***
Tidal Lagoons	*30*
Seasonally Tidal Lagoons	*100*
Nontidal Lagoons	*5*
River-dominated	*11*
Protected Embayments	*16*
Open Embayment/Coastal Estuarine Front	*NA*
San Francisco Bay	*1*
Total	*163*

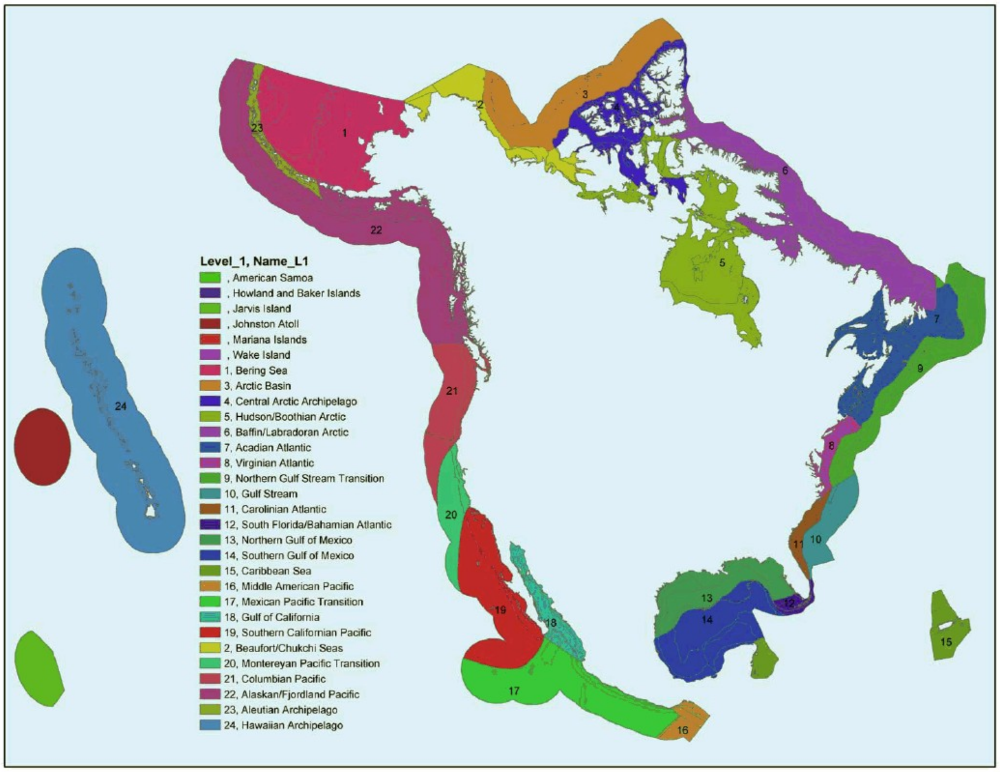


Figure 4-8. CMCES ecoregions (from Madden et al. 2005), which would group California estuaries into two ecoregions (CMECS Region 19 Southern California Pacific and CMECS Region 20 Montereyan Pacific Transition).

4.2 SELECTING BIOLOGICAL RESPONSE VARIABLES

Once a general conceptual model has been established that relates stressors (nutrient loads) to biological response and impaired beneficial uses (Figure 2-2), the next step is to select the appropriate biological response indicators for each estuarine class. These indicators will serve as the means to establish NNEs and, if Regional Boards chose, numeric criteria to determine what constitutes an "impaired" estuary for any designated beneficial use. They will also serve as the primary management endpoints by which nutrient loading from watersheds is regulated. There are three key questions that relate to the selection of biological response indicators for estuarine NNEs:

- What are the appropriate biological response indicators for each class?
- Is the linkage of these indicators with beneficial uses well documented?
- What are the "critical conditions" for their measurement?

The purpose of this section is to discuss potential biological response indicators for each estuarine class and identify associated research needs and data gaps associated with these three questions.

Potential Biological Response Indicators

In general, the characteristics of biological response indicators that are ideal candidates as management endpoints include:

- Clear linkage with beneficial use

- Chosen indicators represent the most sensitive biological values in the estuary, and
- Robust, easy to measure, and are well integrated temporally and spatially.

Table 4-3 summarizes candidate indicators of stressor (nutrient load) and biological response by estuarine class. These indicators were selected considering the most sensitive beneficial uses in the estuary.

One of the key research needs to begin development of NNEs for California estuaries is to develop fully articulated conceptual models of stressor-response-designated use for each estuarine class. This must be done with a good understanding of the range of complexity of estuaries that fall within each class. The importance of this task cannot be understated, given the range of climate and physiographic setting of estuaries within the same class along the 1100 miles of California coastline. As stated earlier, this work must address the question of whether it would be more appropriate to develop NNEs by ecoregion (e.g. southern California Bight, Central Coast (up to Russian River), and North Coast (north of Russian River), since climate and oceanic forcing may exert a strong control on biological response, even within estuaries of the same class.

Selection of the appropriate indicators for each estuarine class represents one of key research areas for the development of California estuarine NNEs. In the section below, each of the indicators is summarized in terms of its advantages and disadvantages for NNE application. In addition, key data gaps and research themes are identified.

Surface Water Dissolved Oxygen

Among the available biological response indicators of eutrophication, surface water dissolved oxygen (DO) best-documented connection to impaired fisheries and benthic populations. The response of aquatic organisms to low DO will depend on the intensity of hypoxia, duration of exposure, and the periodicity and frequency of exposure (Rabalais et al. 2002). Many organisms have developed several physiological and behavioral adaptations to deal with temporary periods of low oxygen availability. However, these are all short-term strategies and will not enable the animal to survive long hypoxic periods. If oxygen deficiency persists, death will ensue. Sublethal effects of hypoxia include reduced growth and reproduction (US EPA 2000). Feeding may also decrease when inadequate levels of oxygen are available (Wannamaker and Rice 2000), which would also reduce growth and reproductive output. Hypoxia can affect the behavior of organisms as well. Organisms avoid hypoxic areas that they would normally use for feeding, breeding and shelter (Wannamaker and Rice 2000; Breitburg 2002) leading to effective loss of habitat (US EPA 2003). As DO concentrations drop, so do fish abundance and species diversity (Breitburg 2002).

While surface water DO has the advantage of well-established protocols and relatively inexpensive instrumentation for measurement, it is an inherently ephemeral measurement – varying greatly over time and space. Thus, the frequency in which DO is measured can have profound effects on the conclusions drawn and the subsequent characterization of a system as either healthy or impaired. Summers and Engle (1993) found that single, daytime, instantaneous measures of DO detected hypoxia only 20% of the time that it was known to occur based on 31 days of continuous (15-minute intervals) sampling data. When randomly selected 24-96 hour periods of the continuous data were used, hypoxia was indicated 50% of the time it was detected in the full data set. Hypoxia can be so variable both within and between days that long-term continuous monitoring is needed to detect hypoxia (Summers and Engle 1993). The frequency and duration of sampling needed to adequately characterize hypoxia would be expected to vary by estuarine class. Some thought must also be given to the spatial variability in the estuary with respect to how it may impact determination of hypoxia.

To implement dissolved oxygen as a management endpoint for California estuaries, there are several key data gaps that must be addressed. These data gaps are described below.

First, long-term continuous data sets (2-5 yrs) must be collected in several index systems representing a range of eutrophication for each of the estuarine classes. These data sets would serve two purposes: 1) assist in defining the “critical condition” for DO measurement—including sampling frequency, duration, locations, etc., and 2) assist in determination of management endpoints (biocriteria) by providing a range of reference conditions. Data sets like this are already known to exist for some California estuaries (e.g. National Estuary Research Reserves in Tijuana River Estuary and Elkhorn Slough, Central Coastal Lagoon Ecosystem Assessment Project, Beck et al. 2007, et al.). It will be important to choose several index systems for each estuarine class that represent a range of climate and physiographic setting. These data will be used to devise a model-monitoring plan for each estuarine class.

Second, work must be undertaken to develop DO criteria for California estuaries, using existing criteria from other estuaries as a starting point. This should be done by compiling a list of sensitive or key species by estuarine class and reviewing existing information about DO requirements to protect all stages of their life cycle.

Table 4-3 A list of potential stressor and biological response indicators by estuarine class.

Estuarine Class	Nutrient Loading	Hypoxia	Plankton Biomass, Speciation	SAV Biomass, Cover, or Density	Macro-algal Biomass	Harmful Algal Blooms
Tidal Lagoon	X	X	X	X	X	*X*
Seasonally-Tidal Lagoon	X	X	X	X	X	*X*
Nontidal Lagoon	X	X	X	X	X	*X*
River Dominated Estuary	X	X	X			
Protected Embayment	X	X	X	X	X	*X*
Open Embayment/Coastal Estuarine Front	*X*	*X*	*X*	*X*		*X*
San Francisco Bay Estuary	X	X	X			*X*

Macroalgal Biomass

Macroalgae thrive in waters that receive nutrient pollution and thus, their blooms are one of the most visible indicators of eutrophication in estuaries. Macroalgae have been associated with a variety of secondary impacts on estuarine invertebrate and fish communities, via the shading and decline of seagrass (Valiela et al. 1992) and reduction of surface water dissolved oxygen (Sfriso et al. 1987). It is also hypothesized that macroalgal blooms can reduce the food sources of estuarine fish and bird communities through direct impacts on benthic macroinvertebrate communities. In addition, algal blooms can affect recreational enjoyment of the aquatic systems by impeding boat progress or by producing toxins or noxious odors that keep visitors away.

Despite the association of macroalgal blooms with impaired aquatic habitat, no clear thresholds of impairment exist, making its use as a management endpoint difficult. First of all, the relationship between macroalgal abundance and surface water dissolved oxygen is complicated. Nezlin et al. (2006) noted a time lag between initial observations of macroalgal proliferation and the onset of hypoxia. This was likely due to the time required for macroalgae to senesce, sink to the bottom, and create an oxygen demand in sediments. Thus, macroalgae seen growing in the intertidal zone in June and July may have contributed to bottom water hypoxia several months later. Nezlin et al. (2006) also noted that macroalgal abundance was not quantitatively related to the frequency of hypoxia. The abundance of macroalgae as determined from aerial photography explained very little of the variability in surface and bottom water hypoxia. The system in which they studied, Upper Newport Bay, is a relatively shallow system (average depth <1 m) with relatively short (~7 days) residence time and significant tidal range (~2 m maximum). Wind driven mixing and tidal mixing may limit the occurrence of hypoxia, even during macroalgal bloom events. For macroalgal impacts on SAV or dissolved oxygen, it is probably preferable to use these latter indicators themselves then macroalgae as management endpoints. Additional research is needed to clarify the relationship between macroalgal abundance, sediment oxygen demand, and surface water dissolved oxygen.

Impacts of macroalgal blooms on benthic macroinvertebrates may be an important endpoint with a clear link to endangered bird and fish species, but additional research is needed to establish the nature and importance of this link and provide indications as to appropriate thresholds.

In general, the measurement of macroalgal abundance as a biological response indicator is somewhat problematic. While field-based methods of measuring of macroalgal biomass are well established in the literature, these techniques are inherently limited in that they cannot provide a synoptic view of algal distribution over comparatively large areas. This is due to the limited number of samples that can be collected and processed during each survey and often-insufficient resources to sample the entire study area. This problem is especially difficult in variable marine environments where the distribution of macroalgal mats is extremely patchy, may drift with the tides and change location daily.

Remote sensing (i.e., aerial or satellite image analysis) shows some promise as an alternative to ground-based methods for assessment of macroalgal extent (Nezlin et al. 2006). Nezlin et al. (2006) used high-resolution color infrared (CIR) aerial imagery to characterize macroalgal abundance in Upper Newport Bay; they found that CIR imagery is a good tool to characterize macroalgal cover, though not biomass. The approach developed in Nezlin et al. (2006) should be applied to other California coastal wetlands to determine the robustness of the methodology among systems and to help further refine the algorithms used to translate the image analysis to macroalgal abundance. In addition to CIR, hyperspectral imaging and high-resolution satellite imagery should also be explored in the future to determine its utility with respect to spatial resolution, ability to resolve macroalgal mat thickness, ability to target tidal phase, and cost.

As with surface water DO mentioned previously, long-term continuous data sets (2-5 yrs) must be collected in several index systems representing a range of eutrophication for each of the estuarine classes. These data sets would serve two purposes: 1) assist in defining the "critical condition" for macroalgal measurement (including sampling frequency, duration, locations, etc.), and 2) assist in determination of management endpoints (numeric criteria) by providing a range of reference conditions.

Plankton Biomass/Speciation

Next to hypoxia, phytoplankton biomass and speciation are among the best-studied indicators of cultural eutrophication. In some estuaries, nutrients cause dense algal blooms to occur for months at a time, blocking sunlight to submerged aquatic vegetation, causing deficits in oxygen and degrading water that can adversely affect the use of estuarine resources, including commercial and recreational fishing, boating, swimming, and tourism. Eutrophication can also result in changes in the dominant species of phytoplankton, thus resulting in impacts in the food chain of pelagic fisheries and aquatic birds. Some species, often indicative of eutrophic conditions, are resistant to normal phytoplankton predators and may therefore be more prone to enter the decomposition pathways which contribute to low dissolved oxygen problems. The consequences of changes in phytoplankton species composition on grazers and predators can be great, but as with other primary producer communities, these are generally not well studied.

Zooplankton is the primary consumer of phytoplankton and bacteria, funneling food energy from phytoplankton production and bacterial decomposition up to higher organisms such as fish. In turn, excretion by zooplankton is one of the most significant recycling mechanisms that supply phytoplankton with nitrogen and phosphorus for growth. Therefore, an evaluation of the zooplankton community is critical to understanding both the fate of phytoplankton production and nutrient recycling. Larval fish survival in spawning areas is dependant upon sufficient densities of appropriate zooplankton species to feed upon. The zooplankton food supply in spawning grounds during spring is one of the critical factors currently being examined in relation to the success or failure of striped bass reproduction. Certain fish such as bay anchovy and silversides remain zooplankton feeders throughout their lives. Still other species, such as menhaden, consume zooplankton as larvae and juveniles and then switch to feeding exclusively on phytoplankton as adults. Thus, knowledge of the species composition and abundance of zooplankton communities is required to assess impacts of an altered phytoplankton community on fisheries resources.

The most widely used measure of phytoplankton biomass is chlorophyll a. Thus, high concentrations of chlorophyll a are indicative of problems related to the overproduction of algae. It has several advantages as a measure of phytoplankton biomass, including: (1) the measurement is relatively simple and direct, (2) it integrates cell types and ages, (2) it accounts to some extent for cell viability, and (4) it can be quantitatively coupled to important optical characteristics of water. However, the concentration of chlorophyll a is an imperfect measure of phytoplankton biomass, as the cellular content of this pigment depends on the composition of the phytoplankton community and ambient environmental conditions.

Chlorophyll a is generally measured by fluorescence and, in the case of remote sensing, by ocean color. A great deal of flexibility exists in the variety of methods used to estimate chlorophyll a, including discrete samples with laboratory-based analysis for precise determination of concentrations, in situ fluorescence probes for continuous monitoring, and remote sensing platforms using a variety of different imagery types of varying spatial and temporal resolution. The OrbView-2/SeaWiFs (Sea-viewing Wide Field-of-view Sensor) instrument is one example of a remote sensing platform that is being routinely used to monitor phytoplankton. Subtle changes in the surface water color result from changes in the concentrations of marine phytoplankton, resuspended sediment, and dissolved substances in the water column. SeaWIFs data may be used to supplement local monitoring data in larger systems to understand processes controlling phytoplankton blooms.

As with surface water DO mentioned previously, long-term continuous data sets of phytoplankton biomass, speciation and zooplankton abundance and speciation (2-5 yrs) must be collected in several index systems representing a range of eutrophication for each of the estuarine classes. These data sets would serve two purposes: 1) assist in defining the "critical condition" for macroalgal measurement (including sampling frequency, duration, locations, etc.), and 2) assist in determination of management endpoints (numeric criteria) by providing a range of reference conditions.

Harmful Algal Blooms

Harmful algal blooms (HABs) are an emerging issue of concern in California. HABs have been found in a variety of different estuarine environments ranging from open embayments and continental shelf waters to freshwater and brackish zones of protected embayments and nontidal lagoons. The frequency, extent, and impact of these blooms are just beginning to be understood. As an indicator, there is a clear linkage between the toxins produced by HABs (e.g. domoic acid) and various estuarine beneficial uses (shellfish poisoning, marine mammal strandings, fish kills, etc.). However, many data gaps exist that must be addressed in order to consider whether HABs are a useful management endpoint for eutrophication.

First, the mechanisms controlling toxin production in both marine and freshwater harmful algal blooms must be better understood. For example, not all Pseudo-nitzschia species are capable of producing DA, and toxic species do not produce DA. 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).

Second, additional research must be conducted in an effort to understand the environmental factors that promote toxic harmful algal blooms. Through previous work, 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; Kudela et al., 2005). The importance of nutrients from anthropogenic versus natural sources (e.g. upwelling) must be better quantified through regional nutrient budgets. In addition, additional research must be conducted to understand what physical factors (upwelling, river discharge, etc.) create conditions suitable for HAB formation.

Submerged Aquatic Vegetation (SAV) Biomass

Many estuarine systems support submerged aquatic vegetation (SAV) if they are shallow enough that if sufficient light penetrates the water column to the seafloor. In areas of low nutrient inputs, populations of seagrasses and perennial macroalgae (including kelp beds) can thrive, with light availability the most important factor controlling their growth (Sand-Jensen and Borum 1991; Dennison et al. 1993; Duarte 1995). As a result, nutrient enrichment rarely causes a shift to phytoplankton or bloom-forming benthic. Fast-growing micro- and macroalgae with rapid nutrient uptake potentials can replace seagrasses as the dominant primary producers in enriched systems (Duarte 1995; Hein et al. 1995). Phytoplankton biomass and total suspended particles increase in nutrient enriched waters and reduce light penetration through the water column to benthic SAV communities. Decreased photosynthetic oxygen production at all light levels also reduces the potential for oxygen translocation and release to the SAV rhizosphere, and creates a positive feedback that reduces sulfide oxidation around the roots, further elevating sediment sulfide levels. Thus chronic sediment hypoxia and high sediment sulfide concentrations have been associated with the decline of the SAV (Robblee et al. 1991). Loss of seagrass coverage increases sediment resuspension and causes an efflux of nutrients from the sediment to the overlying water that can promote algal blooms. Low DO has direct negative impacts on fish and can lead to mortality (Coon 1998). These phenomena have been observed repeatedly in estuaries in California including in a range of seasonal tidal lagoons in the Central Coast (Beck et al. 2007).

Changes in SAV cover and density have significant trophic consequences and clear linkages to estuarine beneficial uses. Reduced seagrass depth distribution or replacement by macroalgal blooms will result in marked changes in the associated fauna (Thayer et al. 1975; Norko and Bonsdorff 1996). Seagrasses also provide food and shelter for a rich and diverse array of fauna including invertebrates and fish. As with other indicators, research is needed to understand the appropriate threshold in the density and/or cover associated with impaired habitat. These thresholds may differ with respect to various species of fish versus benthic infauna (as a source of food for birds). It will also be important to undertake historical studies to understand which estuaries previously supported SAV where it no longer exists.

As an indicator, SAV has the advantage that they are not ephemeral in nature; SAV beds, once established, are easy to tag and monitor repeatedly. In addition, the methods for measuring SAV biomass, density and cover are well established. In small systems, field surveys by scuba or snorkeling are feasible. In larger systems, mapping of SAV density and cover is possible through the use of side-scan sonar. Remote sensing methods have also been used, although at the scale of mapping, they do not provide information on density, and cover is mapped at a much coarser scale. Thus, remote sensing methods may not provide as sensitive an indicator of change of impact from management measures.

As with surface water DO mentioned previously, long-term continuous data sets (2-5 yrs) must be collected in several index systems representing a range of eutrophication for each of the estuarine classes. The Coastal Lagoon Ecological Assessment Project study of five central coastal seasonal tidal lagoons provides an excellent dataset with which to begin to explore these relationships. These data sets would serve two purposes: 1) assist in defining the "critical condition" for SAV measurement (including sampling frequency, duration, locations, etc., and 2) assist in determination of management endpoints (numeric criteria) by providing a range of reference conditions. Additional systems from other parts of the California coastline as well as from other estuarine classes should be added to expand this dataset.

4.3 Developing Numeric Nutrient Endpoints

Section 2 provides an overview of the proposed framework for NNEs in California estuaries. The purpose of this section is to highlight the data gaps and research recommendations relating to the development of numeric nutrient endpoints for California estuaries. These NNEs provide the framework and suggested ranges of numerical values for the three beneficial use risk categories and describe the biological integrity of aquatic communities inhabiting waters of a given designated aquatic use. The NNEs can feed into a regulatory framework for addressing water quality problems by providing a means to assess the biological resources, and associated beneficial uses, at risk from chemical, physical or biological impacts. Some of the important benefits of NNEs also include:

- Diagnostic tools to determine whether impairment to estuarine beneficial uses is occurring;
- A cost-effective method for evaluating impacts from eutrophication with a recommended systematic approach to study design, field methods, and data analysis;
- A basis for characterizing high quality waters and identifying habitats and community components requiring special protection under State anti-degradation policies;
- A framework for deciding CWA Section 319 actions for best management practice control of nonpoint source pollution;
- A scientific basis for refinements in water quality standards (including refinement of use classifications); and
- A process for demonstrating improvements in water quality after implementation of pollution controls.

NNE development requires that there be some targeted condition against which the water body can be compared. Historically, there have been three general approaches to determine targets:

- The use of a statistical evaluation of either reference waterbodies or from all waterbodies (percentile approach (US EPA 2002)) to describe natural or minimally impaired wetland systems with respect to the biological response indicators.
- From expert opinion or scientific literature that document ecosystem thresholds or evaluate the ecological or societal significance of different threshold values for the biological response indicators;
- From established and/or predicted load-response relationships for the indicators that will protect estuarine beneficial uses.

The “Reference Condition” approach involves using a statistical evaluation to describe the frequency distribution of condition in a class of “least disturbed” or “reference” estuaries or in a general population of estuaries that span the range of eutrophication. In the case where minimally-disturbed systems are available, the upper range (e.g. 75%) of a particular biological response indicator is selected and used to define the endpoint. In the case where no reference systems exist, another approach is to calculate the lower percentile (e.g. 25th) of the frequency distribution of the general population of estuaries, then use this selected threshold to define the endpoint for the biological response variable. For many classes of estuaries in California, this approach is not possible because of the lack of data in general, and lack of unimpacted reference systems in particular. When possible, however, this approach can aid by setting criteria by describing the natural potential and best attainable conditions.

The second approach consists of a process by which thresholds are selected from expert opinion or scientific literature documenting historical condition, ecosystem thresholds, or the ecological or societal significance of different threshold values for the biological response indicators. This approach again requires that data or studies exist which document impacts to aquatic life use at varying values for the biological response indicators. The process by which thresholds are selected is usually the result of a stakeholder process, involving regional experts, concerned citizens, regulators and the regulated community.

The third mechanism for determining threshold values for NNEs is through the use of established and/or predicted load-response relationships for the indicators that will protect estuarine beneficial uses. This approach requires that load-response relationships have been developed for the various indicators, either through statistical regression techniques or through the use of dynamic simulation models. This process is also a stakeholder-driven, because a clear threshold in which estuarine beneficial uses are impacted is often not clear.

In California, data available for most estuarine classes are insufficient to use any of these three approaches, with the possible exception of San Francisco Bay. Development of NNEs for the various estuarine classes will require a concerted effort to review existing literature and criteria adopted by other states and to collect new data that will serve as the basis for recommended NNE thresholds between BURC tiers I-III. For some indicators, such as dissolved oxygen, this may be relatively straightforward, as precedent exists for the use of dissolved oxygen for ambient water quality criteria, including estuaries such as Tampa Bay (Janicki et al. 2001), Chesapeake Bay (USEPA 2003), as well as other East Coast estuaries (USEPA 2000). Table 4-3 gives an example of these criteria established for five beneficial uses in Chesapeake Bay. These criteria may serve as a starting point and may be adapted according to the tolerances of aquatic species found in California estuaries.

Table 4-4 Example of dissolved oxygen criteria for specific designated uses: Chesapeake Bay

Designated Use	Criteria	Qualifier	Rationale
Migratory spawning and nursery	• 7-day mean ≥ 6 mg/l • Instantaneous ≥ 5 mg/l • Shallow-water/open-water use criteria	• February-May • February-May • June-January	Protect larval and juvenile freshwater species, shortnose sturgeon
Shallow-water bay grass	• Open-water use criteria		Protected habitat and fish and invertebrates
Open-water fish and shellfish	• 30-day mean ≥ 5 mg/l • 30-day mean ≥ 5.5 mg/l • 7-day mean ≥ 4 mg/l • Instantaneous ≥ 3.2 mg/l	• > 0.5 ppt • 0-0.5 ppt	Ensure survival of larval and juvenile fish and invertebrates; Atlantic and shortnose sturgeon
Deep water seasonal fish and shellfish	• 30-day mean ≥ 3 mg/l • 1-day mean ≥ 2.3 mg/l • Instantaneous ≥ 1.7 mg/l • Open-water use criteria	• June-September • June-September • June-September • October-May	Protect egss and larvae of bay anchovy, crabs, oysters, spot, and flounder
Deep-channel seasonal refuge	• Instantaneous ≥ 1 mg/l • Open-water use criteria	• June-September • October-May	Protect hypoxia-tolerant worms and clams in summer, blue crabs and finfish in winter

4.4 DEVELOPING TMDL TOOLS

Employing NNEs to help set Total Maximum Daily Loads (TMDLs) requires a tool(s) to link biological response indicators to watershed nutrient loads. Generally, these tools come in two forms: 1) simple regression or spreadsheet models and 2) dynamic simulation models. Both kinds of tools have an important role to play in easing the implementation of the California