Numeric Nutrient Endpoint Development for San Francisco Bay Estuary: Literature Review and Data Gaps Analysis

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Technical Report 644 - June 2011

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June 30, 2011

Acknowledgements

The authors of this document wish to thank the members of the San Francisco Bay NNE Technical Advisory Team for their vision and guidance during this process. Special acknowledgment is given to Naomi Feger of the San Francisco Regional Water Quality Control Board, who provided guidance and feedback for use of the San Francisco Bay NNE review in the regulatory context. This report was produced under California State Water Board contract to the Southern California Coastal Water Research Project (Agreement Number 07-110-250).

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This report should be cited as:
McKee, L.J., Sutula, Gilbreath, A.N., Beagle, J., Gluchowski, D., and Hunt, J. 2011 Numeric nutrient endpoint development for San Francisco Bay- Literature review and Data Gaps Analysis. Southern California Coastal Water Research Project Technical Report No. 644. www.sccwrp.org

Executive Summary

The California State Water Resources Control Board (SWRCB) is developing nutrient water quality objectives for the State's surface waters, using an approach known as the Nutrient Numeric Endpoint (NNE) framework. The NNE establishes a suite of numeric endpoints based on the ecological response of an aquatic waterbody to nutrient over-enrichment (eutrophication, e.g. algal biomass, dissolved oxygen). In addition to numeric endpoints for response indicators, the NNE framework must include models that link the response indicators to nutrient loads and other management controls. The NNE framework is intended to serve as numeric *guidanc*e to translate *narrative* water quality objectives. The NNE framework is currently under development for estuaries. Because San Francisco Bay represents California's largest estuary (70% by area of estuarine habitat statewide), it merits development of an estuary-specific NNE framework. The purpose of this document is to review literature and data relevant to the assessment of eutrophication in San Francisco Bay, with the goal of providing information to formulate a work plan to develop NNEs for this estuary. The review had three objectives: 1) Evaluate indicators to assess eutrophication and other adverse effects of anthropogenic nutrient loading in San Francisco Bay, 2) Summarize existing literature in SF Bay using indicators and identify data gaps, and 3) Investigate what data and tools exist to evaluate the trends in nutrient loading to the Bay.

Recommended NNE Indicators for SF Bay

The NNE assessment framework is the structured set of decision rules that helps to classify the waterbody in categories from minimally to very disturbed, in order to determine if a waterbody is meeting beneficial uses, or to establish TMDL numeric targets. Development of an assessment framework begins by choosing response indicators, which were reviewed using four criteria: 1) strong linkage to beneficial uses, 2) well -vetted means of measurement, 3) can model the relationship between the indicator, nutrient loads and other management controls, and 4) has an acceptable signal: noise ratio to assess eutrophication. Indicators varied among four habitat types: 1) unvegetated subtidal, 2) seagrass and brackish SAV, 3) intertidal flats, and 4) tidally muted habitats (e.g. estuarine diked Baylands). Two types of indicators were designated. Primary indicators are those which met all evaluation criteria and would therefore be expected to be a primary line of evidence of the NNE assessment framework for SF Bay. Supporting indicators fell short of meeting evaluation criteria, but may be used as supporting lines of evidence. This terminology is used in order to provide a sense of level of confidence in how the indicators should be employed in a multiple lines of evidence context.

The review found four types of indicators met all evaluation criteria and are designated as primary: dissolved oxygen, phytoplankton biomass, productivity, and assemblage, and cyanobacterial abundance and toxin concentration (all subtidal habitats), macroalgal biomass and cover (intertidal habitat, tidally muted habitats, and seagrass habitats). Other indicators evaluated met three or fewer of the review criteria and designated as supporting indicators: HAB cell counts and toxin concentration, urea and ammonium (all subtidal), light attenuation and epiphyte load (seagrass/brackish SAV). Ultimately, the

real distinction between “primary” and “supporting” and how these classes of indicators would be used as multiple lines of evidence in an NNE assessment is entirely dependent on indicator group and particular applications to specific habitat types. Some primary indicators (e.g. dissolved oxygen) could be stand-alone, while for others such as phytoplankton biomass, productivity and assemblage, the SF Bay Technical Advisory Team strongly recommends using them as multiple lines of evidence, as use of any one alone is likely to be insufficiently robust.

The use of ammonium as an indicator received review, due to its hypothesized role in limiting phytoplankton primary production via nitrate uptake inhibition in Suisun Bay and the lower Sacramento River. The SF Bay Technical Advisory Team chose to include it as a supporting indicator because the importance of ammonium inhibition of diatom blooms relative to other factors controlling primary productivity Bay wide is not well understood. Additional review and synthesis are recommended, pending currently funded studies, to identify potential ammonium thresholds.

To What Extent is SF Bay Demonstrating Symptoms of Eutrophication, Utilizing NNE Indicators?

Of the four habitat types, only unvegetated subtidal habitat had adequate data to make an assessment of eutrophication. Dissolved oxygen in SF Bay subtidal habitat is much higher and phytoplankton biomass and productivity is lower than would be expected in an estuary with such high nutrient enrichment, implying that eutrophication is controlled by processes other than a simple nutrient-limitation of primary production. However, all regions of the SF Bay have experienced significant increases in phytoplankton biomass since the late 1990's. Recent analysis of water quality data collected by USGS from 1978 to 2009 show a significant increase in water column chlorophyll a (30-50% per decade from Suisun to South Bay respectively) and a significant decline in DO concentrations (1.6 to 2.5% per decade in South Bay and Suisun Bay respectively). Thus evidence is building that the historic resilience of SF Bay to the harmful effects of nutrient enrichment is weakening. The causes for the Bay wide trends include changes in water clarity due to less suspended sediment, lower metal inhibition due to improvements in wastewater treatment, increased seeding from ocean populations, declines in consumption by bivalves due to increases in predation by juvenile English sole and speckled sanddabs, and declines in phytoplankton consumption by consumers due to recent new invasive species introductions. Data suggest that primary productivity in Suisun Bay is limited by strong grazing pressure by invasive clams, light limitation by high turbidity, and ammonium inhibition of diatom uptake of nitrate. Few harmful algal blooms (HABs) have been reported recently in SF Bay. However, there have been historical occurrences, and recently cyanobacteria and dinoflagellate blooms have been increasingly documented.

What Are the Nutrient Loads to SF Bay From Various Sources?

Nutrients loads to SF Bay from external sources are poorly understood, though data exist with which to improve published load estimates from some sources. For the most part, published load estimates are outdated by one or even two decades or based on data that were not collected for loads estimation.

Data Gaps and Recommended Next Steps

The SF Bay NNE framework consists of two principle components: 1) primary and supporting indicators used in an assessment framework to assess eutrophication of SF Bay habitats and 2) models that link these indicators back to nutrient loads and other management controls on eutrophication. There are five major recommendations: 1) develop an NNE assessment framework for SF Bay, 2) quantify external nutrients loads, 3) develop a suite of models that link NNE response indicators to nutrient loads and other co-factors, 4) implement a monitoring program to support the use of the NNE in SF Bay to manage nutrients, and 5) Coordinate development of the SF Bay NNE workplan with nutrient management activities in Sacramento and San Joaquin Delta. **The SF Bay Technical Advisory Team assumed the San Francisco Bay Water Board will prioritize these next steps, with review/feedback from its advisory groups.**

Develop an NNE assessment framework for SF Bay

Development of an NNE assessment framework for SF Bay involves specifying how primary and supporting indicators would be used as multiple lines of evidence to diagnose adverse effects of eutrophication. The table below summarizes data gaps and recommended next steps for development of an SF Bay NNE assessment framework by habitat type. Data gaps and recommendations generally fall into four categories: 1) Monitoring to assess baseline levels of indicators of interest where data are currently lacking, 2) Analysis of existing data, 3) Field studies or experiments to collect data required for endpoint development, and 4) Formation of expert workgroups to recommend approach to assessment framework development and synthesize information to be used in setting numeric endpoints.

Type	Indicator	Designation	Data Gaps	Recommended Next Steps
Subtidal Habitat	Dissolved oxygen	Primary	Wealth of data exists. Technical Advisory Team does not have expertise to review adequacy of DO objectives. Review did not address dissolved oxygen data in the tidally muted habitats of SF Bay.	Consider update of science supporting Basin Plan dissolved oxygen objectives, if warranted by additional review by fisheries experts. Review could be for entire Bay or limited to the tidally muted areas of the Bay.
	Phytoplankton biomass , productivity, and assemblage	Primary	Need a review of science supporting selection of endpoints. Improved prediction of factors controlling assemblage	Recommend development of a white paper and a series of expert workshops to develop NNE assessment framework for phytoplankton biomass, productivity, taxonomic composition/assemblages,

Type	Indicator	Designation	Data Gaps	Recommended Next Steps
	HAB species abundance and toxin conc.	Cyanobacteria = primary; Other HAB =supporting	Little data on HAB toxin concentrations in surface waters and faunal tissues.	abundance and/or harmful algal bloom toxin concentrations. Recommend augmentation of current monitoring to include measurement of HAB toxin concentrations in water and faunal tissues.
Subtidal Habitat (Continued)	Ammonium and urea	Supporting	Lack of understanding of importance of ammonia limitation of nitrate uptake in diatoms on Bay productivity vis-à-vis other factors. Lack of data on urea in SF Bay	Recommend formulation of a working group of SF Bay scientists to synthesize available data on factors known to control primary productivity in different regions in the Bay, and evaluate potential ammonium endpoints. Recommend collecting additional data on urea concentrations in SF Bay via USGS's water quality sampling over a two year period.
	Macrobenthos taxonomy, abundance and biomass	Co-factor	Lack of information on how to use combination of taxonomy, abundance, and biomass to assess eutrophication	Recommend utilization of IE-EMP dataset to explore use of macrobenthos to be used reliably to diagnose eutrophication distinctly from other stressors in oligohaline habitats. This may involve including biomass in the protocol to improve ability to diagnose eutrophication.
Seagrass Habitat	Phytoplankton biomass, epiphyte load and light attenuation	Phytoplankton biomass = primary, epiphyte load and light attenuation = secondary	Poor data availability of data on stressors to SF Bay seagrass beds. Studies needed to establish light requirements for seagrass and to assess effects of light attenuation	Recommend 1) Continued monitoring of aerial extent of seagrass every 3-5 years (currently no further system scale monitoring is planned beyond 2010), 2) studies to establish light requirements for SF Bay seagrass species, 3) development of a statewide workgroup to develop an assessment framework for seagrass based on phytoplankton biomass, macroalgae, and epiphyte load and 4) collection of baseline data to characterize prevalence of macroalgal blooms on seagrass beds. Studies characterizing thresholds of adverse effects of macroalgae on seagrass currently underway in other California estuaries should be evaluated for their applicability to SF Bay.
	Macroalgae biomass and cover	Primary	Data gaps include studies to establish thresholds of macroalgal biomass, cover and duration that adversely affect seagrass habitat	
Intertidal Flat Habitat	Macroalgal biomass and cover	Primary	Lack of baseline data on frequency, magnitude (biomass and cover) and duration of macroalgal blooms in these intertidal flats	Recommend collection of baseline data on macroalgae, microphytobenthos and sediment bulk characteristics. Recommend inclusion of SF Bay scientists and stakeholders on statewide workgroup to develop an assessment framework for macroalgae on intertidal flats.
	Sediment nutrients	Supporting		
	MPB taxonomy and biomass	Supporting		

Type	Indicator	Designation	Data Gaps	Recommended Next Steps
Muted Subtidal Habitat	Macroalgae	Primary	Lack of baseline data on biomass and cover in muted habitat types	Recommend collection of baseline data on macroalgae, dissolved oxygen, phytoplankton biomass, taxonomic composition and HAB species/toxin concentration in these habitat types. Recommendation to develop an assessment framework based on macroalgae, phytoplankton and dissolved oxygen in these habitat types. One component of this discussion should be a decision on beneficial uses that would be targeted for protection and to what extent the level of protection or expectation for this habitat type differ from adjacent subtidal habitat.
	Phytoplankton biomass, assemblage, HAB toxin conc.	Phytoplankton biomass, cyanobacteria = primary; assemblage and other HABs= supporting	Lack of baseline data on biomass and community composition, HAB toxin concentrations	
	Dissolved oxygen	Primary	Some data on dissolved oxygen exist. Unclear what levels of DO required to protect muted habitat beneficial uses	

Quantify Nutrient Loads

The table below provides a summary of data gaps and recommended next steps. Recommendations generally fall into two categories: 1) Revising and updating estimates of nutrients from the different sources, based on existing data and 2) Identification of data needed to develop a dynamic loading model.

Source	Data Gaps Identified	Recommended Next Steps
Atmospheric Deposition	No recently published data on wet & dry atmospheric deposition	Loads likely relatively small. Literature review to determine range of N and P deposition rates for West Coast coastal urban areas. Recommend baseline atmospheric deposition monitoring of wet and dry N and P deposition over 1-2 year period to better constrain estimates.
Terrestrial Loads from Delta	Dry weather concentrations available through RMP. No data available on wet weather concentrations	Loads likely large. Recommend analysis of existing RMP data to estimate dry weather nutrient loads. Initiate wet weather data collection of nutrients at the Mallard Island DWR sampling location (head of Suisun Bay) to support improved daily loads estimates for 1995-present.
Municipal Effluent	Data available through 15 of approx. 40 Publicly Owned Treatment Works	Loads likely large. Synthesize nutrient discharge and concentration data to estimate loads over period of last 10-20 years. Encourage all treatment plants that discharge to the Bay to begin analyzing effluent for total and dissolved inorganic nutrients and to submit these data to the SFRWQCB on a regular basis. Recommend that the POTWs conduct a laboratory inter-comparison on nutrient methods to assure comparability of estimates.
Industrial Effluent	Some data available from the 1990s	Loads likely small relative to municipal wastewater. Synthesize available data to provide information for prioritization of any future steps.
Stormwater	Lack of wet weather data sufficient to develop a dynamic loading model	Loads likely large. Synthesize data to provide an updated estimate of stormwater contributions to assist prioritization of next steps. Scope the data needs associated with the development of a dynamic loading model.
Groundwater	Data available from 79 USGS monitoring stations. Flow data not well understood	Loads likely small. Refine current loads estimates after review by local USGS groundwater experts in order to support prioritization of next steps if any.
Exchange with Coastal Ocean	Some data available for fluxes of water and sediments during selected tides and seasons	Initiate a workgroup of local experts to design a sampling program for nutrient flux at the Golden Gate boundary. The intent with this program would be to develop models that simulate flux at the ocean-bay interface.

Develop Load-Response Models

An important component of implementing the NNE framework in SF Bay is the development of load-response models that can simulate the ecological response of the Estuary to nutrients and other important co-factors. Several types of models need to be developed, fitting into two general categories: 1) Air, oceanic and watershed loading model(s), which estimate the amount of nutrients and sediment reaching the SF Bay estuary and where they originate, and 2) an Estuary water quality model, which simulates the ecosystem response to nutrient loads and other management controls. Sufficient data and knowledge of SF Bay must exist to support the development of system wide dynamic simulation models to predict phytoplankton biomass/community response and relationships to models of secondary productivity. This is not likely in the short term, so it is important to consider that the development of a more complex model should follow the testing out of key concepts and assumptions in smaller, simpler models.

Scoping the development of these NNE load response models should begin through use of empirical data and studies to develop coarse nutrient budgets for SF Bay. Existing data that describe the timing and magnitude of external sources, internal sources, sinks, and pathways of transformation such as benthic nutrient flux, nitrification, denitrification, etc. would be compiled in order to synthesize current understanding of sources and fate of nutrients as well as identify critical data gaps in advance of the modeling strategy development.

Second, a review of existing models and their applications should be undertaken, with the intent of understanding what existing tools may be used to leverage efforts.

During this strategy workshop, participants would describe the modeling objectives, determine whether existing tools can be used in this effort, identify key data gaps and studies, and identify additional work elements needed to begin this major work element. The product of this effort would be the identification of the appropriate models, a phased workplan, timeline and budget to develop these models, and identification of and coordination among key institutions, programs and stakeholders. This information could be synthesized into a workplan to develop the loading and estuary water quality models and a preliminary timeline and budget for Phase I of the effort.

Conduct a Monitoring Program to Develop and Implement the NNE Framework in SF Bay

The development and use of an NNE framework for San Francisco Bay is completely contingent on the continued availability of monitoring data to formulate, test and periodically assess the status of the Bay with respect to eutrophication. Over the past forty years, the USGS has conducted a research program in the subtidal habitat of SF Bay, with partial support by the SF Bay Regional Monitoring Program (RMP) since 1993. This USGS research program cannot be considered replacement for a regularly funded monitoring program. The SF Bay Technical Advisory Team strongly recommends that a nutrients/eutrophication monitoring strategy be developed and funded for successful development and implementation of the NNE in SF Bay.

Coordinate Development of the SF Bay NNE Framework with Nutrient Management in the Delta

Development and implementation of a NNE framework for SF Bay will require improve coordination with nutrient management activities in the San Joaquin and Sacramento River Delta. Preliminary discussions on this topic have just begun with the Central Valley Water Board staff. Other entities, for example, the Interagency Ecological Program should be engaged. Coordination should be improved, at minimum, with respect to any future monitoring and/or modeling of nutrient loading, transport and source identification, as SF Bay and the Delta exchange nutrients across their aquatic and terrestrial boundaries. Coordination would be further enhanced by a similar review of NNE candidate indicators, summary of existing science, and identification of data gaps and recommended next steps specifically for the Delta.

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1. Introduction

1.1 Background

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 (HABs), hypoxia, and impacts on aquatic food webs (Valiela, Foreman et al.,1992; Kamer and Stein, 2003). These ecological impacts of eutrophication of coastal areas can have far-reaching consequences, including fish-kills and lowered fishery production (Glasgow and Burkholder, 2000), loss or degradation of seagrass and kelp beds (Twilley, 1985; Burkholder, Noga et al., 1992; McGlathery, 2001), smothering of bivalves and other benthic organisms (Rabalais and Harper, 1992), nuisance odors, and impacts on human and marine mammal health from increased frequency and extent of HABs and poor water quality (Bates et al., 1989; Bates, DeFreitas et al., 1991; Trainer, Hickey et al., 2002). These modifications have significant economic and social costs, some of which can be readily identified and valued, while others are more difficult to assess (Turner, Qureshi et al., 1998). According to United States Environmental Protection Agency (USEPA), eutrophication is one of the top three leading causes of impairments of the nation's waters (USEPA, 2001).

In California, the impacts of nutrient loading on estuaries and coastal waters have not been well monitored (Bricker, Clement et al., 1999), with the notable exception of San Francisco (SF) Bay where there has been research and ongoing publication by a number of authors (Cloern, 1982; Cloern, Cole et al., 1985; Cloern, 1991; Cloern, 1996; Cloern, 1999). Without management actions to reduce anthropogenic nutrient loads and other factors controlling eutrophication, symptoms are expected to develop or worsen in the majority of systems, in part due to projected population increases along the coastal areas. Scientifically-based statewide water quality objectives and tools that relate these objectives to management controls are needed to prevent eutrophication from occurring and to provide targets for restoration or mitigation of systems where adverse effects of eutrophication have already occurred.

The USEPA initiated the National Nutrient Management Strategy in 1998 to begin addressing the pervasive impacts of excessive nutrient loading to both fresh and marine waters (Wayland, 1998). A primary goal of the strategy was to develop numeric nutrient criteria to measure the progress of the management strategy. The USEPA issued a series of technical guidance manuals for the development of nutrient criteria. Initial national guidance on nutrient criteria development advocated the use of a statistical approach to establish thresholds based on the nutrient concentrations in surface waters (USEPA, 1998). In this approach, reference conditions were based on 25th percentiles of all nutrient concentration data including a comparison of reference condition for the aggregate ecoregion versus the subecoregions. These 25th percentile concentrations were characterized as criteria recommendations that could be used to protect waters against nutrient over-enrichment. The "Nutrient Criteria Technical guidance Manual: Estuarine and Coastal Waters" was released by USEPA in 2001.

Several studies have demonstrated the shortcomings of using ambient nutrient concentration criteria alone to predict eutrophication, in streams (Welch, Horner et al., 1989; Fevold, 1998; Chetelat, Pick et al., 1999; Heiskary and Markus, 2001; Dodds, Smith et al., 2002) and estuaries (Cloern, 2001; Dettman, Kohn et al., 2001; Kennison, Kamer et al., 2003). Use of ambient, surface water nutrient concentrations is generally not effective for assessing eutrophication and the subsequent impact on beneficial use because ambient concentrations reflect the biological processing that has already occurred. In addition, biological response to nutrients (e.g., algal productivity) depends on a variety of mitigating factors such as basin morphology and substrate characteristics, tidal energy, stratification, temperature, light availability, biological community structure, and seed populations. Thus high concentrations are not an obligatory indicator of eutrophication and low concentrations do not necessarily indicate absence of eutrophication.

Given these problems, in 1999 the USEPA Region 9 and the California State Water Resources Control Board (SWRCB) chose an alternative approach to developing nutrient objectives (USEPA, 2006). This approach, known as the Nutrient Numeric Endpoint (NNE) framework, establishes a suite of numeric endpoints based on the ecological response of an aquatic waterbody to nutrient over-enrichment (eutrophication, e.g., algal biomass, dissolved oxygen). It was suggested that numeric endpoints, if successfully developed, would serve as *guidanc*e to translate *narrative* water quality objectives (State of California's term for water quality criteria) for nutrients and biostimulatory substances. A key component of the NNE framework is the availability or development of stressor- response tools that link the ecological response indicators with nutrient loads and other potential management controls for TMDL development and implementation.

The California NNE framework was first developed for streams and lakes (USEPA, 2006) and is currently under development for estuaries. A scientific framework has been presented to support the development of numeric endpoints for a suite of biological response indicators and highlight data gaps and research recommendations for their development (USEPA, 2007). A subsequent document articulated a broad work plan to address data gaps, develop numeric endpoints, and support the efficient and cost-effective development of stressor-response TMDL tools (USEPA, 2008). Within this work plan, one key step was to summarize existing literature relevant to the development of a set of NNEs and TMDL tools in relation to monitoring and assessment of eutrophication in SF Bay estuary. A key outcome of this initial step is a work plan vetted by the scientists and stakeholders that work and live around the estuary.

1.2 Objective, Geographic Scope and Organization of this Report

The purpose of this document is to present the review of literature and monitoring programs relevant to the assessment of eutrophication in SF Bay, with the goal of providing a baseline of available information to formulate a work plan to develop NNEs for this estuary. The review had four specific objectives:

- Evaluate appropriate indicators to assess eutrophication in SF Bay;
- Summarize existing literature and identify data gaps on the status of eutrophication in SF Bay with respect to these indicators;

- Describe what data and tools exist to evaluate the trends in nutrient loading to the Bay; and
- Summarize, to the extent possible (What do they reveal about trends in nutrient loads over time?)

For the purposes of this literature review, the geographic scope of this effort is limited to the areas of the Bay included within the San Francisco Bay Regional Water Quality Control Board (SF Water Board) jurisdiction (Figure 1). The upstream boundary of the SF Water Board is roughly coincident with the 2 ppt isohaline of bottom waters (a.k.a. "X2", Jassby et al., 1994). This X2 isohaline has a significant statistical relationship with measures of SF estuary resources, including: 1) supply of phytoplankton and phytoplankton-derived detritus, 2) benthic macroinvertebrate, 3) larval fish survival, and the abundance of fish.

The intention is this will be a living document, updated over time to reflect input from scientists, stakeholder groups, and the interested public. Drafts will be identified by date of released and should be cited as such. The report is organized into six sections:

Section 1 gives the introduction, purpose of the document, the organization, and definitions of key terms used throughout the report.

Section 2 gives a brief summary of the conceptual framework of the NNE, preliminary classification and consideration of habitat types, and candidate estuarine NNE (E-NNE) indicators.

Section 3 provides an overview of relevant physiographic information for the Bay Area including human population trends, climate, habitats (both in Bay and fringing), beneficial uses and water quality criteria designated by the State of California through the San Francisco Regional Water Quality Control Board (hereto referred to as the "SF Water Board").

Section 4 provides a review of the current understanding of external nutrient loads and ambient nutrient concentrations in SF Bay.

Section 5 reviews and summarizes existing information on candidate NNE indicators for the SF Bay estuary. The section focuses on seven main indicator groups: phytoplankton blooms and HAB species, dissolved oxygen (Hypoxia and anoxia), macroalgae, submerged aquatic vegetation (sea grass and brackish submerged aquatic vegetation), benthic macroinvertebrates, jellyfish, and ammonium including ammonium nitrate ratio, urea, and toxicity.

Section 6 summarizes the review, identifies important data gaps and recommends next steps.

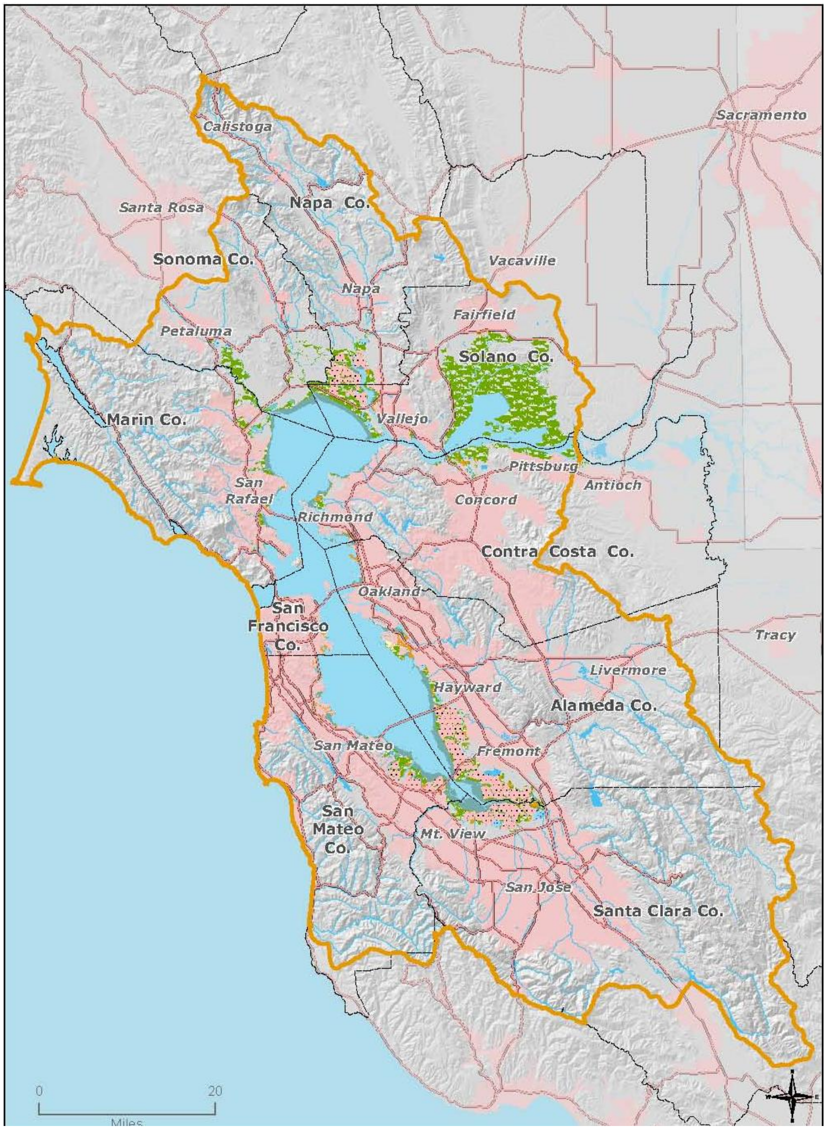

Figure 1.1. Geographic scope of the literature review, defined by SF Water Board jurisdiction.

1.3 Important Definitions

For those outside the regulatory world, distinction between terms like “criteria,” “standards”, “objectives,” and “endpoints” can be confusing. The purpose of this section is to provide definitions of the terms that are linked closely to how the NNE framework will be implemented.

<u>Eutrophication</u>: Eutrophication is defined as the acceleration of the delivery, in situ production of organic matter, and accumulation of organic matter (Nixon, 1995). One main cause of eutrophication in estuaries is nutrient over enrichment (nitrogen, phosphorus and silica). However, other factors influence primary producer growth and the build-up of nutrient concentrations, and hence modify (or buffer) the response of a system to increased nutrient loads (hereto referred to as **co-factors**). These co-factors include hydrologic residence times, mixing characteristics, water temperature, light climate, grazing pressure and, in some cases, coastal upwelling.

***Indicator*:** A characteristic of an ecosystem that is related to, or derived from, a measure of biotic or abiotic variable, that can provide quantitative information on ecological condition, structure and/or function. With respect to the water quality objectives, indicators are the ecological parameters for which narrative or numeric objectives are developed.

Water Quality Standards: Water quality standards are the foundation of the water quality-based control program mandated by the Clean Water Act. Water Quality Standards define the goals for a waterbody by designating its uses, setting criteria to protect those uses, and establishing provisions to protect water quality from pollutants. A water quality standard consists of three basic elements:

1. **Designated uses** of the water body (e.g., recreation, water supply, aquatic life, agriculture)
2. **Water quality criteria** to protect designated uses (numeric pollutant concentrations and narrative requirements)
3. A**ntidegradation policy** to maintain and protect existing uses and high quality waters

Water Quality Criteria: Section 303 of the Clean Water Act gives the States and authorized Tribes power to adopt water quality criteria with sufficient coverage of parameters and of adequate stringency to protect designated uses. In adopting criteria, States and Tribes may:

- Adopt the criteria that USEPA publishes under §304(a) of the Clean Water Act;
- Modify the §304(a) criteria to reflect site-specific conditions; or
- Adopt criteria based on other scientifically-defensible methods.

The State of California's water criteria are implemented as "water quality objectives," as defined in the Water Code (of the Porter Cologne Act; for further explanation, see below).

States and Tribes typically adopt both **numeric** and **narrative** criteria. **Numeric** criteria are quantitative. **Narrative** criteria lack specific numeric targets but define a targeted condition that must be achieved.

Section 303(c)(2)(B) of the Clean Water Act requires States and authorized Tribes to adopt numeric criteria for priority toxic pollutants for which the Agency has published §304(a) criteria. In addition to narrative and numeric (chemical-specific) criteria, other types of water quality criteria include:

- Biological criteria: a description of the desired biological condition of the aquatic community, for example, based on the numbers and kinds of organisms expected to be present in a water body.
- Nutrient criteria: a means to protect against nutrient over-enrichment and cultural eutrophication.

- Sediment criteria: a description of conditions that will avoid adverse effects of contaminated and uncontaminated sediments.

Water Quality Objectives: The Water Code (Porter-Cologne Act) provides that each Regional Water Quality Control Board shall establish water quality objectives for the waters of the state i.e., (ground and surface waters) which, in the Regional Board's judgment, are necessary for the reasonable protection of beneficial uses and for the prevention of nuisance. The State of California typically adopts both **numeric** and **narrative** objectives. **Numeric** objectives are quantitative. **Narrative** objectives present general descriptions of water quality that must be attained through pollutant control measures. Narrative objectives are also often a basis for the development of numerical objectives.

Numeric Endpoint: Within the context of the NNE framework, numeric endpoints are thresholds that define the magnitude of an indicator that is considered protective of ecological health. These numeric endpoints serve as guidance to Regional Boards in translating narrative nutrient or biostimulatory substance water quality objectives. They are called “numeric endpoints” rather than “numeric objectives” to distinguish the difference with respect to SWRCB policy. Objectives are promulgated through a public process and incorporated into basin plans. Numeric endpoints are guidance that can evolve over time without the need to go through a formal standards development process.

2. NNE Conceptual Approach, Classification, and Key Indicators

This section describes the NNE conceptual approach, estuarine classification and key habitat types and the rationale for selection of candidate NNE indicators identified for SF Bay estuary. The material in this section is derived from Sutula et al. (2011), which conducted an extensive review of candidate NNE indicators for California estuaries.

2.1 NNE Conceptual Approach

The Nutrient Numeric Endpoints (NNE) framework is a term coined to describe the SWRCB staff strategy for developing nutrient objectives for the State of California. This draft strategy includes developing a narrative objective, plus numeric guidance that would be incorporated by default into the Basin Plans of the Regional Water Quality Control Boards. The purpose of developing NNEs for California estuaries is to provide the State Water Resources Control Board and the Regional Water Quality Control Boards with a scientifically-defensible framework that can serve as guidance for adopting water quality objectives for nutrients.

The development of an NNE assessment framework for SF Bay is consistent with the findings of the review of candidate indicators for California estuaries (Sutula et al., 2011), but this work represents a more focused effort to develop a framework for assessment eutrophication in SF Bay, with the intent to incorporate specific indicators and thresholds into the SF Regional Water Quality Control Board's (hereto referred to as "SF Water Board") Basin plan.

2.1.1 Why Nutrient Concentrations Should Not Be Used to Set Nutrient Water Quality Objectives in Estuaries

Nutrient objectives are scientifically challenging. Nutrients are required to support life, but assessment of how much is "too much" is not straightforward. Typical paradigms used to set thresholds for toxic contaminants do not apply, in part because adverse effects of nutrient over enrichment are visible at orders of magnitude below recognized toxicity thresholds for ammonium and nitrate.

USEPA guidance on nutrient objective development generally recommends three means to set nutrient criteria (USEPA, 2001): 1) reference approach, 2) empirical stress-response approach, and 3) cause-effect approach. The reference waterbody approach involves characterization of the distributions of nutrient in "minimally disturbed" waterbodies. Nutrient concentrations are chosen at some statistical percentile of those reference waterbodies. The empirical stress-response approach involves establishing statistical relationships between the causal or stressor (in this case nutrient concentrations or loads) and the ecological response (changes in algal or aquatic plant biomass or community structure, changes in sediment or water chemistry (e.g., dissolved oxygen, pH). The cause-effect approach involves identifying the ecological responses of concern and mechanistically modeling the linkage back to nutrient loads and other co-factors controlling response (e.g., hydrology, grazers, denitrification, etc.).

SWRCB staff and USEPA Region 9 staff evaluated these three approaches for setting nutrient objectives in California waterbodies and determined that, while it may choose to ultimately incorporate some elements of all approaches into California's strategy for setting nutrient objectives, it would rely most heavily on the cause-effect approach. There were several reasons for this. First, the cause-effect approach has a more direct linkage with beneficial uses and is generally thought to lend itself to a more precise diagnosis of adverse effects. Second, the alternative approaches require a tremendous amount of data not currently available in such a large state. Third, the reference approach is particularly problematic because it automatically relegates a certain percentage of the reference sites to an "impaired" status. In addition, for many waterbody types, minimally disturbed reference sites are largely unavailable. Fourth, statistical stress-response relationships can be spurious, or have lots of unexplained variability (i.e., poor precision). This poor precision is translated to a larger margin of safety required (more conservative limits) for load allocations and permit limits. While waterbody typology, to some degree, can assist in explaining some of this variability, it cannot completely remove the concern. Thus, while simpler than the cause-effect approach, the empirical stress-response approach will result in more false negative and false positive determinations of adverse effects and in the end will be more costly to the public.

For estuaries, reliance on the cause-effect approach is strongly suggested, because in the majority of circumstances, the reference or empirical stress-response approaches are simply untenable (Cloern 2001). Estuaries within California are highly variable in how they respond to nutrient loading due to differences in physiographic setting, salinity regime, frequency and timing of freshwater flows, magnitude of tidal forcing, sediment load, stratification, residence time, denitrification, etc. This combination of "co-factors" results in differences in the dominant primary producer communities (i.e., phytoplankton, macroalgae, benthic algae, submerged aquatic vegetation, emergent macrophytes). It also creates variability in the pathways that control how nutrients cycle within the estuary. At times, these co-factors can play a larger role in mitigating estuarine response to nutrient loads or concentrations, blurring or completely obscuring a simple prediction of primary productivity limited by nutrients (e.g., Figure 2.1). For example, in many lagoonal estuaries, benthic algal blooms can act to reduce surface water concentrations of nutrients to non-detectable levels. Thus while the estuary may be in a clearly impacted state, it would appear to meet N and P ambient water quality objectives. In estuaries such as SF Bay, synthesis by Cloern and Dugdale (2010) have clearly shown that ambient nutrient concentrations do not correlate with measures of primary productivity , in part because of important co-factors that override simple nutrient limitation of primary production.

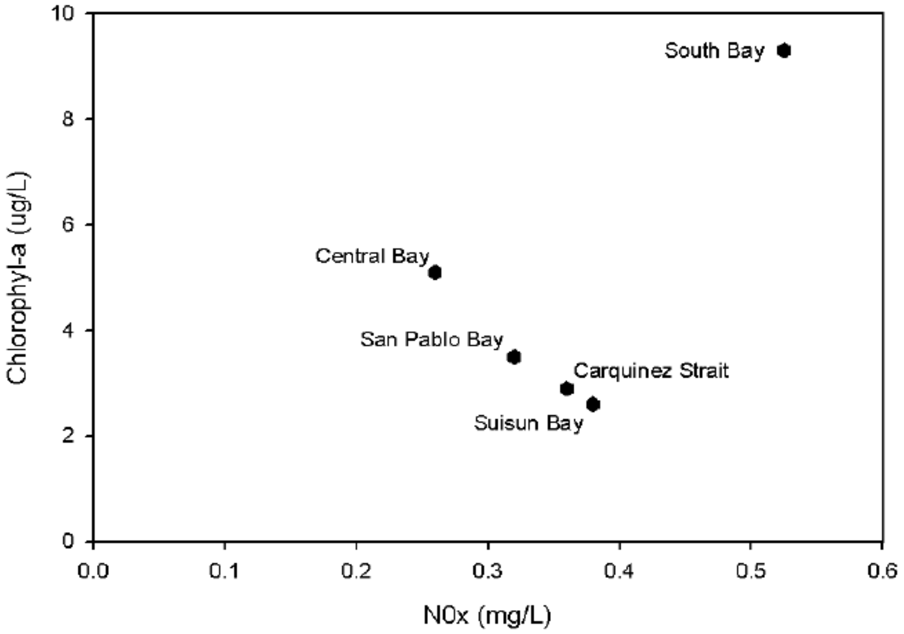

Figure 2.1. Mean chlorophyll a concentrations as a function of mean nitrate+nitrite (NOx) concentrations in SF Bay for the period January 1999 to February 2009 (Data Source: USGS: http://sfbay.wr.usgs.gov/access/wqdata).

2.1.2 Key Tenets of the NNE Approach

The NNE framework for California waterbodies is basely largely on the cause-effect approach. The framework has three organizing principals (Creager, Butcher et al., 2005):

1. *Ecological response indicators provide a more direct risk-based linkage to beneficial uses than nutrient concentrations or loads alone. Thus the NNE framework is based on the diagnosis of eutrophication or other adverse effects and its consequences rather than nutrient over enrichment.*

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

2. *A weight of evidence approach with multiple indicators will produce a more robust assessment of eutrophication.*

When possible, 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 (Karr and Chu, 1999).

3. *Use of models to convert response indicators to site-specific nutrient loads or concentrations.*

A key premise of the NNE framework is the use of models to convert numeric endpoints, based on ecological response indicators, to site- specific nutrient load goals appropriate for assessment, permitting, and TMDLs.

Thus the intent of the NNE framework is to control excess nutrient loads to levels such that the risk or probability of impairing the designated uses is limited to a low level. If the nutrients present – regardless of actual magnitude – have a low probability of impairing uses, then water quality standards can be considered met.

2.2 How Response Indicators Would Be Used: Development of a Diagnostic Assessment Framework

Within the regulatory context, waterbody assessments are made in order to make determination of whether the waterbody is meeting beneficial uses or impaired, as an example, for nutrients. In this context, a diagnostic assessment framework is the structured set of decision rules and guidance for interpretation that helps to classify the waterbody in categories of minimally disturbed (fully sustaining beneficial uses) to moderately disturbed (still sustaining beneficial uses, but functions reduced), to very disturbed (clearly not meeting beneficial uses). Although scientists can provide a lot of guidance and data synthesis to illustrate how the assessment framework could be formed, ultimately the decision of what levels to set thresholds that separate the categories (e.g., minimally versus moderately and very disturbed) is a policy decision. These thresholds are what are referred to as “nutrient numeric endpoints.”

Development of the diagnostic assessment framework begins by choosing indicators that would be measured and used to determine waterbody status. It is important to distinguish between three types of indicators for an NNE assessment framework:

1. Primary indicators
2. Supporting indicators
3. Co-factor indicators required for data interpretation

Primary indicators will play a central role in the NNE assessment framework. Designation of these indicators as “primary” implies a higher level of confidence in these indicators to be used to make an assessment of adverse effects, based on a wealth of experience and knowledge about how this indicator captures and represents ecological response. Primary indicators are those which are considered to meet all explicit criteria (see Section 2.5) established to evaluate candidate NNE indicators.

Supporting indicators are those which could be collected to provide supporting lines of evidence. These indicators may have met many, though not all evaluation criteria, but are consider important because they are commonly used to assess eutrophication in scientific studies, albeit with a lower level of confidence to assess adverse effects of eutrophication. Use of the indicator as supporting evidence over time may increase confidence and cause it to be promoted to "primary."

Finally, co-factors are indicators that could be part of a routine monitoring program and important for data interpretation and trends analysis.

2.3 Context for Indicator Selection: Estuarine Classes and Major Habitat Types

Discussion of estuarine numeric nutrient endpoint (E-NNE) candidate ecological response indicators requires mention of estuarine classes and key habitat types. The approximately 400 estuaries found in the State of California are highly variable in terms of physiographic setting, salinity regime, frequency and timing of freshwater flows, magnitude of tidal forcing, sediment load, stratification, residence time, etc. (Engle et al., 2007). This combination of factors results in differences in the dominant primary producer communities (i.e., phytoplankton, macroalgae, microphytobenthos, submerged aquatic vegetation, emergent macrophytes). It also creates variability in the pathways for nutrient cycling within estuaries. As a result of these differences, estuaries are expected to be variable in how they respond to nutrient loading (NRC, 2000). Partitioning this apparent natural variability into classes will improve the E-NNE framework by eliminating the need to research and define indicators for each of the 400 individual estuaries. Instead, indicators will be defined and tested for each estuarine class (numbering just six).

Classification approaches can by driven by conceptual, empirical or statistical approaches. The NNE Technical Team has proposed a preliminary classification of California estuaries, based on a conceptual approach modeled after the Coastal Marine Ecological Classification Standard (CMECS; Madden et al., 2005; Sutula et al., 2011). The preliminary classes are shown in Table 2.1.

Table 2.1. Preliminary classification of California estuaries.

GEOFORM	SEASONALITY OF SURFACE WATER CONNECTION TO OCEAN
Enclosed Bay	Perennial
Lagoon	Perennial Intermittent Ephemeral
River mouth	Perennial Intermittent

According to this classification, SF Bay estuary is an enclosed bay. However, 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. The central component of the estuary receives very little freshwater input and is greatly influenced by tidal action. 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. Given the size and geomorphic complexity of the estuary, a more detailed review of estuarine classification and dominant habitat types of SF Bay estuary is required in order to understand relevant ecological response indicators (USEPA, 2007).

Within these classes, several key habitat types can be distinguished that organize what indicators may be relevant to consider. For example, Table 2.2 summarizes the relevant aquatic primary producer groups that could be used to diagnose eutrophication, expressed across a range of water depth and salinity regime (Table 2.2.; Day et al., 1989). Thus within each estuarine class, the indicators appropriate to assess eutrophication can change by habitat type.

Table 2.2. Dominant primary producer groups present in California estuaries as a function of water depth and salinity range.

Depth	**Dominant Primary Producers**
Intertidal	Macroalgae Microphytobenthos Seagrass (intertidal Central & No.Calif.)
Shallow subtidal (<10 m)	Macroalgae Microphytobenthos Brackish water SAV and Seagrass Phytoplankton
Deep or light limited subtidal (>=10 m)	Microphytobenthos Phytoplankton Drift or Floating Macroalgae (in oligohaline habitats)

2.4 Conceptual Models and Candidate Ecological Response Indicators

Eutrophication is defined as the acceleration of the delivery, in situ production of organic matter, and accumulation of organic matter within an aquatic ecosystem (Howarth, 1988; Nixon, 1995; Cloern, 2001). One of the main causes of eutrophication in estuaries is nutrient over enrichment (nitrogen, phosphorus and silica). Other factors influence primary producer growth and nutrient availability, and hence modify (or buffer) the response of a system to increased nutrient loads (referred to as **co-factors**). These co-factors include hydrologic residence times, mixing characteristics, water temperature, light climate, grazing pressure and, in some cases, coastal upwelling (Figure 2.1). A simple conceptual model of estuarine ecological response to eutrophication can be described (Figure 2.1). The increased nutrient loads and alterations in co-factors can result in:

1. Changes to aquatic primary producers,
2. Altered water and sediment biogeochemistry, and
3. Altered community structure of secondary (invertebrates) and tertiary consumers (fish, birds, mammals).

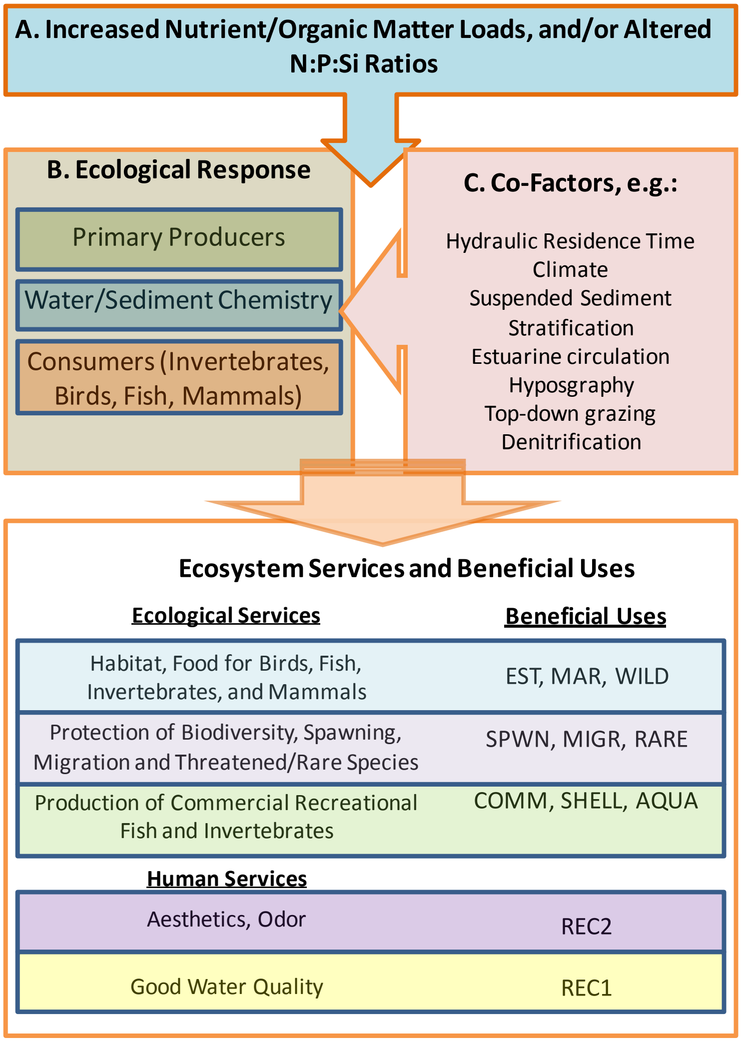

Figure 2.2. Conceptual framework of linkage of nutrient loading (A), ecological response (B), which includes altered primary producers, sediment and water biogeochemistry, and secondary & tertiary consumers), co-factors modulating response (C), and altered ecological services and beneficial uses.

This cascade of effects has a direct effect on the ecosystem services and beneficial uses an estuary provides, including reduced:

- Habitat for aquatic life (including EST, MAR, WILD)
- Protection of biodiversity including rare, threatened and endangered species and migratory and spawning habitat (RARE, SPWN, MIGR)
- Productivity of commercial and recreational fisheries (SHELL, COMM, AQUA).
- Good aesthetics and lack of odors (REC2)
- Maintenance of good water quality (REC1, COMM, AQUA, SHELL)

The three identified components of the ecological response to eutrophication (Figure 2.4 component (B), Figure 2.5) can be used as an organizing framework within which to list and review possible indicators for the E-NNE. Each component is further explained below, along with a list of corresponding indicators under consideration for the E-NNE framework.

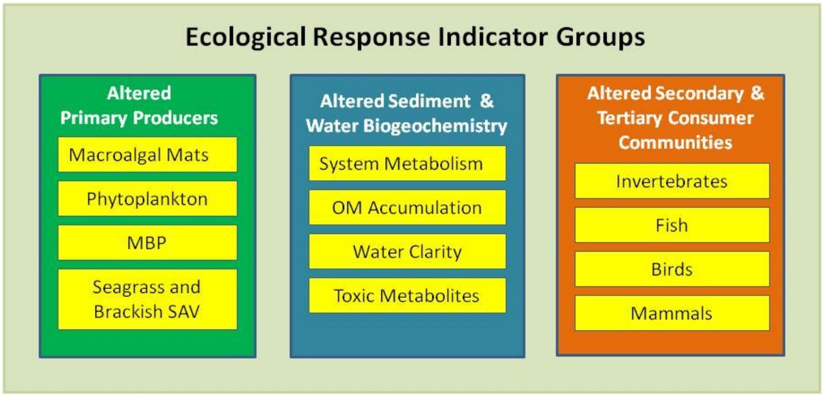

Figure 2.3. Ecological indicator groups, which include altered primary producers, sediment and water biogeochemistry, and secondary & tertiary consumers. MPB = microphytobenthos, OM = sediment organic matter accumulation.

2.4.1 Changes in Aquatic Primary Producer (APP) Community Structure

As an estuary becomes increasing eutrophic, predictable changes occur with respect the types and relative abundance of the primary producer communities, as depicted in Figure 2.6. Estuaries in a “minimally disturbed” condition are typically dominated by primary producers tolerant of low nutrient conditions, such as microphytobenthos (benthic microalgae), seagrasses, or, in deep or turbid estuaries, a high diversity of phytoplankton at relatively low biomass. As nutrient availability increases, the growth of epiphytic micro-, macroalgae as well as opportunistic ephemeral macroalgae is favored in shallow subtidal estuaries. In deep or turbid estuaries, phytoplankton biomass increases, favoring nutrient tolerant and often, HAB species that can produce toxins harmful to marine life and humans (Fong et al., 1993, Valiela et al., 1997, Viaroli et al., 2008). In the extreme end of the eutrophication gradient, macroalgae and cyanobacterial mats dominate intertidal and shallow subtidal habitat, while in deepwater or turbid habitat, cyanobacteria and/or picoplankton blooms can dominate, causing dystrophy.

These changes along a gradient of increasing nutrient availability provide the basis for selecting one or more primary producers as indicators for the E-NNE framework. The precise indicators that will be relevant are dependent on the habitat type and estuarine class. Table 2.3 lists the indicator groups and specific metrics under evaluation for the E-NNE framework. Literature used to evaluate these indicators is summarized in Sutula (2011).

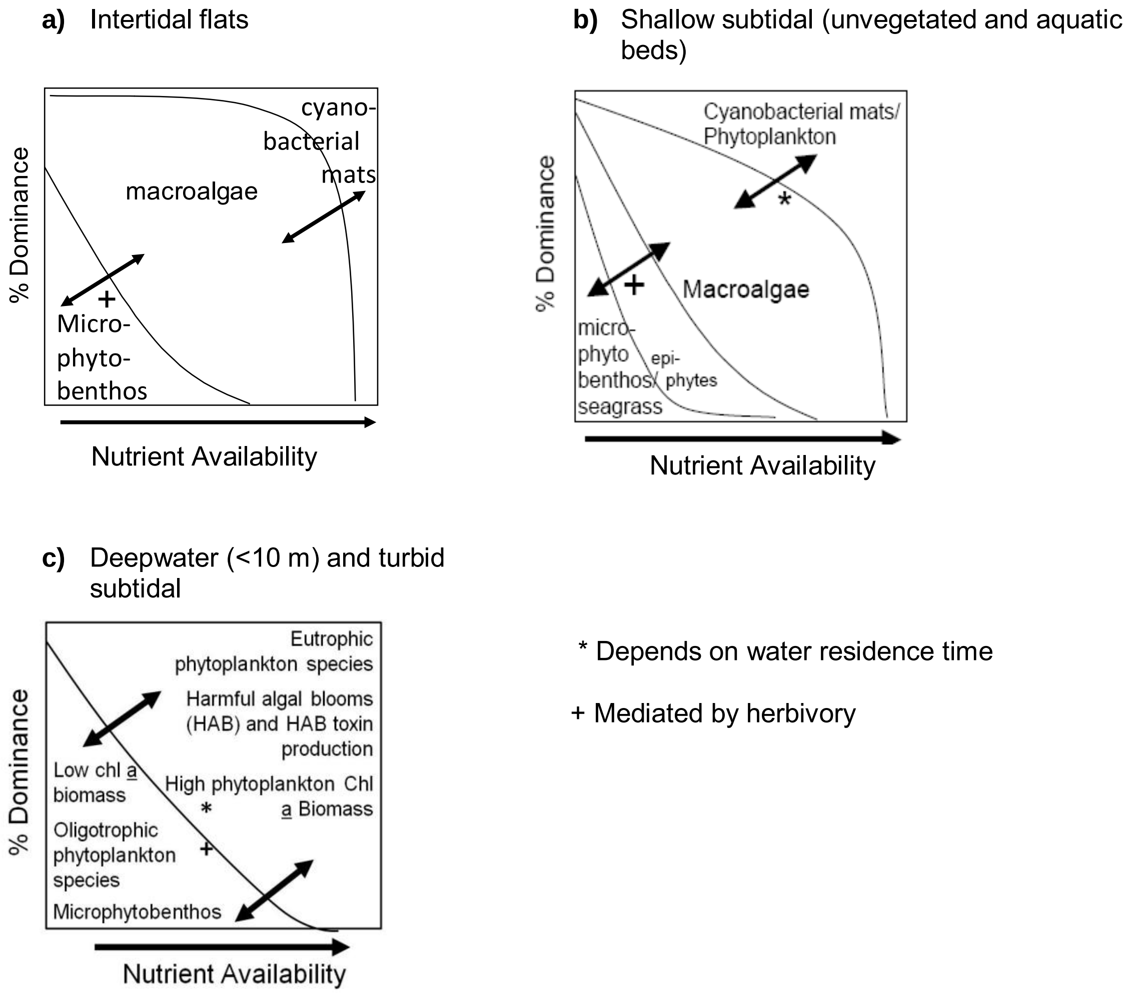

Figure 2.4. Conceptual model of relationship between nutrient availability and relative dominance of primary producers in California estuaries by major habitat type: (a) intertidal flats, (b) shallow subtidal and (c) deepwater or turbid subtidal.

Table 2.3. List of primary producer indicator groups and specific metrics reviewed as candidate indicators for the E-NNE.

Primary Producer Indicator Group	Indicator or Metric
Macroalgae	Percent Cover Biomass
Phytoplankton	Chlorophyll a Concentration (Biomass) Productivity Assemblage/Taxonomic Composition Harmful Algal Bloom Species Abundance Harmful Algal Bloom Species Toxin Concentration
Microphytobenthos	Sediment Chlorophyll a Taxonomic Composition
Seagrass and Brackish Water Submerged Aquatic Vegetation	SAV Aerial Distribution
	SAV Taxonomic Composition
	SAV Biomass SAV Density Epiphyte Load on SAV Macroalgal Biomass/Cover on SAV Water Column Chlorophyll a Water Column Light Attenuation

2.4.2 Altered Water and Sediment Chemistry (Biogeochemical Cycling)

As the process of eutrophication progresses, elevated live and dead aquatic primary producer (APP) biomass provide an elevated supply of labile organic matter, setting off a cascade of altered biogeochemical cycling in the sediments and surface waters. These effects include increased respiration in the sediments and surface waters, increased extent, frequency and duration of hypoxia, and increased concentrations of sediment pore water ammonium, sulfide, increasing the potential for toxicity to benthic organisms (D'Avanzo and Kremer, 1994; Nixon, 1995; Diaz, 2001; Howarth, Sharpley et al., 2002). The efficiency of nitrogen and carbon cycling decreases, which fuels increased organic matter accumulation in the sediments.

With respect to review of candidate E-NNE indicators, changes in biogeochemical cycling in estuarine sediments and surface waters due to eutrophication can be broken down into four general categories (Table 2.4) each having a set of discrete candidate indicators:

- Changes in water clarity, due to increased suspended live and dead biomass
- Altered rates of system metabolism, which capture the relative rates of carbon production and respiration within a system
- Increased sediment organic matter accumulation, which is the rate at which organic matter is accumulates within sediments
- Altered rates of nutrient cycling, which can be defined as the rates of in key transformation mechanisms for nitrogen, phosphorus, and associated elements involved in redox reactions such as sulfur, iron and manganese

Table 2.4. Table of candidate water column and sediment chemistry indicators reviewed for the E-NNE framework (from Sutula, 2011).

Indicator Group	Indicator or Metric
Nutrients	Ammonia Urea N:P Ratio
Water Clarity	Secchi Depth Kd (Light Extinction) Turbidity
Dissolved Oxygen	Dissolved Oxygen Concentration Biological or Chemical Oxygen Demand Sediment Oxygen Demand
Benthic Metabolism	Benthic Production: Respiration Ratio Benthic TCO2 Flux
Organic Matter Accumulation and Sediment Redox Status	Sediment %OC, %N, and %P Sediment C:N: P Ratio
	Sediment TOC:TS and Degree of Pyritization
Nitrogen Cycling	Denitrification Efficiency

2.4.3 Altered Community Composition of Secondary and Tertiary Consumers

Poor habitat quality and altered abundance of primary producers causes shifts in the secondary consumers (benthic infaunal, epifauna and pelagic invertebrates) that are directly impacted by alterations in primary producer community structure and degradation in water and sediment chemistry. Higher level consumers, such as fish, birds, mammals, and other invertebrates that prey upon these secondary consumers (referred to here as tertiary consumers), experience reduced food availability and quality, reduce reproductive success, increased stress and disease, and increased mortality.

While secondary and tertiary consumers are closely linked to ecosystem services and beneficial uses (Figure 2.1), use of these organisms as indicators for the E-NNE framework is problematic because organism and population measures of health are impacted by a variety of different stressors in a complex environment which is not easy to model. Within the group of secondary and tertiary consumers, benthic macroinvertebrates are the sole taxonomic group recommended pursuing for possible inclusion as an E-NNE indicator in some key habitat types and estuarine classes.

Because invertebrates that live in or on sediments are exposed to environmental stressors on an ongoing basis, the benthic life present at a particular location often provides a good indicator of sediment habitat quality. Benthic community composition can be impacted by contamination, eutrophication as well as natural variations in habitat and physical disturbance. The State of California has been developing a benthic response index (BRI) for bays and estuaries with salinities of 18 ppt or greater. Benthic indices apply standard mathematical formulas to data on the number and diversity of benthic organisms at a particular location to find a score that rates the disturbance of the community. This provides a simple means for communicating complex ecological data to environmental managers. The BRI is a component of the SWRCB's sediment quality objectives (www.waterboards.ca.gov/water_issues/programs/bptcp/sediment.shtml), which establishes numeric endpoints for sediment quality due to toxic contaminants.

2.5 Indicator Review Criteria and Candidate NNE Indicators for SF Bay

Sutula (2011) reviewed candidate indicators for use in assessing eutrophication in California estuaries. The following criteria were used in the reviews of existing science to evaluate the utility of each indicator for the E-NNE assessment framework.

Indicators Should:

- Have well documented links to estuarine beneficial uses and, if possible, organisms at multiple trophic levels
- Have a predictive relationship with causal factors such as nutrient concentrations/loads and other factors known to regulate response to eutrophication (hydrology, etc.). This relationship could be empirical (modeled as a statistical relationship between load/concentration and

response or modeled mechanistically through tools such as a simple spreadsheet or dynamic simulation models)

- Have a scientifically sound and practical measurement process that can be accurately and precisely measured over large areas and over multiple years (long term) to quantify the spatial and temporal variability in the forcing and response variables typical of California estuaries
- Must be able to show a trend either towards increasing or/and decreasing eutrophication with an acceptable signal: noise ratio

Based on the review by Sutula (2011) and early discussions with the SF Bay Technical Team, the following indicators were short-listed for further review and synthesis of existing data for the SF Bay estuary (Table 2.5).

Table 2.5. Short-list of candidate E-NNE indicators for SF Estuary by applicable habitat type. Shaded boxes represent applicable habitat.

Indicator	**Habitat Type**			
	Tidal Flats	Shallow Subtidal Unvegetated	Seagrass/Brackish SAV	Deepwater/Turbid Subtidal
Dissolved oxygen		√	√	√
Macroalgae biomass/% cover	√	√	√	
Epiphyte load			√	
Phytoplankton biomass and productivity		√	√	√
Phytoplankton taxonomy, abundance, and/or harmful algal bloom toxin concentrations		√	√	√
Macrobenthos taxonomy/biomass		√		√
Ammonium and urea		√	√	√
Light attenuation		√	√	√

Note that seagrass areal extent and density and macrobenthos taxonomy are known to be affected by a variety of stressors including eutrophication, but cannot be considered to be specific diagnostic indicators of eutrophication (see Sutula, 2011). These indicators would be considered if part of a multimetric assessment protocol for eutrophication, but not as stand-alone indicators.

3. Geographic Setting and Regulatory Context

3.1 Geographic Setting: San Francisco Bay Estuary

The San Francisco Bay estuary (37°27’ - 38°10’ N, 121°45’ - 122°31’ W) lies between the Sacramento-San Joaquin Delta and the Pacific Ocean and receives flow from approximately 160,000 km^2 (37% of California). The “urbanized estuary” (Comomos (ed.), 1979) is surrounded by nine counties with a total resident population of 6.78 million (2000 census) (Figure 3.1) 70.0% of whom reside within watersheds draining to the Bay south of the Richmond San Rafael Bridge (Hwy 580) (the Central and Southern portions of the Bay) within and south of the cities of Larkspur and El Cerrito.

Population increase has not been uniform and build-out has occurred mainly through conversion from agriculture to urban land use. By far the most rapid population growth occurred in the Bay Area during the decades of 1940, 1950 and 1960 largely through medium density residential urban infill adjacent to the Bay (the populations of Contra Costa, Santa Clara and San Mateo increased by 5-6 times and the population of Marin increased by 4 times from the 1940 census to the 1970 census). However more recently (1970 – 2000 census), the largest population increases have been occurring in outlying cities of Napa County (e.g., Calistoga, Napa, and American Canyon more than doubling in population), Solano County (e.g., Suisun City increasing by 9 times), Contra Costa County (e.g., Hercules, Oakley, San Ramon, Brentwood, Clayton averaged together increasing by a staggering 25 times), Alameda (e.g., Pleasanton increasing 3.5 times) and Santa Clara County (e.g., Gilroy and Morgan Hill increasing by over 3 and 6 times respectively). During the more recent decades, urban build out has been through conversion from mainly agricultural land to a mix of medium density urban and lower density suburban residential. It is likely that agricultural and urban lands are continuing to release nutrients that get to the Bay via river and urban stormwater runoff and this release might be exacerbated by disturbances during land use conversion and related construction activities.

The climate in the area is generally mild. Average temperature in the summer ranges from the low to high 60’s, and in the winter between the mid-forties to mid-50’s F (Figure 3.2). Daylight hours in the region range from 9.5 to 15 per day. Available data for 2008 indicates mean hourly solar radiation was 362 Ly/hr in Oakland (Oakland Hills gage), 415 Ly/hr in the Napa area (Carneros gage) and 408 Ly/hr in Santa Clara (Morgan Hill gage) (hourly data, CIMIS, 2008). Peak daily solar radiation occurred during June and July at all three stations.

According to analysis of precipitation data available between 1907 and 1956 from gauges across the Bay Area, mean annual rain directly over the Bay ranges between 14.75 inches (375 mm) in the far South Bay to 28 inches (710 mm) on the western margins of San Pablo Bay in the North (Figure 3.3). In general, rain over the land area of the nine counties adjacent to the Bay is greater than over the Bay itself ranging from about 14 inches (350 mm) near sea level to 48 inches (1,220 mm) on high western facing slopes at higher elevations.

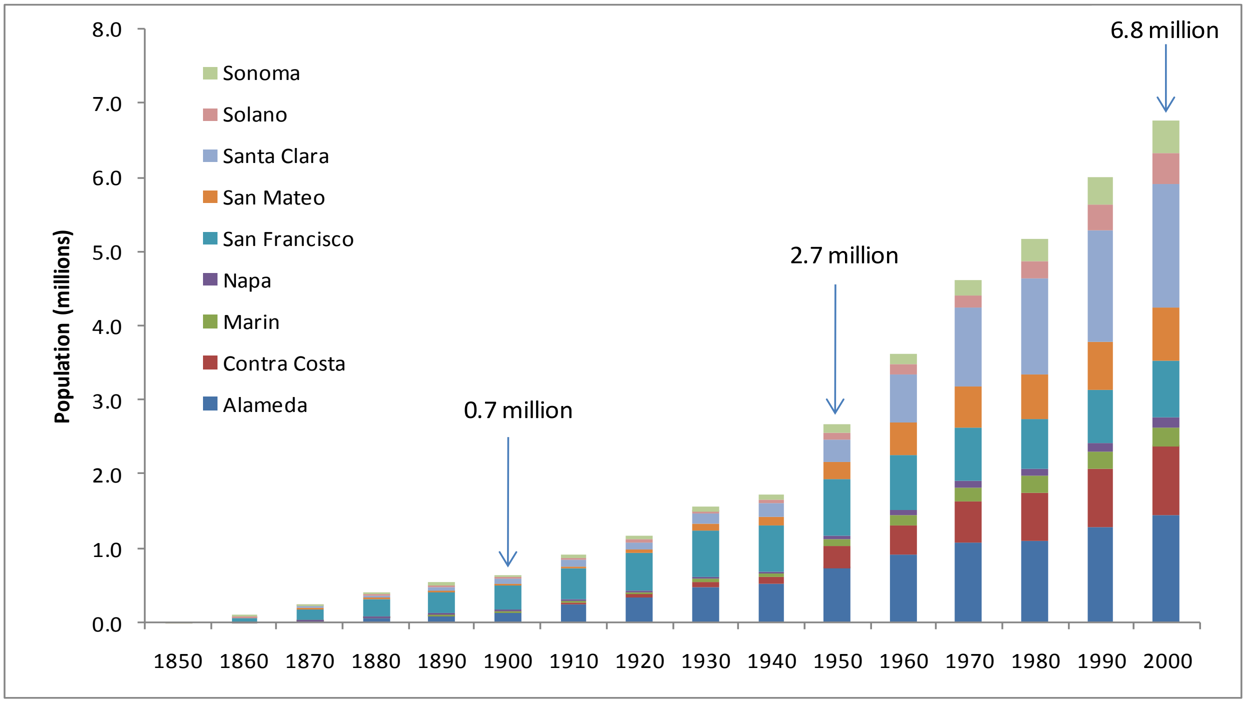

Figure 3.1. Population totals in the nine Bay Area Counties on a decadal time series since 1850. Source: Census Bureau.

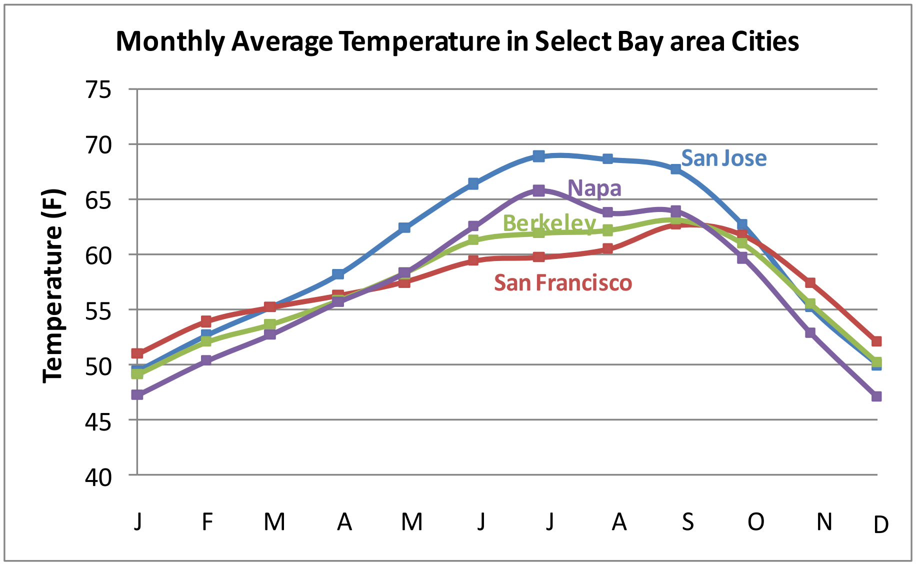

Figure 3.2. The monthly average temperature. Data downloaded from the Western Regional Climate Center.

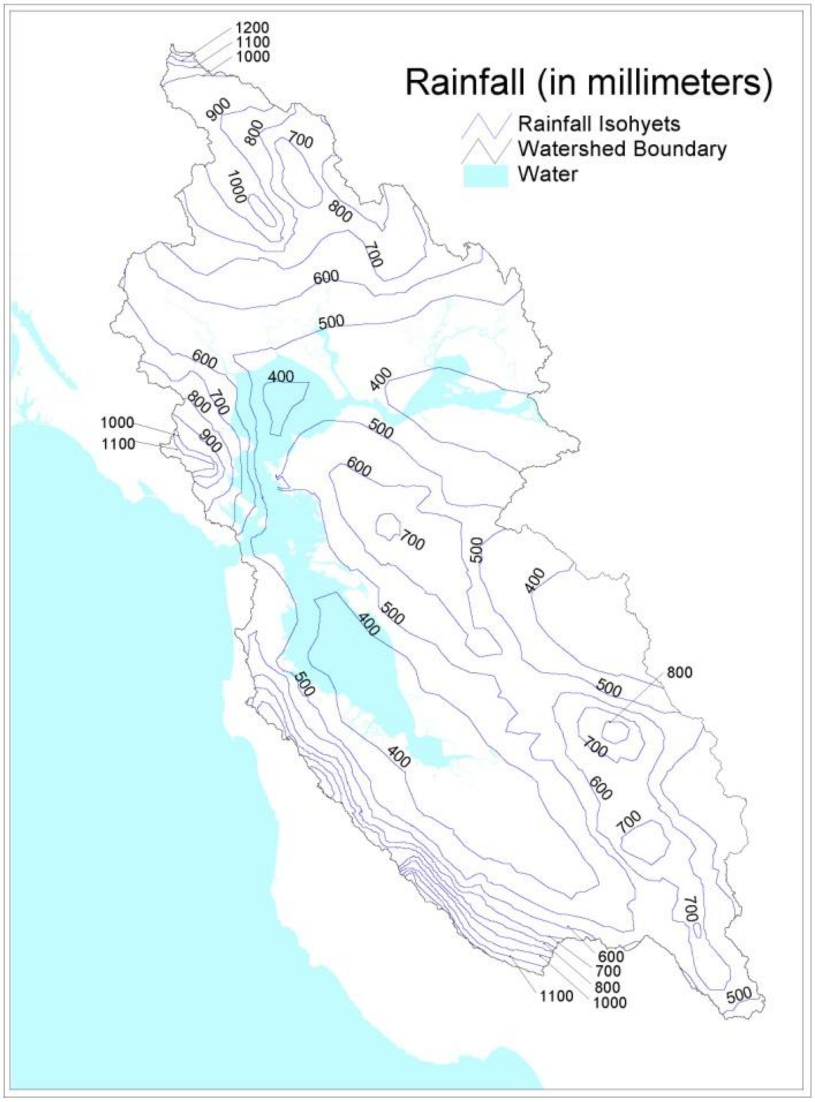

Figure 3.3. Distribution of rainfall (mm) in the Bay Area (after Rantz, 1971).

The estuary itself has an open water surface area of 460 mi^2 (~1200 km^2) at mean sea level (msl) with a maximum depth of 469 ft (143 m) below msl under the Golden Gate Bridge, and an average depth across the estuary of 23 ft (7m) combining to a total msl volume of 8.4 km^3. Tides are mixed semidiurnal with a tidal amplitude (mean high water to mean low water) at San Francisco near the Golden Gate Bridge (NOAA station 18649 established 1854) of 4.1 feet (1.25 m) http://co-ops.nos.noaa.gov/geo.shtml?location=9414290. The latest information from NOAA shows that msl presently rising at a mean rate of 2.01±0.21 mm (0.08±0.008 in) per year. Water temperatures range from about 46-50°F (8-10°C) in the winter to 68-77°F (20-25°C) in the summer. Of interest to both phytoplankton productivity and density gradient driven water fluxes, temperatures at the

Golden Gate are warmer in the winter than in the South or North Bays. In contrast the reverse is true in the summer months (Table 3.1).

Table 3.1. Water temperatures in San Francisco Bay (Source: USGS Surface-Water Data for USA, URL: http://waterdata.usgs.gov/nwis/sw).

Bay Segment	Representative Gage	Winter Mean Low (ºC)	Summer Mean High (ºC)
Lower South Bay	Marker 17	9.5	25
South Bay	San Mateo Br	10	24
Central Bay	Alcatraz	10	19.5
San Pablo Bay	Point San Pablo	9	22
Carquinez Straight	Carquinez	8.5	22.5
Suisun Bay	Benicia Br	8	23

Major components of the freshwater flux into the estuary include precipitation, evaporation, STP effluent influx, river flow and runoff. Smith and Hollibaugh (2006) computed a water budget for the northern and southern segments of the SF Bay for the period 1990-1995 (Figure 3.4). Based on this work it appears that the North SF Bay is overwhelmingly dominated by river inflows and runoff. In contrast, all inputs are important in the South Bay budget (Smith and Hollibaugh, 2006). STP effluent (assumed to be constant) is particularly important in the South Bay, as are evaporative losses in the summertime which sometimes results in net water loss from the South Bay during summer periods. Other important notes include the strong seasonality in runoff between winter and summer months, as well as high inter-annual variability (e.g., 1993 and 1995 are much wetter than the other years). The North and South Bays each exchange water with the Central Bay segment (budget not computed for this segment), which in turn exchanges water with the Pacific Ocean.

San Francisco Bay, like most estuaries, is a complex mix of a variety of habitats which can be conceptually categorized as subtidal, intertidal, and seasonal (fringing) wetlands (locally many of these are diked Baylands) (Figure 3.5). Although, in fact, there is a continuum with multiple subcategories within each, SFEI has mapped the intertidal and diked Baylands. A geographic information system (GIS) geo-referenced map of bathymetry (Figure 3.6) along with substrate character (texture) and habitat types is important for managing and modeling nutrient related water quality, especially, the linkage between nutrient loads and endpoint response. The proportions of habitat and bathymetry vary between Bay segments Table 3.2). The most common habitat is deep-Bay/channel[1] followed by shallow Bay/channel. Historically there were about one third more tidal marshes but this was converted to

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[1] Definitions of habitat type.

Deep Bay/Channel:	Bottom is deeper than 18 ft (5.5 m) below MLLW.
Diked Wetland:	Areas of historical tidal marshes that have been isolated from tidal influence by dikes or levees, but which remain primarily wetland features.
Shallow Bay/Channel:	Bottom is entirely between 18 ft (5.5 m) below Mean Lower Low Water (MLLW) and MLLW.
Tidal Flat:	Occurs from below MLLW to Mean Tide Level (MTL) and supports less than 10% cover of vascular vegetation, other than eelgrass. Includes mudflats, sandflats, and shellflats.
Tidal Marsh:	Vegetated wetland that is subject to tidal action.

either dike wetland or salt pond habitat. Today there is a large effort to restore many of the salt pond areas following the South Bay Salt Pond Restoration 30-year Restoration Plan[2].

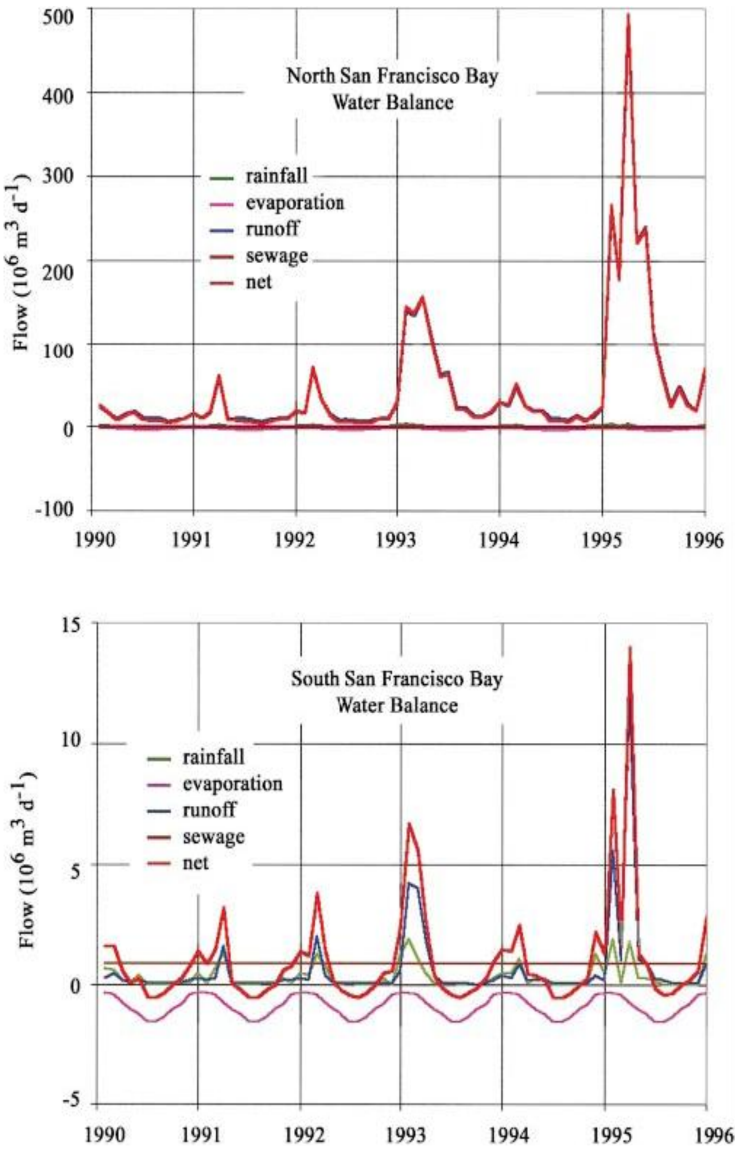

Figure 3.4. Water budget for the northern and southern segments of SF Bay (reproduced without permission from Smith and Hollibaugh, 2006).

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[2] South Bay Salt Pond Restoration official website http://www.southbayrestoration.org/

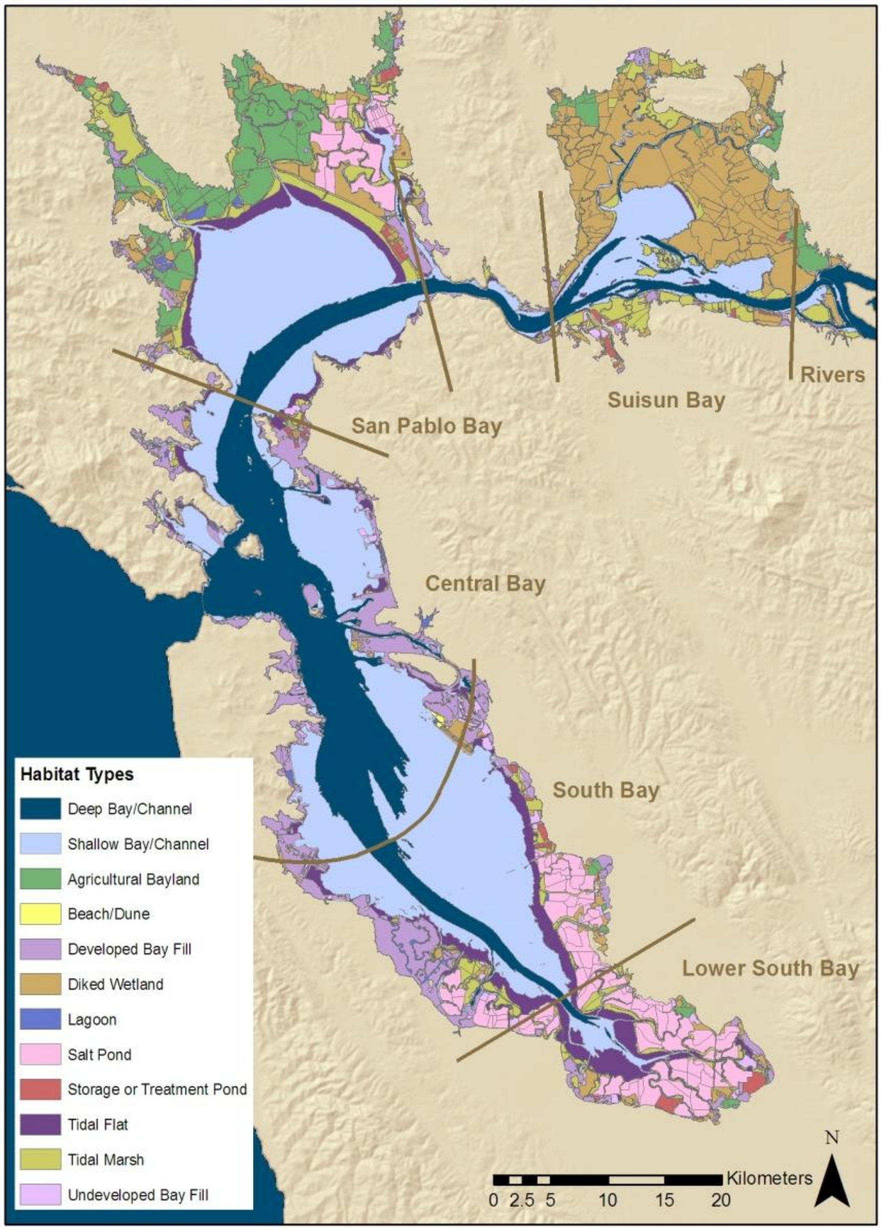

Figure 3.5. Habitat types of SF Bay and adjacent Baylands. Primary sources underlying map data include: CA State Lands Commission, US Geological Survey, US Fish and Wildlife Service, US National Aeronautical and Space Administration, and local experts. Science coordination, GIS and map design by the San Francisco Estuary Institute (1997).

Table 3.2. Relative percent of habitat types by RMP segment for select habitat categories. Diked wetlands are those isolated from tidal influence and dominated by mudflats and marsh. Source: SFEI Ecoatlas: (www.sfei.org/ecoatlas/gis)

	Rivers		**Suisun Bay**		**Carquinez Str.**		**San Pablo Bay**		**Central Bay**		**South Bay**		**Lower South Bay**	
	(km^2)	**(%)**	**(km^2)**	**(%)**	**(km^2)**	**(%)**	**(km^2)**	**(%)**	**(km^2)**	**(%)**	**(km^2)**	**(%)**	**(km^2)**	**(%)**
Tidal Marsh	4.0	19	51	14	5.9	14	61	14	4.5	1.0	20	6.2	19	13
Tidal Flat	0	0	3.7	1.0	2.2	5.5	34	8.1	14	3.1	36	11	28	19
Shallow Bay/Channel	5.6	27	80	22	10	25	212	50	201	45	174	55	13	8.8
Deep Bay/Channel	5.4	26	31	8.5	13	31	39	9.3	216	49	26	8.4	2.4	1.7
Diked Wetland	6.0	28	198	54	5.0	12	35	8.5	5.3	1.2	8.6	2.7	11	7.3
Lagoon	0	0	0	0	0	0	4.0	1.0	2.7	0.6	2.7	0.84	0	0
Salt Pond	0	0	0	0	2.7	6.5	30	7.2	0	0	48	15	68	47
Storage or Treatment Pond	0	0	3.1	0.84	1.9	4.6	4.6	1.1	1.5	0.34	2.3	0.74	5.0	3.4
Total	**21**		**367**		**41**		**420**		**445**		**317**		**146**	

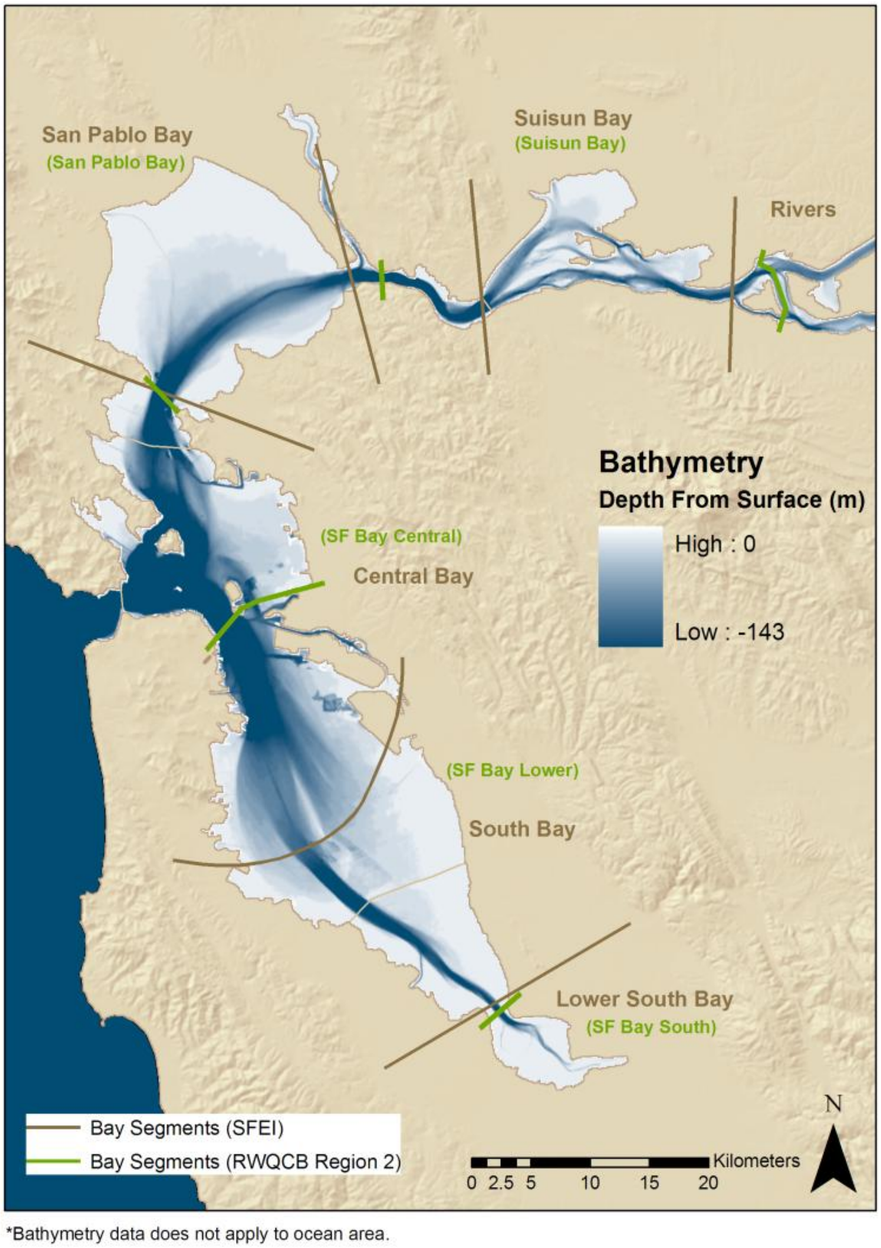

Figure 3.6. Segmentation and bathymetry on SF Bay (Source: NOAA bathymetric soundings). Green lines show the Regional Water Quality control Board (RWQCB) segmentation scheme and the brown lines show the Regional Monitoring Program for Water Quality (RMP) segmentation scheme developed at San Francisco Estuary Institute (SFEI).

Recently, a series of new maps of habitat types have been developed for sub-tidal areas of the Bay that define soft substrate, rocky outcrops, artificial structures, shellfish beds (Olympia oysters-*Ostrea lurida*, California mussels-*Mytilus californianus*, hybridized Bay mussels-*Mytilus trossulus/galloprovinciallis*), SAV (Widgeon grass-*Ruppia maritima*, Eelgrass-*Zostera marina*), and macro algae-insufficient data for map development (*Ulva spp., Gracilaria pacifica, Fucus gardneri*, and introduced *Sargassum muticum*)[3] (NOAA, 2010).

The Regional Water Quality Control Board divides the SF Bay into seven hydrological segments for regulatory purposes based on CalWater, USGS drainage basin maps and largely defined by major bridges. In 2002, a Regional Monitoring Program Work Group developed a new segmentation scheme based on expert opinion of natural hydrological and ecological boundaries, as well as a cluster and partition analyses using 10 years of sediment and water quality data (Grosso and Lowe, 2002; Lowe et al., 2005). The most dramatic difference in the two segmentation schemes is the division between the Central and South Bay segments. The RWQCB makes this division at the Bay Bridge, while the RMP defines this division at the San Bruno Shoal (Figure 3.6).

3.2 San Francisco Bay Beneficial Uses and Existing Water Quality Objectives Relevant to Eutrophication

The SF Water Board has existing standards for SF Bay estuary, consisting of designated beneficial uses, narrative and numeric water quality objectives, and an implementation plan to achieve these standards. The purpose of this section is to summarize the beneficial uses and water quality objectives relevant to the development of NNEs in SF Bay estuary.

3.2.1 Beneficial Uses

Aquatic ecosystems have many resources, services, and qualities that provide the basis for a variety of benefits to the people of the state. Beneficial uses are designated uniquely to aquatic systems based on resources, services, and qualities. The Water Board's definitions for each of these beneficial uses is listed in Table 3.3 below (see RWQCB Basin Plan, 2011 for more information about each beneficial use category). The Water Board is charged with establishing water quality objectives and discharge limits to protect these beneficial uses from pollution and nuisance (RWQCB, 2011). In general most Bay segments have similar designation however there are some exceptions (Table 3.4). For example, the northern reaches of the Bay (Sacramento-San Joaquin Delta) are managed for freshwater and therefore are designated for agricultural, municipal and freshwater uses. The Sacramento-San Joaquin Delta, Suisun Bay, and SF Bay Central reaches are used for industrial process supply and all reaches south of San Pablo Bay (inclusive) are designated for harvesting shellfish (e.g., clams, oysters, and mussels) for human consumption, commercial, or sport purposes.

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[3] Marilyn Latta at the Coastal Conservancy is heading up the Subtidal Goals project for San Francisco Bay.

Table 3.3. Definitions of Beneficial Uses Designated within SF Bay.

OCEAN, COMMERCIAL, AND SPORT FISHING (COMM): Uses 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.

SHELLFISH HARVESTING (SHELL): 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.

ESTUARINE HABITAT (EST): Uses of water that support estuarine ecosystems, including, but 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.

FISH MIGRATION (MIGR): 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.

PRESERVATION OF RARE AND ENDANGERED SPECIES (RARE): 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.

FISH SPAWNING (SPWN): Uses of water that support high quality aquatic habitats suitable for reproduction and early development of fish.

WILDLIFE HABITAT (WILD): 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.

WATER CONTACT RECREATION (REC1): 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.

NONCONTACT WATER RECREATION (REC2): 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.

NAVIGATION (NAV): Uses of water for shipping, travel, or other transportation by private, military, or commercial vessels.

Table 3.4. Designated beneficial uses for segments of SF Bay based on the 2011 basin plan.

	Human Consumptive Uses											
			Aquatic Life Uses						Wildlife Use	Recreational Uses		
	IND	PROC	COMM	SHELL	EST	MIGR	RARE	SPWN	WILD	REC-1	REC-2	NAV
Suisun Bay	E	E	E		E	E	E	E	E	E	E	E
Carquinez Straight	E		E		E	E	E	E	E	E	E	E
San Pablo Bay	E		E	E	E	E	E	E	E	E	E	E
San Francisco Bay Central	E	E	E	E	E	E	E	E	E	E	E	E
San Francisco Bay Lower	E		E	E	E	E	E	P	E	E	E	E
San Francisco Bay South	E		E	E	E	E	E	P	E	E	E	E

*Adapted from Table 2-1 in the Basin Plan (RWQCB, 2007). Segments listed are those defined by the RWQCB. “E” means existing beneficial use.

3.2.2 Existing Water Quality Criteria Related to Nutrients and/or Eutrophication

SF Water Board numeric and narrative objectives relevant for SF Bay are given in Table 3.5. Water quality criteria specifically for nutrients in surface waters are not defined in the Basin Plan.

Table 3.5. Numeric objectives for constituents related to nutrient over enrichment or eutrophication in SF Bay.

Constituent	**Numeric Objectives**
Ammonia	The discharge of wastes shall not cause receiving waters to contain concentrations of un-ionized ammonia in excess of the following limits: **Un-ionized ammonia (mg L^{-1} as N)** Annual Median 0.025 Maximum, Central Bay 0.16 Maximum, Lower Bay 0.4 The intent of this objective is to protect against the chronic toxic effects of ammonia in the receiving waters. An ammonia objective is needed for the following reasons: 1) Ammonia (specifically un-ionized ammonia) is a demonstrated toxicant. Ammonia is generally accepted as one of the principle toxicants in municipal waste discharges. Some industries also discharge significant quantities of ammonia, 2) Exceptions to the effluent toxicity limitations in Chapter 4 of the Plan allow for the discharge of ammonia in toxic amounts. In most instances, ammonia will be diluted or degraded to a nontoxic state fairly rapidly. However, this does not occur in all cases, the South Bay being a notable example. The ammonia limit is recommended in order to preclude any buildup of ammonia in the receiving water, and 3) A more stringent maximum objective is desirable for the northern reach of the Bay for the protection of the migratory corridor running through Central Bay, San Pablo Bay, and upstream reaches.
Dissolved Oxygen	For all tidal waters, the following objectives shall apply: Location Tidal minimum (mg L^{-1}) Downstream of Carquinez Bridge 5.0 Upstream of Carquinez Bridge 7.0 Dissolved oxygen is a general index of the state of the health of receiving waters. Although minimum concentrations of 5 mg L^{-1} and 7 mg L^{-1} are frequently used as objectives to protect fish life, higher concentrations are generally desirable to protect sensitive aquatic forms. In areas unaffected by waste discharges, a level of about 85 % of oxygen saturation exists. A three-month median objective of 80 % of oxygen saturation allows for some degradation from this level, but still requires consistently high oxygen content in the receiving water.

4. Summary of Trends in Nutrient Loading to San Francisco Bay

4.1 Introduction

Development of the NNE framework for the SF Bay requires an accurate understanding of the sources, magnitude and timing of nutrient loads delivered to the Bay. These data are important to properly calibrate our understanding of the biological effects of nutrients on the Bay. It is also important to understand the primary sources and predominant forms of nutrients delivered to the Bay. The purpose of this section is to assess the availability of data and summarize, to the extent possible, the trends in nutrient loading to SF Bay. In most cases it was not possible to find loading information specific to the three major Bay segments (the northern reaches north of the Richmond-San Rafael Bridge, Central Bay between the Richmond-San Rafael Bridge and San Bruno shoals (RMP Central Bay segment) and the southern portions of the Bay south of the San Bruno shoals (RMP south Bay and Lower South Bay segments). Spatial resolution of data overall remains a pervasive gap in current knowledge. In addition, there was generally a lack of understanding of inter-annual variability of nutrient loads. This is of particular concern given that the freshwater inflow to the estuary can vary considerably between dry years and wet years.

4.2 A Primer on Nutrients: Sources and Forms

Nutrients are supplied to SF Bay via a variety of pathways including:

- Atmospheric deposition (both wet and dry) directly to the Bay surface,
- Stormwater from watersheds that drain to the Bay from the nine counties adjacent to the Bay,
- Groundwater from these same tributaries,
- Terrestrial runoff from 37% of the Central Valley via the Sacramento and San Joaquin Rivers,
- Urban wastewater,
- Industrial wastewater, and
- Exchange with coastal ocean (via Golden Gate).

Although each of these pathways is not entirely mutually exclusive (for example atmospheric deposition is probably a large component of urban runoff for some nutrient forms), this section focuses on what passes into the Bay via the main pathways rather than the ultimate source. Should a call for management of nutrient supply to the Bay occur in the future, it will become important to learn more about ultimate sources and the processes that cause the release of and transport of various forms of nutrients into the Bay. In addition, it is important to note that this section focused on “new” sources of nutrients to SF Bay and makes no attempt to account for additional sources or sinks for nutrients within the Bay. As an example, within an estuary, nutrients can undergo a variety of transformations and exchanges among the "compartments" (e.g. water column, sediment, animal and plant biomass, etc.). Nutrients that are deposited to the estuary from a watershed can undergo a series of biological and chemical processes cause the buildup and net release of nutrients (and other compounds) from the sediment pore waters to surface waters in a process known as "benthic flux" (Berner 1980). Net benthic fluxes of nutrients in some estuaries can support a major percentage of primary productivity (e.g. Cowan and Boynton 1996). By the same token, processes such as denitrification can be responsible for the loss of nitrogen from an estuary.

Analytically, nutrients are divided into a number of forms (Table 4.1). Practically, in terms of estimating nutrient loads in relation to standing nutrient concentrations and impacts to beneficial uses in SF Bay, a nutrient budget should primarily focus on total nitrogen and total phosphorus and the main dissolved inorganic species of each. The organic components for nitrogen can then be derived by subtraction using the equation that follows Table 4.1.

Table 4.1. Nutrient species relevant to estimating nutrient loads in relation to standing nutrient concentrations and impacts to beneficial uses in San Francisco Bay.

	Nitrogen	**Phosphorus**
Dissolved Inorganic	Nitrate (NO_3^-) + nitrite (NO_2^{2-}) collective called NO_x almost wholly in the dissolved phase	Phosphate (PO_4^+) mostly in dissolved phase but also adsorbs readily to particles
	NH_3/NH_4^+ (in a dynamic equilibrium in natural waters influenced mainly by temperature and pH)	
Dissolved Organic	Dissolved organic nitrogen (often a large portion of total nitrogen in natural waters especially those less impacted by human activities)	Dissolved organic phosphorus (can be a large portion of total phosphorus in natural waters unless impacted by human activities or there is a natural source of phosphate from mineral or animal (guano) origin
Particulate	Particulate organic nitrogen (detritus left from pieces of undecayed or partially decayed organic matter)	Particulate organic phosphorus (detritus left from pieces of undecayed or partially decayed organic matter)
	Particulate inorganic nitrogen (insignificant in natural waters and usually not considered)	Particulate inorganic phosphorus (PO_4^+ sorbs readily to inorganic and organic particle; also associated with minerals)

Organic nitrogen = Total nitrogen – (Nitrate+nitrite (NO_x)) –ammonium (NH_4) (making the reasonable assumption that negligible inorganic nitrogen is particulate)

or from laboratory analysis of Total Kjeldahl Nitrogen (TKN) which is the sum of organic nitrogen, and ammonium (NH_4^+)

Organic nitrogen = TKN – NH_4^+

Similarly, total nitrogen can be determined by the addition of concentrations found in analyzed natural water samples:

Total nitrogen = TKN + NO_x

Organic forms are typically only a small portion of total phosphorus. As such, in most cases, literature describing studies of phosphorus in watersheds and estuaries largely ignores organic forms. That said, with effort, all forms of phosphorus can be determined and relate via the following equation:

Total phosphorus = dissolved inorganic phosphorus (DIP, phosphate, PO_4^{2-}) + dissolved organic phosphorus (DOP) + total particulate phosphorus (TPP).

Practically, quantification of these forms is made using just two methods, the molybdate blue method for phosphate and the persulfate method applied to filtered samples and whole water samples.

4.3 Freshwater budget for the Estuary

Freshwater enters the Estuary predominantly via freshwater flow from the Central Valley and from flow from smaller tributaries in the nine-county Bay area (Figure 4.1). Freshwater flow from the Central Valley via the Delta dominates (89%) and flow from the smaller tributaries in the nine-county Bay Area is about double that of wastewater input. Flows are highly variable. For example, annual flow into the Bay from the Central Valley via the Delta varied by 26 times between wetter years and drier years from 1971-2000 (McKee et al., 2006). Daily inflow from the Delta is even more variable ranging from near zero to 1,540 million m^3 on which occurred on February 20th, 1986 (Figure 4.2). In order to measure accurate loads of any contaminant of interest including nutrients, it will be important for future studies to focus on capturing data during high flow events when daily flow exceeds about 40,000 cfs (98 million m^3/day) (e.g. David et al., 2009 who discussed monitoring design in relation to mercury).

Flow from local small tributaries is much more difficult to qualify given there are more than 250 individual drainages that flow to the Bay within the nine-county Bay Area. Recently SFEI has developed a 5-station index for the South Bay south of the Bay Bridge and an 3-station index for the North Bay north of the Bay Bridge (L. McKee unpublished). These indexes were developed for the period Water Year 1971 to 2010 (40 years) and adjusted using average annual flow from a calibrated rainfall-runoff model developed for the whole watershed of the nine-county Bay Area (Lent and McKee, 2011). Based on this analysis, annual flow from the small tributaries south of the Bay Bridge has varied from 84-2,419 million m^3 and maximum daily flow was 121 million m^3 on February 19th, 1986 (Figure 4.3) and annual flow from the small tributaries north of the Bay Bridge has varied from 16-2,911 million m^3 and maximum daily flow was 348 million m^3 on February 17th, 1986 (Figure 4.3). It is interesting to note that flow from northern watersheds peaked on a different day in the northern watersheds although all were wet for a full 7 days of heavy rain during, this, the largest storm in the past 40 years. It is also interesting to note that the maximum daily discharge entering the Bay from the Central Valley via the Delta is of the same magnitude at the average annual flow from the small tributaries in the nine-county Bay Area (1,589 million m^3 for the period WY 1971-2000).

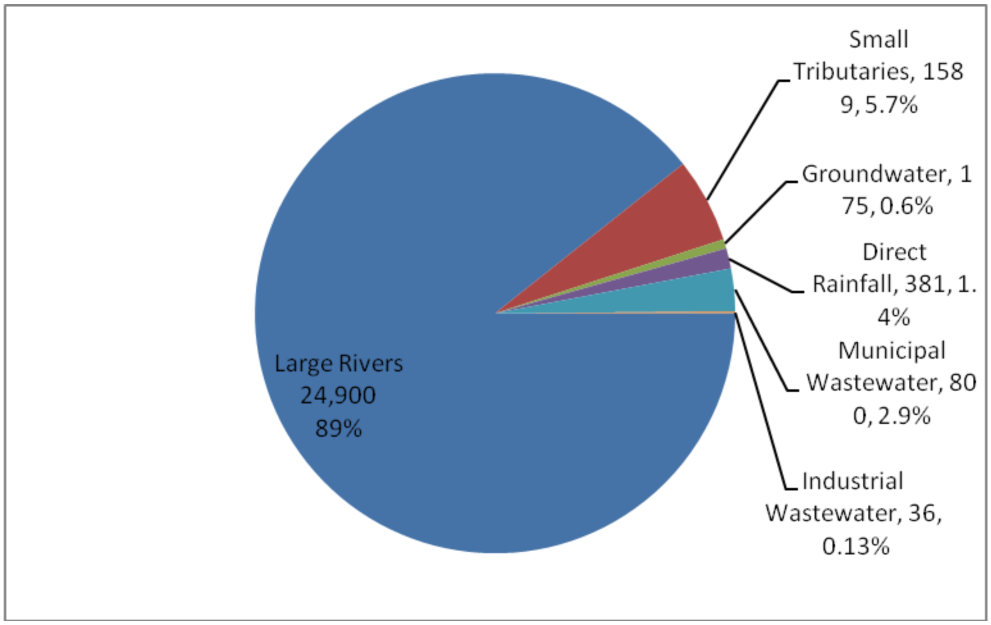


Figure 4.1. Relative flow from each of the freshwater main sources to the Bay (million m^3 per year).

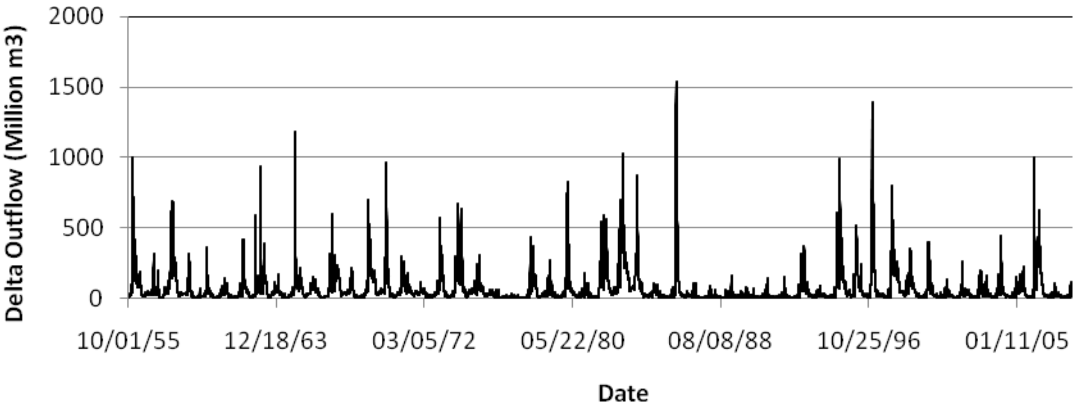


Figure 4.2. Daily Delta outflow from the Dayflow model (Source DWR website: http://www.water.ca.gov/dayflow/output/Output.cfm).

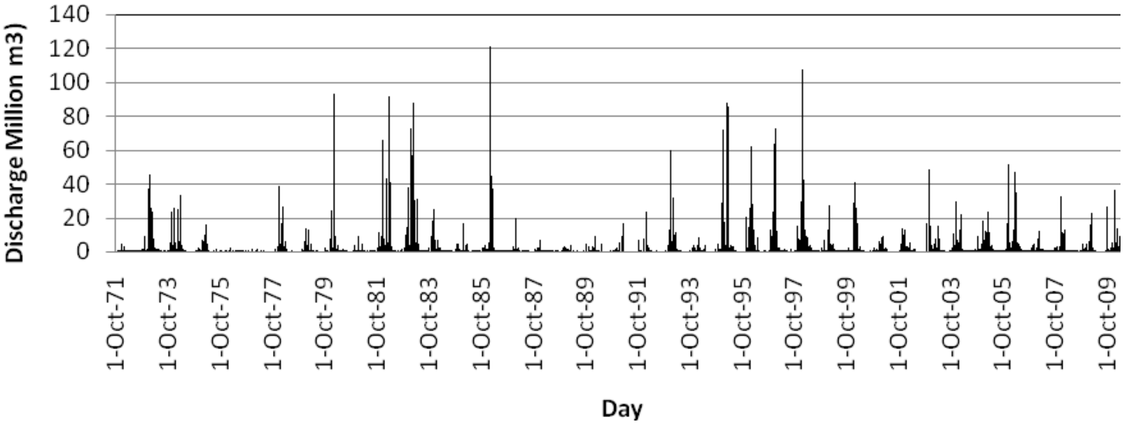


Figure 4.3. Daily flow from the local tributaries in the nine-county Bay Area to the South Bay south of the Bay Bridge based on an 5 station index (Dry Creek at Union City, Alameda Creek at Niles, Guadalupe River at Hwy 101, San Francisquito at Stanford University, and Saratoga Creek at Saratoga) adjusted to the annual average flow (586 million m^3) for water years 1971-2000 (Lent and McKee, 2011).

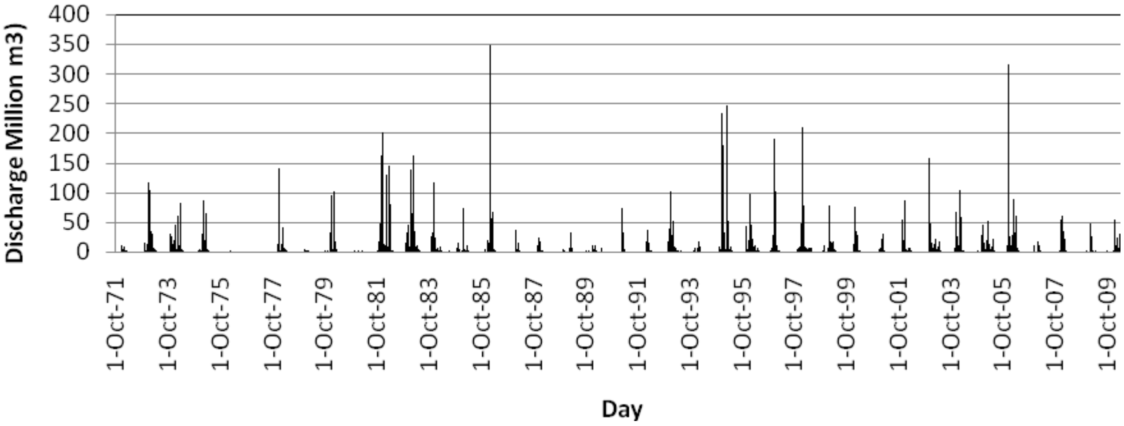


Figure 4.4. Daily flow from the local tributaries in the nine-county Bay Area to the North Bay north of the Bay Bridge based on an 3 station index (Novato Creek at Novato, Napa River near Napa, and San Ramon Creek at San Ramon) adjusted to the annual average flow (1,003 million m^3) for water years 1971-2000 (Lent and McKee, 2011).

4.4 Atmospheric Nutrient Loads Direct to the Bay Surface

Nutrients derived from atmospheric deposition to estuary surfaces has gained attention especially in oligotrophic (low nutrient) systems or systems with a small watershed area to surface area ratio where direct load to the surface may be a larger portion of the overall annual loads. In addition, there is some evidence that atmospheric derived nitrogen may be more bioavailable than terrestrially derived loads (Paerl, 1995). There is a wide variety of methods used to collect and analyze nutrient atmospheric deposition with some studies collecting wet deposition only and others focusing on dry deposition. The portion of deposition in wet and dry loads is variable. For example, in the case of nitrogen, perhaps only an additional 20% is associated with dry deposition over the ocean (Paerl, 1995). In contrast, Jassby et al. (1994) reported dry deposition on Lake Tahoe comprising 28% of nitrate input, 33% of NH_4^+ input, 70% of phosphate input, 58% of total nitrogen input (equivalent to 1.4:1 dry:wet) and 70% of total phosphorus input (equivalent to 2.3:1 dry:wet). The estimates of Kratzer et al. (2010) for the Central Valley ranged between 1.7-2.8 dry:wet for total nitrogen deposited on land surfaces.

There was only one previous estimate of nutrient deposition to SF Bay. Russell et al. (1980) estimated an annual wet and dry deposition of total nitrogen and total phosphorus of 980 and 120 metric tons (mt) respectively (Table 4.2) however they did not disclose the source of data or methods for their calculations. These estimates were made for the whole Bay only and are not spatially resolvable. Normalizing them to the whole area of the Bay (1,200 km^2), Russell et al.'s estimates are equivalent to 817 and 100 kg km^{-2} y^{-1}. Comparison of these measurements to those in Lake Tahoe provided by Jassby et al. (1994) (562 kg N km^{-2} y^{-1} and 32.6 kg P km^{-2} y^{-1}) suggest that the estimates of Russell et al. (1980) might be reasonable. Recently, estimates were made for the Central Valley of between 1.18 (1987) and 3.55 (1998) tons mi^{-1} y^{-1} (Kratzer et al., 2010) equivalent to 413-1,243 kg N km^{-2} y^{-1}. These also bracket Russell's estimates. However, population in the Bay Area has increased by 31% from 1980-2000 (2010 census data pending), vehicle miles traveled has also increased, laws regarding vehicle emissions have improved, industrial land use has decreased, and trends in fossil fuel combustion for home and office heating have undoubtedly occurred. All these changes in particular likely render previous estimates of nitrogen deposition outdated. That said, in comparison to currently available nutrient loads from wastewater and stormwater, atmospheric loads appear to be about 5% of the annual average load. It is recommended that recent data on nitrogen and phosphorus in wet and dry deposition from western US cities (Los Angeles, Portland, and Seattle) be reviewed and used to make more up-to-date estimates for the Bay Area.

4.5 Nutrient Loads from the Delta via Delta Outflow to the Bay

Nitrogen and phosphorus loads entering the Bay from the Delta have been estimated by six authors. Russell et al. (1980) estimated annual inputs of total nitrogen and total phosphorus of 13,000 and 2,400 mt respectively for 1978 but did not describe their calculation methods or data sources. Russell et al. speculated that loads would decrease due to the balance of sediment load trends, continuing changes in population and wastewater treatment and changes to agricultural drainage water practices. Jassby and

Cloern (2000) made an estimate of total organic nitrogen load from the Central Valley to the Bay of 6,205 mt. Smith and Hollibaugh (2006) made an estimate of total dissolved inorganic nitrogen (NO_x+$NH_{3/4}$) and phosphate entering the Bay from the Delta of 13,404 and 1,880 mt per year respectively. Russell's estimates are dwarfed by those of Davis et al. (2000) who combined concentration measurements from the SF Bay Regional Monitoring Program (RMP) collected during base flow conditions at the most upstream Bay locations with average annual Delta outflow. These estimates were 45,200, 5,100, and 6,400 mt for nitrate, ammonium, and phosphate respectively. Through a large data synthesis exercise to support management of drinking water supply in the Delta, Heidel et al. (2006) estimated monthly nutrient loads exported from the Delta by combining monthly Dayflow Delta outflow with total nitrogen (sum of nitrate, nitrite, and TKN) and total phosphorus concentrations. They estimated loads of 7,435 metric t for dry years and 30,885 metric t for wet years for total N and 1049 metric t and 4473 metric t for total P for dry and wet years. Most recently, Kratzer et al. (2010) reported loads based on a thorough compilation of data collected through the U.S. Geological Survey's National Water Information System database, the California Department of Water Resources, the University of California at Davis, and the U.S. Environmental Protection Agency's STOrage and RETrieval database. They estimated loads at Freeport on the Sacramento River and at Vernalis on the San Joaquin River from 1974 to 2004. Taking the average for and summing the two stations (assuming no storage or losses between these stations and the head of the Bay at Mallard Island) the loads in metric t were nitrate (6,593), ammonia (1,857), total nitrogen (16,642), phosphate (1,130), and total phosphorus (2,635). These loads are equivalent to a flow weighted average of 0.265 mg L^{-1} nitrate, 0.075 mg L^{-1} ammonia, and 0.045 mg L^{-1} phosphate assuming an annual average flow of 24,900 million m^3 (McKee et al., 2006). These concentrations are very similar to averages calculated from the RMP monitoring at the head of the estuary (sites BG20 and BG30) (nitrate: 0.286 mg L^{-1}; phosphate: 0.069 mg L^{-1}).

Comparing all these estimates (Table 4.2) it can be seen that the available estimates to do not make a lot of sense. For example, the estimates of dissolved forms of nitrogen (Davis et al., 2000; Smith and Hollibaugh, 2006) are greater than the estimate of total nitrogen load by Russell et al. (1980). Similarly, the estimate of phosphate load by Davis et al. (2000) is about 3 times greater than the estimate by Smith and Hollibaugh (2006) and greater than the total phosphorus load estimate that Russell et al. made. Some of these discrepancies are probably due to temporal trends, however, in truth, no data used by these authors were collected during high flow or for the purposes of calculating loads.

The RMP has continued to collected data in the northern segments of the Bay. While these data are not collected during wet weather flow, they can be used to make more up-to-date estimate of nutrient loads during the dry season that are relatively accurate; wet season loads estimate can be improved by careful manipulation of the data taking into account knowledge about sediment transport (McKee et al., 2006). It is recommended that some effort be put into making these improved estimates as an interim measure to help support immediate planning efforts and decisions about priority information development.

In the medium term, to support the development of a hydrodynamic model on estuarine nutrient response, it is recommended that wet weather data collection of nutrients be initiated at the DWR sampling location at Mallard Island at the head of Suisun Bay. Nutrient forms monitored should include nitrate, ammonium, total nitrogen, phosphate and total phosphorus. Given the existence of a long term turbidity data set at Mallard Island (supported by the USGS) and the likelihood that total phosphorous and total nitrogen correlate at least to some extent with turbidity, just a few years of data will likely support a reasonable estimate of daily loads during wet and dry seasons for the period Water Year 1995 – present if we make the assumption that nutrient loads are not trending. Given the size of the Sacramento River system and the fact that it can take many days to weeks for a flood wave to pass down the system, a daily time step is entirely sufficient for describing loading dynamics at the head of the estuary. Sampling and interpretation methods have been developed by McKee et al. (2006) and further refined by David et al. (2009). These methods could be augmented with automated sampling technology.

4.6 Nutrient Loads from tributaries in the Nine-County Bay Area

There have been several annual scale estimates of nutrient loads entering the Bay via urban and non-urban tributary flow emanating from the nine counties that fringe the Bay. Russell et al. (1980) estimated that approximately 2,300 and 470 mt of total nitrogen and phosphorus was entering the Bay on average in 1978 and suggested that there would likely be no change into the future. The estimate for total nitrogen appears consistent with the estimate of 1,500 mt of nitrate per year made by Davis et al. (2000) in contrast to the estimate of Smith and Hollibaugh (2006) for total dissolved inorganic nitrogen (nitrate and ammonium/ammonium) of just 245 mt. The estimates for phosphate (34 mt) are similarly not in agreement with other estimates (Table 4.2). Smith and Hollibaugh's estimates were made in the absence of any data on nutrient concentrations in local tributaries and were based on assuming concentrations in urban runoff were the same as Delta outflow. In addition, estimate of water flow from small tributaries from around the Bay were challenged by a lack of data. There have been a number of reliable spatially resolved estimates made of flow associated with small tributaries in the nine-county Bay Area (Russell et al., 1980; Davis et al., 2000; McKee et al., 2003; Lewicki and McKee, 2009; Lent and McKee, in preparation). For the most part, these authors have reported an annual average flow of approximately 1,000 million m^3 per year. However, runoff from Bay Area tributaries is very well understood based on a number of currently well maintained USGS and county operated gauges. In addition, Gilbreath and McKee (2010) collated runoff data from 44 urban stormwater pump stations; a data set that could be continually maintained. In addition, nutrient data have been collected in the Napa River and Sonoma Creek watersheds (McKee and Krottje, 2005), Pinole Creek watershed (Pearce et al., 2006), Cerrito Creek and Ettie Street pump station watershed (EBMUD, 2010), and in Zone 4 Line A watershed (Gilbreath and McKee et al., in preparation) and perhaps others. It is recommended that these data be used to make new estimates of nutrient loads for tributaries entering the Bay from the urbanized counties around the Bay to support planning efforts to prioritize new information development. Depending on the data needs of an estuarine nutrient response model, new data collection may need to be initiated to support either improved empirical loads calculations or the development of a watershed loads model with outputs at needed at potentially an hourly if not daily time step.

4.7 Nutrient Loads from Municipal Wastewater

Modern sewage treatment practices are designed to remove solids, biological oxygen demand, and pathogens during primary and secondary treatment phases. During the first two phases some phosphorus and, to a lesser extent, nitrogen is removed through sedimentation, but greater nutrient removal is achieved through tertiary treatment. On average, about 871 million m^3 of wastewater is currently discharged to the Bay annually (Oram et al., 2008), just 10% less than that of stormwater discharge (McKee et al., 2003). There have been three estimates of nutrient loads from wastewater. Smith and Hollibaugh (2006) remarked on the importance of wastewater nutrient loads; they found that wastewater accounted for 50% of the wet season nutrient loads and 80% of the dry season loads in the South Bay. They collated flow information from 12 wastewater treatment plants and nutrient concentration data for five of the larger plants for the period 1990-1995 and interpolated the data to make estimates for all of the plants. Using these data, estimates of 5,983 and 1323 mt of dissolved inorganic nitrogen ($NOx+NH_{3/4}$) and dissolved inorganic phosphorus (DIP or PO_4) were made were made for the South Bay and 1,994 and 230 mt of DIN and DIP were made for the North Bay. These estimates appear to be similar to those of Davis et al. (2000) for nitrogen and about 4 times lower for phosphate. In contrast, the load estimates for total nitrogen by Russell et al. (1980) appear to be much greater (given most nitrogen discharged after secondary treatment is likely to be nitrate and ammonium). In contrast, the total phosphorus loading estimate of Russell et al. is 10 to 100 times lower. Again these numbers are not making sense; there are large discrepancies between authors and partitioning between total and dissolved phases are not logical.

Table 4.2. Published nutrient loading estimates for San Francisco Bay (mt). Note these estimates are mostly based on very limited data assembled from monitoring programs that were not designed for estimating mass loadings.

Source or Pathway	Author	Bay Segment	Time period of estimate	Total Nitrogen (TN-N)	Nitrate + Nitrite (NOx-N)	Ammonium (NH4-N)	Total Inorganic Nitrogen (TIN)	Total organic nitrogen (TON-N)	Total phosphorus (TP-P)	Phosphate (PO4-P)
Aerial deposition	Russell et al., 1980	Whole Bay	1978	980					120	
		Whole Bay	2000 (Authors estimate)	980					120	
Delta outflow	Russell et al., 1980	North Bay	1978	13,000					2,400	
		North Bay	2000 (Authors estimate)	78,000					1,600	
	Heidel et al., 2006	North Bay	Wet year	30,885					4,473	
	Davis et al., 2000	North Bay	Average year		45,200	5,100				6,400
	Jassby et al., 1993	North Bay	Average year (1980 estimate)							
	Jassby and Cloern, 2000	North Bay	Average year (1978-91 estimate)					6,205		
	Smith & Hollibaugh, 2006*	North Bay	Average (1990-95)				13,404			1,880
	Kratzer et al., 2010	North Bay	Average (1974-04	16,642	6,593	1,857			2,635	1,130
Local small tributaries (Urban + non-urban stormwater)	Russell et al., 1980	Whole Bay	1978	2,300					470	
		Whole Bay	2000 (Authors estimate)	2,400					480	
	Davis et al., 2000	Whole Bay	Average year		1,500					510
	Smith & Hollibaugh, 2006*	South of the Richmond Bridge (Central and South Bays)	Average (1990-95)				245			34
Waste water	Russell et al., 1980	Whole Bay	1978	21,000					10	
		Whole Bay	2000 (Authors estimate)	24,000					15	
	Davis et al., 2000	Whole Bay	Average year		3,110					970
	Smith & Hollibaugh, 2006*	South of the Richmond Bridge (Central and South Bays)	Average (1990-95)				5,983			1,323
		North Bay	Average (1990-95)				1,994			230

* Converted from moles to mass using a molecular weight of 14.01 g per mol for N and 30.97 g per mol for P.

The local Water Board issues permits effluent limits to wastewater agencies. In response to these permits, a number of data sets on both flow rates and nutrient concentrations have been generated in recent times. We are currently aware that 15 of the roughly 40 treatment plants in the Bay Area have data available for ammonium concentrations on a monthly basis. Six of these 15 data sets are for systems with tertiary treatment; four of these six measure nitrate. In one case (Fairfield-Suisun WWTP) there are also data for organic nitrogen, total nitrogen, and total phosphorus.

It is recommended that available data be combined with flow data for each of the plants to make new estimates of nutrient loads taking into account treatment methods and population trends. It should be possible to make estimates for annual and wet and dry season loads for at least the last 10 years with reasonable confidence and for the last 20 years with lower confidence for ammonium, nitrate and with overall lower confidence for phosphate. In addition, all treatment plants that discharge to the Bay should be encouraged to begin analyzing effluent for total and dissolved inorganic nutrients and to submit these data to the SFRWQCB on a regular basis. Finally, it is recommended that the POTWs conduct a laboratory inter-comparison on nutrient methods to assure comparability of estimates.

4.8 Loads from Industrial Dischargers

Presently there is no estimate for nutrient loads for industrial discharges to the Bay. For the most part, industrial waste is not treated on site but rather introduced to the municipal sewer system and treated by the local wastewater treatment plant. However, in a few cases treatment is performed on site and treated wastewater is discharged to the Bay. Examples include the oil refineries and C&H sugar (Table 4.3). While we do know that these industries have characterized their effluent streams in the 1990s, we are not aware if there is more recent data available or if their reuse practices have changed in the last 15 years. It is recommended that a request be made to the industrial dischargers of the Bay Area to provide the latest data on flow and concentrations of nutrients in their waste effluent streams.

Table 4.3. Industrial dischargers in the Bay Area with data from the 1990s on flow and nutrient concentrations.

Facility	Volume (MGD)	Treatment type
C&H Sugar	1	Activated sludge
Tosco Corp. at Avon	5	Pond/RBC/carbon
Tosco Corp. at Rodeo	3	Pond/RBC/carbon
Shell Oil Company	6	Activated sludge/carbon
EXXON	3	Activated sludge/carbon
Chevron U.S.A.	8	Activated sludge/wetland

4.9 Nutrient Loads from Groundwater

Nutrient loads entering the Bay from groundwater sources are not available. A number of drinking water supply agencies in the Bay Area monitor losses from their groundwater recharge systems via seepage to the Bay (SFPUC, 1997; Hanson et al., 2004; Thomas Neisar pers. comm., 2010; Muir, 1996 cited in Water Board, 2010). Based on these four study areas, it is estimated that groundwater discharge occurs at an average rate of 0.7 Mm^3 per km shoreline length per year. The perimeter of the Bay is approximately 250 km thus ground water discharge for the whole Bay is estimated to be 175 Mm^3 or about 17.5% of the surface water discharge. Given the extensive use of ground water recharge in the Bay Area for drinking water supply, the use of extensive landscape irrigation which maintains dry-weather flow in our urban drainage systems, and the presence of large alluvial deltas at the months of our larger urban tributaries that ring almost the entire Bay margin (Alameda Creek, Coyote Creek, Guadalupe River, San Francisquito Creek, Novato Creek, Petaluma River, Sonoma Creek, Napa River, Green Valley Creek, Walnut Creek, San Pablo/Wildcat Creeks, San Leandro Creek, and San Lorenzo Creek), this portion of groundwater discharge seems believable despite our clay soils nearer the surface.

Nutrient concentrations have been measured by the USGS in 79 wells tapping the ground water systems of the Bay Area (Ray et al., 2009). Data is available for ortho-phosphorus (phosphate or DIP) for all 79 wells, whereas data are sparser for ammonium, and NOx (Table 4.4). It can be seen that nitrate concentrations are very high in our groundwater systems whereas ammonium and phosphate are at lower concentrations. The nitrate concentrations in this study are not dissimilar to those observed in the groundwater basins of Sonoma and Napa Counties where maximum concentrations of nitrate of 5.2 mg L^{-1} were observed (Kulongoski et al., 2010; USGS Scientific Investigations Report 2010–5089). Combining median concentrations with estimates of groundwater flow provides first order estimates of nutrient loads to the Bay from groundwater (Table 4.4). The load of nitrate in groundwater moderately large relative to other pathways and is greater than the nutrient loads estimate for small tributaries made of Smith and Hollibaugh (2006). However, as mentioned, the estimates by these authors were based on very limited data and assumptions. Loads of ammonium and phosphate are estimated to be small relative to other pathways. Given its overall magnitude of these groundwater estimates in comparison to other pathways, further work may not be a high priority. It is recommended that we seek expert review from the USGS groundwater section as part of the decision making and prioritization process for any next steps with regards to groundwater flows and loads of nutrients to the Bay.

Table 4.4. Nutrient concentrations and loads estimate for San Francisco Bay based on median concentrations found in groundwater of 79 wells in the Bay Area (Ray et al., 2009) and an estimate of groundwater discharge to San Francisco Bay of 175 million m^3 per year.

	Ammonium (mg L^{-1})	**NOx (mg L^{-1})**	**PO4 (mg L^{-1})**
Count (n)	22	66	79
Minimum	0.017	0.05	0.006
Maximum	3.88	12.7	1.27
Mean	0.488	3.38	0.102

Median	0.099	3.01	0.051
Load estimate (mt per year)	17	530	8.9

4.10 Exchange with Coastal Ocean

Nutrients and biogenic materials pass in and out of estuaries in response to tides and freshwater forcing. It is well known that in systems like SF Bay which have seasonal freshwater patterns, the net flux during the wet season is from estuary to ocean (e.g., McKee et al., 2000). In contrast, during the dry season, net flux for some nutrient forms (e.g., organic nitrogen) can be from the ocean into the estuary (McKee et al., 2000) and this can be enhanced during upwelling events when nutrient concentrations (particularly phosphorus) in the coastal ocean can be enhanced. Over the years there have been a number of measurements made of water and salt flux through the ocean boundary of the Bay known as the Golden Gate (e.g., Largier, 1996; Fram et al., 2007). There have been no estimates of nutrients flux in this x-section that we are aware of. However, one study (Martin et al., 2007) did quantify chlorophyll a flux during a neap and spring tide during wet season runoff (March 2002), summer upwelling (July 2003), and autumn relaxation (October 2002). The found that that net flux (advective + dispersive) was large and net seaward during the wet season observations, large and net landward during the summer observations and small and indiscernible from zero in the autumn. It is this very type of outcome that could be enhanced to build a statistical relationship between hydrological forcing and flux conditions (e.g., McKee et al., 2000). In their case, they sampled during spring and neap tides during wet season, mid and late dry season conditions (upwelling) and during three flood events of a range of sizes and use the data to build a statistical understanding between freshwater flow and season and net flux. Therefore we recommend that a data set be developed during the next deployment of ACDP instrumentation that quantifies the nutrient concentrations in any surface layer and the bottom layer in the x-section every 1-1.5 hours for 25 hours during spring and neap tides. This should be repeated for a range of seasonal and flow conditions. Alternatively, a data set that captures the annual variability in the ocean-estuary gradient could be combined with the estimates of exchange coefficients (Fram et al., 2007; Martin et al., 2007) to define a net nutrient flux (Mark Stacey, UC Berkeley, personal communication, March 2011). Stacey suggests that the nutrient data set should include samples from a few depths along a line from Central Bay out to the Gulf of the Farallones perhaps monthly, but ensuring that the samples are consistently collected on the same tidal phase (like the USGS Polaris cruises) but final design of a sampling program would need to be the subject of a workshop that would include a number of local experts. Ultimately the data set collected should be suitable for both immediate flux estimates based on either statistical or event modeling and would provide data to support the calibration and verification of the ocean boundary of a system scale hydrodynamic model.

4.11 Summary and Recommendations

SF Bay is regarded as a nutrient enriched estuary, based on the ambient concentrations and estimated loads of nutrients to the Bay (Cloern and Dugdale, 2010). As discussed in this section, estimates of nutrients loads from external sources and pathways are poorly understood. For the most part, published load estimates are outdated by one or even two decades and were either based on data collection methods that were not designed for loads estimation, were based on assumptions that provided guesses at best or were based on data sets that have now been substantially improved with ongoing collection through time. Given changes to wastewater treatment technologies, increases in population, changes to land use, home heating methods, pet husbandry, fertilizer use in agricultural and urban areas, and other factors that influence nutrient loads, it would seem likely that nutrient loads are changing through time. However, data sets are of limited use to make any suggestion of the overall effect of these factors on nutrient load trends through time.

In order to develop models that provide a linkage between indicators of SF Bay health in relation to nutrient enrichment and the nutrient management knobs that can be turned, accurate estimates of nutrient loads are needed with sufficient temporal and spatial resolution. Given the magnitude of the nutrient loads from the Central Valley, wastewater, and stormwater, it is recommended that these pathway a major focus; loads from atmospheric deposition and groundwater are much smaller and together constitute no more than 10% of the total loads to the Bay and thus should receive a smaller emphasis. Table 4.5 provides a summary of data gaps and recommended next steps. Recommendations generally fall into two categories:

1) Revising and updating estimates of nutrients from the different sources, based on existing data
2) Identification of data needed to develop a dynamic watershed model.

The exercise of revising and updating estimates of nutrients from the various sources, based on existing data would help to better inform our understanding of the dominant nutrient sources for each distinct region of the Bay. This would, in turn, assist in decision-making to prioritize new data collection to develop the watershed, airshed and oceanic exchange/loading subcomponents of the loading model.

The loading model would be used to establish load allocations of nutrients that the SF Bay estuary can sustainably assimilate. Although data could be collected to make empirical estimates, the ultimate utility of a loading model is to generate simulations of the past, present or future state of the Estuary and watershed, airshed and ocean (e.g., population growth, climate change, etc.) to explore potential effects of management actions and evaluate alternatives. Thus these models would be a key component of a strategy to adaptively manage SF Bay. The loading model, which would incorporate information about land use, industrial and wastewater plant discharges, wet and dry atmospheric deposition, oceanic exchange, weather and other sources, would include four components: 1) a hydrologic sub-model, 2) a non-point source sub-model (wet and dry weather runoff), 3) a river sub-model which routes flow and associated nutrient loads to the Estuary from the Delta and other major tributaries that drain to the Bay, and 4) a oceanic submodel that would create boundary conditions for exchange of the estuary with the coastal ocean.

Table 4.5. Summary of data gaps and recommended next steps for quantification of nutrient loads to San Francisco Bay.

Source	Data Gaps Identified	Recommended Next Steps
Atmospheric Deposition	No recently published data on wet & dry atmospheric deposition.	Loads likely relatively small. Literature review to determine range of N and P deposition rates for West Coast coastal urban areas. Recommend baseline atmospheric deposition monitoring of wet and dry N and P deposition over 1-2 year period to better constrain estimates.
Terrestrial Loads from Delta	Data available through RMP on dry season concentrations. No data available on wet weather concentrations during storm flow.	Loads likely large. Recommend analysis of existing RMP data to estimate dry season nutrient loads. Initiate wet weather sampling at the DWR gauge at Mallard Island at the head of Suisun Bay to support improved daily loads estimates for 1995-present.
Municipal Effluent	Data available for 15 of approx. 40 POTWs.	Synthesize existing nutrient discharge and concentration data to estimate loads over period of last 10-20 years. Encourage all treatment plants that discharge to the Bay to begin analyzing effluent for total and dissolved inorganic nutrients and to submit these data to the SFRWQCB on a regular basis. Recommend that the POTWs conduct a laboratory inter-comparison on nutrient methods to assure comparability of estimates.
Industrial Effluent	Some data available from the 1990s. Recent data availability unknown.	Synthesize available data to provide information for prioritization of any future steps.
Stormwater	Some data available but general lack of land use-specific wet weather data sufficient to calibrate and verify a watershed loads model.	Synthesize data to provide an updated estimate of stormwater contributions to assist prioritization of next steps. Scope the data needs for development of a dynamic watershed loading model.
Groundwater	Some data available from 79 USGS monitoring stations surrounding the Bay. Flow data currently less well understood.	Refine current loads estimates with review from local USGS groundwater experts in order to support prioritization of next steps.
Exchange with Coastal Ocean	Some data available for fluxes of water and sediments during selected tides and seasons in the past decade collected by USGS and US Berkeley using comparable methods.	Initiate a workgroup of local experts to design a sampling program for nutrient flux at the Golden Gate boundary, with the intent of developing a hydrodynamic and material flux dynamic model to describe exchange with coastal ocean

5. Evaluation of Candidate NNE Indicators for Application in San Francisco Bay Estuary and Summary of Existing Literature

5.1 Introduction

Development of an NNE framework for SF Bay estuary requires the selection of appropriate ecological response indicators to diagnose eutrophication or other adverse effects of nutrient over enrichment. The purpose of this section was to summarize existing information available on each indicator (Table 5.1.1), evaluate the appropriateness of candidate NNE candidate indicators for SF Bay, and identify data gaps in information needed to develop NNE thresholds.

Table 5.1.1. Candidate indicators reviewed for potential development within the NNE framework for San Francisco Bay.

Type	Indicator Group	Indicator or Metric	Section to refer to
Primary Producers	**Phytoplankton**	Phytoplankton Biomass (chlorophyll a concentration)	5.2
		Productivity (carbon fixed per unit volume and time)	
		Assemblage/Taxonomic Composition	
		Harmful algal bloom species -- cell count	
		Harmful algal bloom species – toxins	
	Macroalgae	Percent Cover and Biomass	5.3
	Seagrass and Brackish Water Submerged Aquatic Vegetation	Phytoplankton Biomass	5.4
		Macroalgal Biomass and Cover	
		Epiphyte Load	
		Light Attenuation	
Consumers	**Benthic Macroinvertebrate**	Benthic infauna taxonomic composition, abundance and biomass	5.5
	Jellyfish	Taxonomic composition and abundance	5.6
Water Column Physio-chemistry	**Nutrient Concentrations and Ratios**	Ammonium	5.7
		Urea	
	Dissolved Oxygen	Concentration	

The appropriateness each of the candidate indicators was evaluated relative to four criteria:

1. Ample scientific evidence demonstrating a linkage to SF Bay estuary beneficial uses
2. The existence or potential to develop a predictive relationship with causal factors such as nutrient concentrations/loads and other factors known to regulate response to eutrophication (hydrology, etc.)
3. Availability of a scientifically sound and practical method to measure the indicator
4. The ability to show a trend either towards increasing or/and decreasing eutrophication with an acceptable signal: noise ratio

5.2 Phytoplankton

Phytoplankton have a variety of characteristics that make them potentially useful as indicators of eutrophication in estuaries. Phytoplankton are highly sensitive indicators of nutrient availability in surface waters since their growth rates are relatively rapid, growth responses occur at a wide range of nutrient concentrations and photosynthetic responses can be measured using an array of sensitive techniques (Paerl et al., 2007). Phytoplankton can be described by a number of indicators that may be relevant for use in the SF Bay NNE framework. They include:

- Biomass, as measured by water column chlorophyll a;
- Productivity, as measured by the rate of carbon fixed per unit time per square meter (areal) or per cubic meter (volumetric)
- Assemblage), as measured by the species taxonomic composition, the relative abundance of species (as measured by cell counts), and/or size class of the cells.
- Abundance of HAB species and HAB toxins

In this sub-section we describe the current understanding of spatial and temporal variation in phytoplankton on seasonal, interannual and decadal scale trends, the factors affecting phytoplankton biomass and community structure in SF Bay, and discuss the suitability of phytoplankton as an indicator of eutrophication.

5.2.1 Applicable Habitat Types

Phytoplankton require light to photosynthesis and therefore are typically limited to the shallow to deepwater subtidal regions of an estuary. As depths decrease towards the shallow subtidal zone and particularly in macrotidal estuaries, benthic microalgae and macroalgae that are attached to sediment are at a competitive advantage over phytoplankton which can be easily flushed out during tidal cycles or torn apart by tidal currents or wave energy. With increasing depth, phytoplankton's advantage over benthic algae and rooted bed-forming submerged aquatic vegetation and seagrass increases, because phytoplankton are able to position themselves in the upper portion of the water column and outcompete other primary producers for light and nutrients. In shallow subtidal habitats, phytoplankton can be found in codominance with SAV, microphytobenthos, and macroalgae. In turbid or deepwater subtidal habitats, particularly in wave dominated environments, phytoplankton species tends to be the dominant primary producer, or co-dominant with microphytobenthos in deepwater habitats with high water clarity (Day et al., 1989; Wetzel, 2001).

North, Central and South Bay are dominated by subtidal habitat (71, 96, and 68%, respectively). For this reason, phytoplankton is the largest component of primary producer biomass in SF Bay (Cloern et al., 2000) with carbon production from planktonic species historically making up roughly 70% of total production (Jassby et al., 1993). Measures of phytoplankton are thus key candidate indicators for the SF Bay NNE framework in subtidal habitats.

5.2.2 Available Data on Phytoplankton Biomass, Productivity, and Assemblage

Although nutrient concentrations are relatively high in SF Bay, algal biomass has been relatively low compared to other River dominated systems (e.g., Chesapeake Bay; Cloern, 2001), though most recent estimates for the Bay as a whole show productivity in the normal range of other temperate latitude estuaries (Cloern et al., 2006). Much of the annual production occurs not from the year-round baseline persistence of phytoplankton but rather when algae blooms occur. Algal blooms have been defined by Cloern (1996) as:

> *…events of rapid production and accumulation of phytoplankton biomass that are usually responses to changing physical forcings originating in the coastal ocean (e.g., tides), the atmosphere (wind), or on the land surface (precipitation and river runoff). These physical forcings have different timescales of variability, so algal blooms can be short-term episodic events, recurrent seasonal phenomena, or rare events associated with exceptional climatic or hydrologic conditions (Cloern, 1996, p 127, 133).*

Cloern (1982) defined blooms in SF Bay to be chlorophyll a concentrations >10 $\mu g\ L^{-1}$. Algal blooms are natural events and are the foundation for the secondary productivity which supports the SF Bay food web. There is a concern that increases in the phytoplankton biomass or changes in species composition (in particular, shifts in the frequency and duration of blooms dominated by harmful algal species) may occur in the future in response to changing nutrient loads, turbidity and other limiting factors.

The USGS (Menlo Park Laboratory) has been collecting water quality data in SF Bay on nutrient concentrations and related ancillary data continuously for 39 years beginning 1968 and on phytoplankton since 1977. Their research program includes measurements of water quality from a monthly ship cruse of 39 fixed locations 3-6 km apart along the 145 kilometer spine of the entire Estuary. Since the USGS sample-collection was driven by research questions, it has not always been as regular or systematic as would occur in a monitoring program. For example, the USGS stopped sampling completely in 1981 after the spring bloom and didn't sample in the North Bay from about 1980-1987. That accepted, the database generated presently includes >11,000 discrete laboratory measurements of the chlorophyll a in water samples and 156,610 estimates of chlorophyll a made from a linear relationship between fluorometer voltage and discrete lab measurements. In addition to information collected during these regular monthly cruses, real-time remote observing instrumentation has greatly enhanced the surveillance in recent years (Cloern et al., 2005b), though it should be noted that remote sensing captures only surface blooms and many blooms dominated by HAB species are not easily distinguished using readily available, multi-spectral remote sensing products (e.g., MODIS). On many occasions, the USGS group and collaborating coauthors have also carried out special studies in locations off transect (e.g., Cloern and Oremland, 1983; Cloern et al., 1985; Powell et al., 1986; Lucas and Cloern, 2002; Thompson et al., 2008; discussed in detail below). Estimates of pelagic primary production were made estimated either directly using the ^{14}C radioisotope tracer method (Steeman Nielson 1952, Cole and Cloern 1984) or indirectly through an empirical model that derives productivity from biomass and light attenuation (Cole and Cloern 1987).

More recently, continuous monitoring has also conducted by scientists at the San Francisco State University's Romberg Tiburon Center for Environmental Studies. They have been collecting information on chlorophyll a and a number of ancillary parameters every 6 minutes using instruments mounted just offshore at the end of a 200-ft pier adjacent to the RTC campus on the Tiburon peninsular (R. Dugdale and F. Wilkerson, pers. comm.). This data is part of the observing networks of the Council on Ocean Affairs, Science and Technology (COAST) and the Central and Northern California Ocean Observing System (CeNCOOS). Data on chlorophyll a have been collected 0.5 m below the water surface using a flotation platform that adjusts with the tides from April 2006 to January 2009 and at a fixed datum 1 meter below lower low tide from 12/2008-present. In addition, the group has been publishing on a number of focused research projects on the ecology and controls of diatom productivity in the northern reaches of SF Bay. The research groups at the USGS and the RTC have been responsible for the majority of systematically collected measurements on phytoplankton biomass and community composition in SF Bay downstream from the Sacramento – San Joaquin confluence near the Region 2/Region 5 Water Board boundary.

5.2.3 *Factors Effecting Temporal and Spatial Variation of Indicator Phytoplankton Biomass and Productivity*

In SF Bay, the biomass and primary productivity associated with phytoplankton varies in space and time in response to nutrient availability from external loads (e.g., Wilkerson et al., 2006; Dugdale et al., 2007) and internal regeneration (Grenz et al., 2000), grazing (Cloern et al, 1985; Thompson et al., 2008), stratification (Cloern, 1991; Cloern, 1996), water temperature (Cloern et al., 2007; Lehman et al., 2008), tidal energy (Lucas and Cloern, 2002), transparency (May et al., 2003), wind/wave energy (May et al., 2003), the availability of seed cysts (Cloern and Dufford, 2005; Cloern et al., 2007), UV radiation effects on nitrate versus ammonium assimilation perhaps due to disruptions of enzyme pathways (Hogue et al., 2005), differential uptake of nitrate and ammonium by larger versus smaller cells (Wilkerson et al., 2006), inhibition of nitrate uptake by ammonium (Wilkerson et al., 2006; Dugdale et al., 2007), predation by benthic invertebrates (e.g., Thompson et al., 2008), and variations in the phase of the Pacific Decadal Oscillation and related changes to top down predation of benthic invertebrates (Cloern et al., 2007). These factors lead to spatial gradients across shoals to the axis, between segments of the Bay, and temporal variation at scales ranging from days to years to decades.

Spatial Variability

In the broadest sense, the Bay can be divided into two main regions, the North Bay and the South Bay. The North Bay is a river dominated estuary where spatial and temporal variability is driven by intra- and inter-annual variations in freshwater, sediment, and nutrient input from urban and agricultural sources within the Sacramento and San Joaquin River watersheds (Sigleo and Macko, 2002; Smith and Hollibaugh, 2006; Wilkerson et al., 2006). The estimated average freshwater flushing time of the North Bay is 72 days (Engle et al., 2007). The South Bay in contrast acts more like a tidal lagoon with relatively low freshwater input relative to basin volume; it is dominated in the summer months by wastewater discharge (Cloern et al., 2000; Smith and Hollibaugh, 2006). The average estimated freshwater flushing time of the South Bay is over 4,000 days (Engle et al., 2007). Within these broad classes, due mainly to physiographic controls on freshwater and tidal flow (Powell et al., 1986), the Bay can be further divided into six strata or segments (see Figure 3.6 in Section 3 of this report) that have small within strata variance relative to variability along the whole gradient between marine and freshwater conditions (Cloern et al., 2000).

Chlorophyll a varies laterally from shallow areas to the axis (Cloern et al., 1985; Thompson et al., 2008) often associated with variations in turbidity and the timing of wind relative to the tidal cycle, fetch, and tidal forces (May et al., 2003). For example, while the focus of an early study by Cloern et al. (1985) was on intra-annual temporal variability in phytoplankton biomass, the paper also illustrated biomass variability across lateral gradients in SF Bay (Figure 5.2.1). More recently Thompson et al. (2008) discussed strong lateral gradients in the South Bay (Figure 5.2.3). Their observations supported the hypothesis that bloom generation began on the east shoals in most years and spread into the channels if the bloom persisted. There was one instance, however, when a channel produced phytoplankton bloom was observed perhaps attributable to persistent stratification (Lucas et al., 1998).

However, by far the most persistent spatial gradient of phytoplankton biomass variation occurs between the ocean entrance at the Golden Gate Bridge and the fresh water extremities in the Lower South Bay and the Sacramento River Delta (Cloern et al., 2000). Algal productivity varies widely in each region of the Bay. Based on data collected from 1995 to 2009, average chlorophyll a concentrations vary from 13 μg L^{-1} in the lower South Bay to 2.6 μg L^{-1} in the river-dominated North Bay (Table 5.2.1). Suisun Bay, although high in nutrients, exhibits relatively low mean chlorophyll a concentrations relative to the South Bay (Wilkerson et al., 2006). Concentrations are more temporally variable both within a year and between years further from the Golden Gate.

The causes for the Bay wide trends include changes in water clarity due to less suspended sediment (Schoellhamer, 2009), lower metal inhibition due to improvements in wastewater treatment, increased seeding from ocean populations (Figure 5.2.3; Cloern et al., 2005), declines in consumption by bivalves due to increases in predation by juvenile English sole and speckled sanddabs, and declines in phytoplankton consumption by bivalves and zooplankton due to recent new invasive species introductions (Cloern et al., 2006).

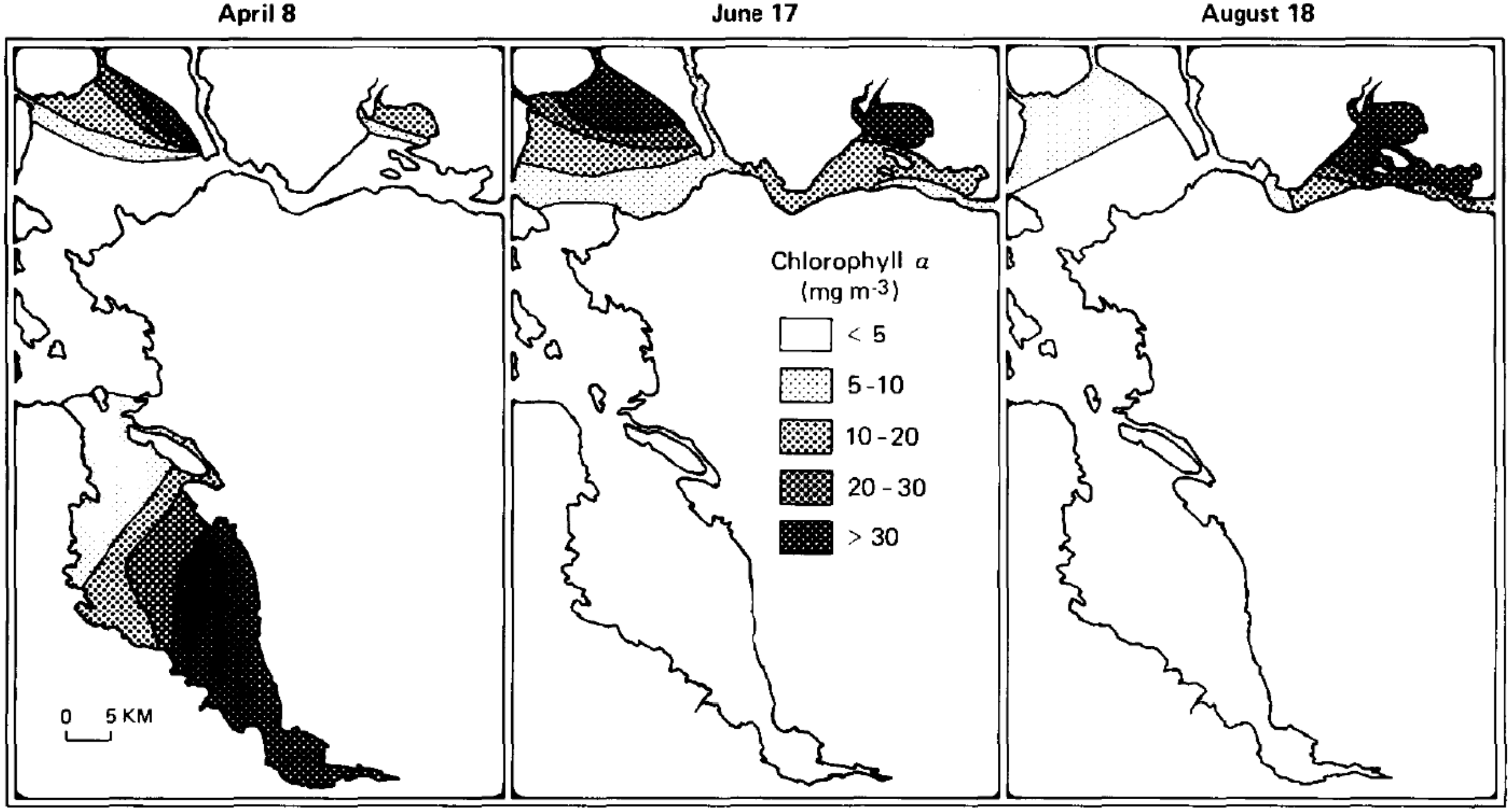


Figure 5.2.1. Lateral variability in chlorophyll a concentrations based on measurements at 106 sites during 1980. Figure extracted from Cloern et al. (1985).

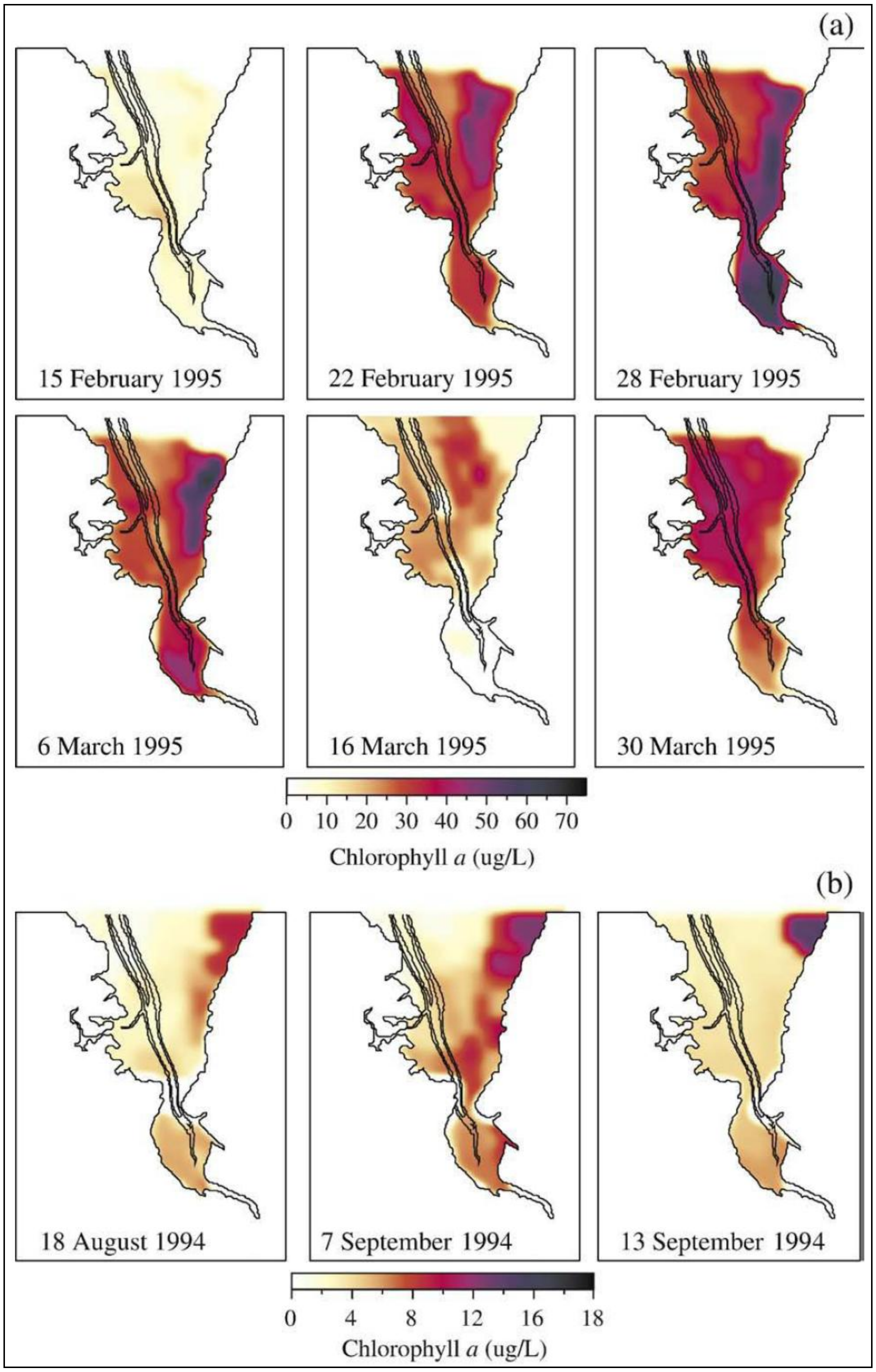


Figure 5.2.2. Lateral variability in chlorophyll a concentrations in the South Bay sites during 1995 (59 stations; a) and 1994 (49 stations; b). Figure extracted from Thompson et al. (2008).

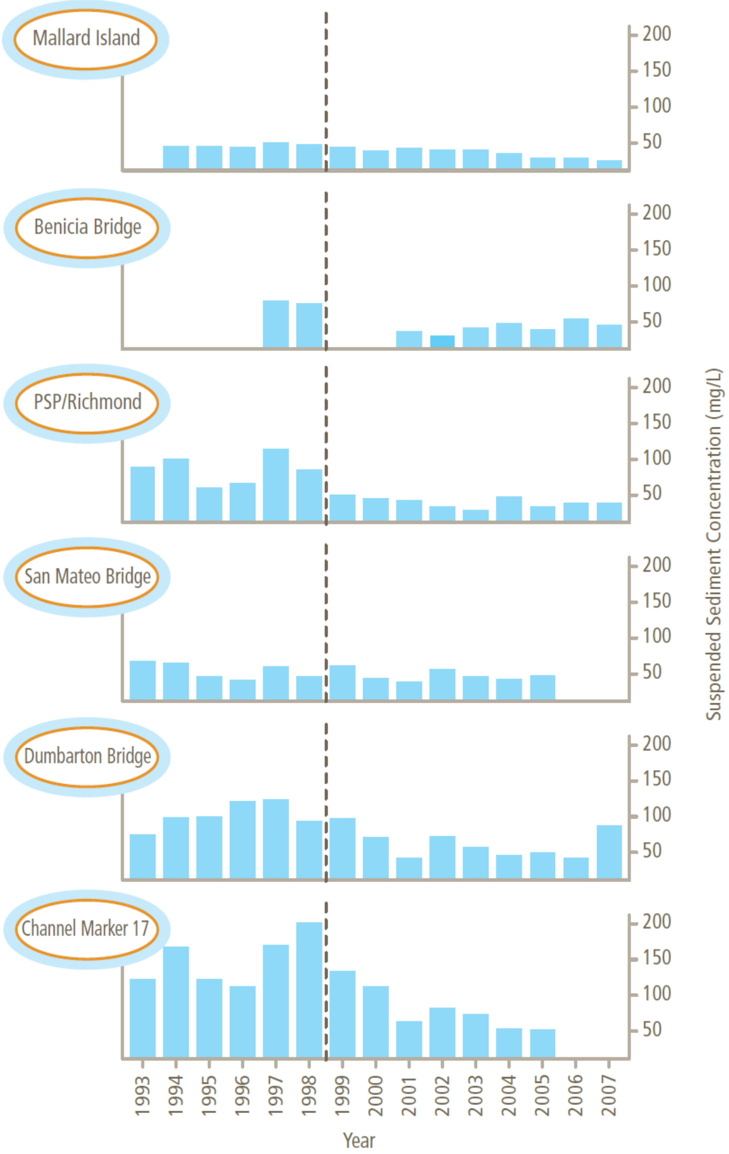


Figure 5.2.3. Trends in suspended sediment concentrations in San Francisco Bay (source Schoellhamer, 2009).

Table 5.2.1. Variation of chlorophyll a among estuary segments. Period 1999-2009. (Source: James Cloern, USGS: http://sfbay.wr.usgs.gov/access/wqdata).

Style	**Segment**	**Chlorophyll a ($\mu g\ L^{-1}$ equivalent to mg/m^3)**		
		Minimum	**Maximum**	**Mean**
River dominated	Rivers	0.4	14	2.6
	Suisun Bay	0.1	12	2.6
	Carquinez Straight	0.4	30	2.9
	San Pablo Bay	0.1	44	3.5
Oceanic	Central Bay	0.1	48	5.1
Lagoonal	South Bay (SF Bay Lower)	0.9	106	9.3

Intra- and Inter-annual Temporal Variability

Temporal variability in chlorophyll a and/or phytoplankton has been observed at scales ranging from hours to years (Cloern et al., 1985, 2000; Hogue et al., 2001; Cloern et al., 2003; Thompson et al., 2008) and to decades (Cloern et al., 2007; Jassby, 2008; Cloern et al., 2010). In general, both the North Bay and the South Bay experience low phytoplankton concentrations during the winter (December, January) and summer months (June, July) typically <5 $\mu g\ L^{-1}$ and greater concentrations during most spring periods (Figure 5.2.4). The blooms in the North Bay reach much lower peak concentrations than the blooms in the Central or South Bay and can be absent all together during years of low runoff (Cloern et al., 2000). Averaging the data since 1999, it is seen that the largest blooms occur in the South Bay during the spring (February to May inclusively; Figure 5.2.4), when river runoff sufficiently stratifies the water column and light penetrates more easily (Cloern et al., 2006). Phytoplankton biomass in the South Bay are characterized by strong intra-annual or within season variability; concentrations vary markedly between months over short time scales (Cloern and Jassby, 2010). That said, larger more prolonged blooms appear to last for 6 weeks or more during wetter years (e.g., 1993, 1995) reaching >70 $\mu g\ L^{-1}$, whereas blooms in drier years (e.g., 1991, 1992) lasted only 2 weeks and reached concentrations <20 $\mu g\ L^{-1}$ (Thompson et al., 2008).

Other than supply of nutrients and stratification, bivalve grazing appears to be a strong control on bloom magnitude, extent and longevity (Thompson et al., 2008). The seasonal absence of bivalve grazers in the winter months on the shoals sets up the potential for bloom each spring. Phytoplankton dynamics are strongly controlled by timing and recruitment process of bivalves, which in turn may be controlled by predation from fall migratory birds and fish (Thompson et al., 2008). However, ultimately bivalve biomass is also triggered and controlled by the available phytoplankton food resources; for example in 1995, bivalves that recruited on the shoals at the beginning of the bloom grew sufficiently in six weeks to control the shoal phytoplankton biomass (Thompson et al., 2008). This concept of coupled ecosystem capacity through the transfer of nutrients from phytoplankton biomass into secondary consumers, senescence and death, and recycling of regenerated nutrient back to primary producers was also discussed by Cloern (2007).

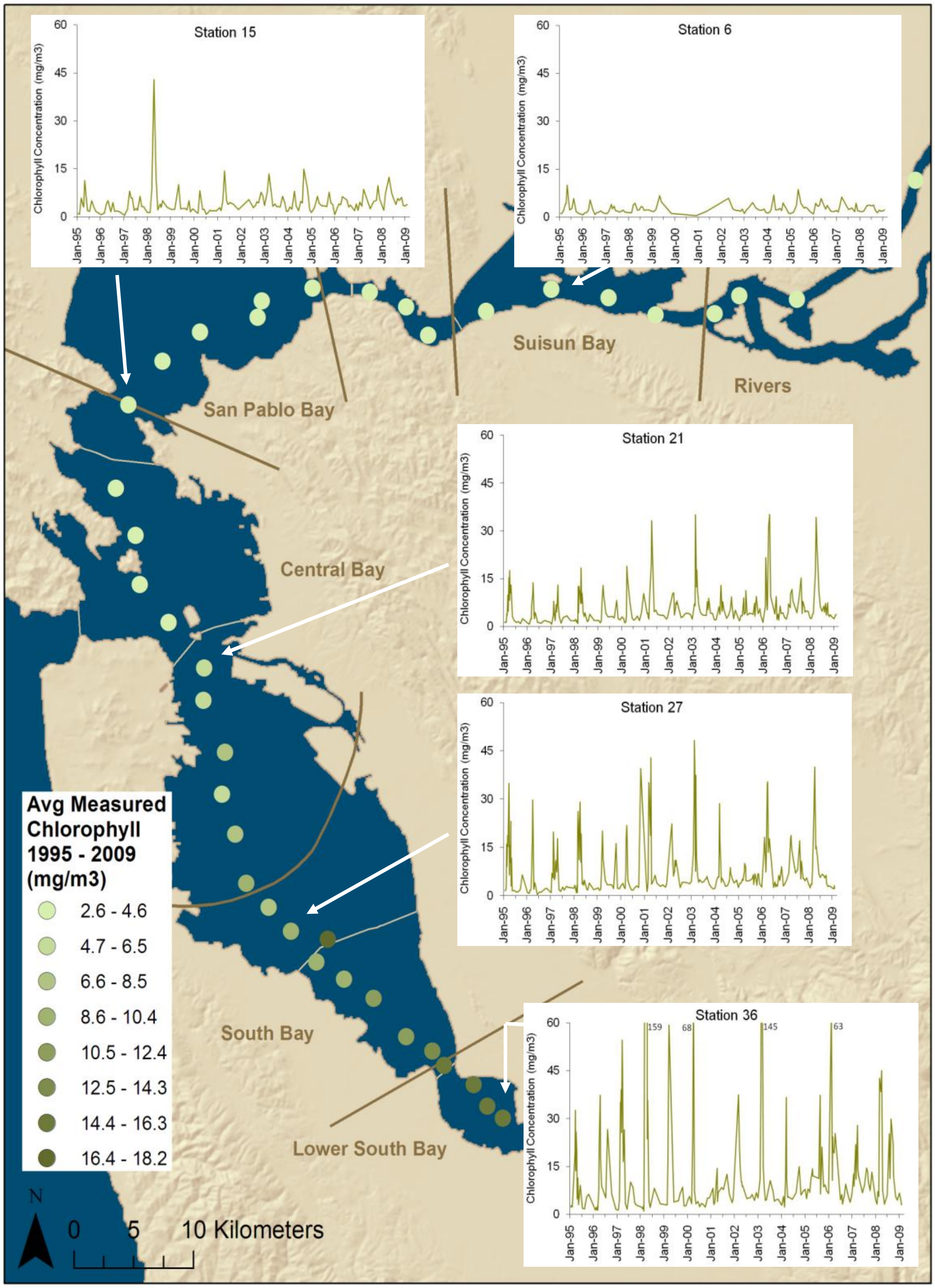


Figure 5.2.4. Seasonal chlorophyll a concentrations in San Francisco Bay based on monthly sampling by the USGS (Source: USGS: http://sfbay.wr.usgs. gov/access /wqdata).

Blooms in the North Bay occur when vertical salinity stratification occurs also in the late winter/spring (improving light penetration) and in the presence of low ammonium (Wilkerson et al., 2006). For similar reasons, the largest spring blooms occur after the wettest winters (e.g., 1998, 2003) (Figure 5.2.4, station 15 and station 6). In a similar fashion to the South Bay, averaging data collected since 1999 indicates a pattern of bloom in the spring and again in the fall (Figure 5.2.5).

Dugdale et al. (2007) summarized work to-date for the North Bay describing a conceptual model that includes a sequence of events that lead to blooms in the North Bay: 1) stabilization of the water column by stratification and or reduced tidal energy, 2) reduced NH4 concentrations (to a critical level below 4 µM) through dilution during runoff or by phytoplankton uptake, and 3) uptake (secondarily) of NO_3 autumn blooms are characteristically smaller than the spring blooms perhaps because phytoplankton does not deplete the ammonium enough to switch over to NO_3 uptake (Dugdale et al., 2007). In the spring, phytoplankton more often depletes the ammonium (especially in the North Bay) perhaps because ammonium in the Bay at this time is diluted by spring runoff or because ammonium regeneration is lesser than ammonium consumption by the growing bloom. Phytoplankton biomass in the North Bay is characterized by weaker variability between months but higher and dominant intra-annual variation in phytoplankton biomass (Cloern and Jassby, 2010), with some years exhibiting little bloom activity and other years having significant events. This strong inter-annual variability appears to be driven by variation in river runoff, the balance between ammonium and nitrate (in relation to sources that include wastewater, urban, and agricultural runoff; Wilkerson et al., 2006; Dugdale et al., 2007), and the introduction of the nonindigenous clam *Potamocorbula amurensis* (Alpine and Cloern, 1992).

Many models of phytoplankton mass in SF Bay have been developed over the past three decades. For example, Cloern and Cheng (1981) developed a pseudo-two-dimensional model to simulate the dynamics of a single dominant phytoplankton species in the North Bay. Using this model they were able to account for most of the variability of biomass as a function of light availability, temperature, salinity and copepod grazing; nutrients were deemed non-limiting. The model supported the premise that populations established over the shoals and were enhanced by reduced transport due to estuarine gravitational circulation. Later Lucas et al. (1998) presented a model for South SF Bay that included benthic grazing, zooplankton grazing, vertical phytoplankton sinking through a stratified water column, and respiration losses. They specifically did not incorporate nutrient availability since, in South SF Bay, bloom initiation was not thought controlled by nutrients; rather bloom termination can sometimes occur when nutrients are depleted although this still warrants further investigation (Thompson et al., 2008).

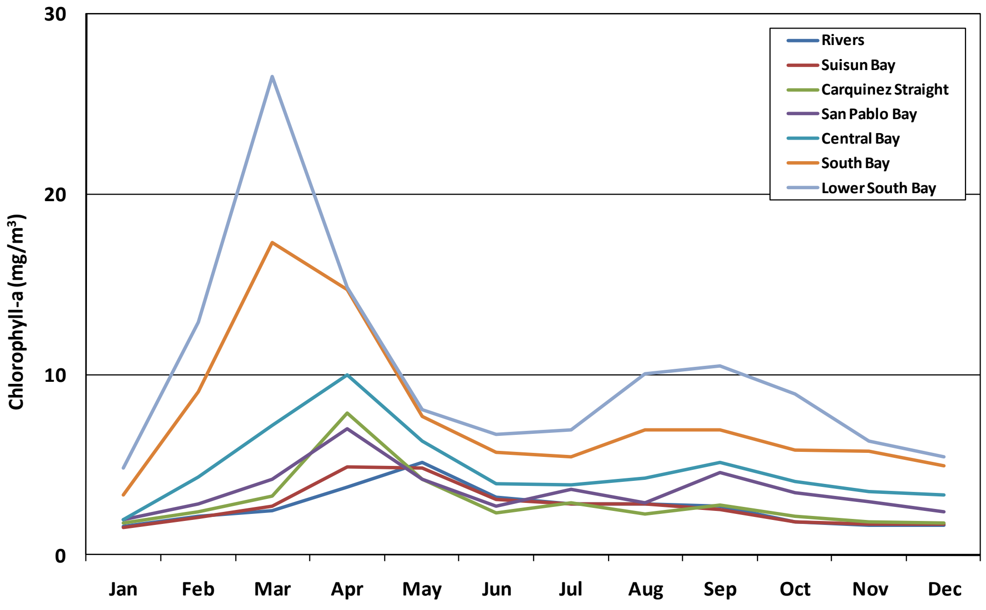


Figure 5.2.5. Average monthly chlorophyll a concentrations in San Francisco Bay based on monthly sampling by the USGS between January 1999 and February 2009 (Source: James Cloern, USGS: http://sfbay.wr.usgs.gov/access/wqdata).

In a later two-paper series, Lucas et al (1999 a, b) looked at the two main processes for governing bloom formation, 1) local mechanisms such as depth, light availability, and benthic grazing, and 2) transport related mechanisms which control the extent and distribution of the resulting bloom (Lucas et al., 1999a,b). They concluded that local conditions control the balance between phytoplankton loss and production and that initiation can occur in both shallow and deeper areas. They also pointed out that due to transport, greatest biomass may occur in areas that are not the most productive and vice-versa. Interestingly, again they did not include nutrients in the components of the models that simulated phytoplankton production, but in order to constrain peak biomass during long simulation times, an ad-hoc representation of nutrient limitation for bloom termination was included (Lucas et al., 1999a).

Lucas and Cloern (2002) explored the influence of tidal deepening and shallowing on phytoplankton production. They also assumed nutrients were not limiting and concluded that if tidal range is large relative to water depth, then tidal range may significantly influence net phytoplankton growth. Following on from this study, May et al. (2003) developed a coupled one dimensional model that simulated vertical and horizontal mixing processes to explore the impacts of turbidity on phytoplankton dynamics. Turbidity variation associated with wind strength was implicated as a control mechanism for the development of spring blooms in the South Bay. It was suggested that during years with high wind during the critical bloom period, phytoplankton productivity can be suppressed in contrast to years of lower wind (May et al., 2003).

The concept of the role of physical and biological processes in one sub-region controlling phytoplankton biomass and bloom production in an adjacent sub-region was explored using a coupled pseudo-two-dimensional model Lucas et al. (2009). They concluded that positive coupling occurs between productive shallow shoal areas and adjacent relatively unproductive deeper water channels. They also further supported the earlier premise that turbidity (May et al., 2003), benthic grazing (Thompson et al., 2008), and vertical density stratification (Lucas et al., 1999a,b; Thompson et al., 2008) control bloom occurrence, longevity, and spatial extent.

Most recently, it has been proposed that the cause of the annual autumn bloom might be attributed to sharp declines in bivalve mollusks (phytoplankton consumers) resulting from a trophic cascade caused by the onset of the East Pacific "cold phase" (Cloern et al., 2007). In the south Bay, Cloern et al. (2007) deduced that trends are not likely caused by changes in other reasonable factors alone such as nutrients (an observed decline), temperature (no change), stratification (no change), and turbidity (an observed weak increase) (note – this turbidity trend appears to contrast with data in Figure5.2.3: Schoellhamer, 2009). Cloern et al. (2007) argued instead that a 20-fold decrease in benthic water column filtering herbivores (e.g., *Corbula amurensis, Venerupis japonica, Musculista senhousia*, and *Mya arenaria*) has coincided with phytoplankton increases in southern areas of SF Bay in part caused by an collective increase in shrimp, crab, and sole biomass of about 4x (Cloern et al., 2007). An argument is now presented that the classic model of nutrient enrichment and light limitation as primary controls on phytoplankton in South SF Bay (and other estuaries) may be overshadowed by shifts in top-down control sometimes associated with connective shifts in sea surface temperatures and upwelling. In the case of SF Bay, sea surface temperatures and upwelling are a function of the Pacific Decadal Oscillation (PDO) and the broader ocean Basin (Smetacek and Cloern, 2008).

Decadal Scale Temporal Trends in Phytoplankton Biomass and Productivity

Long term monitoring data indicate decadal scale trends. Blooms are generally on the rise in the marine domains of the Bay with the exception being the River dominated Suisun Bay (Cloern et al., 2006) where *Corbula amurensus* is implicated as providing high grazing pressure (Alpine and Cloern, 1992) and ammonium may be inhibiting growth (Wilkerson et al., 2006; Dugdale et al., 2007). In an analysis of chlorophyll a concentrations since 1980, Cloern et al. (2006) showed that spring blooms since 1999 have been much larger than those prior to 1999, and that autumn-winter blooms are now occurring where they did not previously regularly occur, an observation they have called a "regime shift" (Cloern et al., 2006; 2007; 2010). In addition, baseline chlorophyll a concentrations have increased since the mid-1990s and these trends are significant year round in all locations from San Pablo Bay south. Suisun Bay and the Delta appear to be different (Jassby, 2008). Although overall since 1970 there has been a decrease on productivity in Suisun Bay and the Delta, since 1996 phytoplankton biomass appears to have stabilized in Suisun Bay and shown a positive increase throughout the Delta (Jassby, 2008). Beginning in 1999, the Bay began exhibiting autumn/winter blooms (September to December inclusively) (Figure 5.2.4), although these are generally have lower biomass than the annual spring bloom. In later years this annual autumn/winter bloom, although mainly comprised of diatoms, even included dinoflagellate red tides (Cloern et al., 2007) (see HABs discussion below). Increasing phytoplankton in the central and southern sectors of the Bay is manifested as increasing baseline concentrations of small cell plankton, increasing magnitude of spring blooms (larger cell diatoms), and occurrence of small cell autumn/winter blooms (Cloern et al., 2007).

While the causes of these majors changes are still being evaluated, over all, from San Pablo Bay to the lower South Bay, mean annual primary production has increased 75% over 1993-96 levels (Cloern et al., 2006). Carbon production by phytoplankton was estimated to be 200,000 US tons, or about 150 g C m^{-2} in 1980 (Jassby et al., 1993). At that time, the carbon budget of the south Bay was dominated by autotrophic production (92%); in contrast North Bay carbon was 68% allochthonous and supplied from Rivers (Jassby et al., 1993). Phytoplankton was responsible for 67% of the autochthonous production in the South Bay and 70% in the North Bay. Estimates of autochthonous total carbon production in 1993 - 1996 were about 120 g C m^{-2} (similar to the 1980 figure) and most recently production has increased again to an annual average of about 215 g C m^{-2} associated with both enhanced bloom and non-bloom biomass (Cloern et al., 2006). This has included a more than doubling of the autumn/winter (August-December) production from 32 g m^{-2} (pre-1998 mean) to 73 g m^{-2} (post 1998 mean). Based on an analysis of monthly trends, eight out of 12 months distributed across the whole year showed an upward trend (Cloern et al., 2007). Presently, a reanalysis of data is being completed to further evaluate summer trends. Preliminary data analysis conducted by Alan Jassby and James Cloern shows increasing chlorophyll a in South Bay, San Pablo Bay, and Suisun Bay since the mid-late 1980s at an average rate of 3-5% per year (James Cloern Personal Communication, March 2011). These new analyses provide further evidence that the Bay is changing, perhaps motivating further interest to understand the effects of nutrient loads and other co-factors.

Future trends are hard to predict. One hypothesis for the northern Bay (particularly Suisun Bay) is that any alleviation of the mechanisms currently limiting phytoplankton growth during the spring bloom, whether it is ever proven unequivocally what these mechanisms are, should lead to greater dominance of larger celled diatoms (Wilkerson et al., 2006). Because many of the HAB species common to the West coast are large celled (R. Kudela, personal communication March 2011), it is not clear whether additional factors may promote the dominance of HABs, including the toxic diatom genus Pseudo-nitzschia, versus non-harmful diatoms which better support Bay beneficial uses. In contrast, if the autumn bloom increases due to increased ammonium regeneration, phytoplankton species which have a preference for ammonium, including HAB species such as Pseudo-nitzschia and many toxic dinoflagellates, may become more prevalent (Kudela et al. 2010).

5.2.4 Phytoplankton Assemblage and Harmful Algal Blooms

The benefits of enhanced primary production during blooms are directly correlated with the species that dominate the bloom. Large cell diatom production tends to fuel the pelagic food web supporting zooplankton including jellyfish, filter feeding shell fish and crustaceans, fishes, and mammals including humans. In contrast, blooms of toxic smaller celled flagellates and some large-celled HAB species can suppress herbivores and impact beneficial uses for aquatic wildlife and humans (Cloern, 1996; Ning et al., 2000; Cloern et al., 2005b). This section covers two types of indicators: 1) assessment of health based on phytoplankton community structure and 2) abundance of HAB species and HAB toxin concentrations.

Phytoplankton Assemblage

San Francisco Bay contains over 500 phytoplankton taxa. Based on analysis using light microscopy, it appears that 10 and 20 species account for 77% and >90% of the total biomass respectively (Cloern and Dufford, 2005). Diatoms (Bacillariophyta) dominate the biomass making up 81% of the total cumulative biomass; dinoflagellates and cryptophytes (Pyrrophyta and Cryptophyta) made up 11% and 5% respectively (Cloern and Dufford, 2005). Cell sizes range between <3 and >100 um but in the nutrient enriched SF Bay system, large cells >30 µm contribute 40% of the biomass; attributed to the lack of a competitive advantage for smaller species. Like many nutrient enriched systems, SF Bay is characterized by a bloom-bust cycle of larger cell species periodically dominating a more stable community of small cell species (Hogue et al., 2001; Cloern and Dufford, 2005; Wilkerson et al., 2006); an observation attributed to the close coupling of small cell consumers in the microbial food web, the lagged response of metazoan consumers (Cloern and Dufford, 2005), and the take up of nitrate by larger cells (Wilkerson et al., 2006). Presently there is no explanation as to why diatoms dominate in SF Bay during blooms; hypotheses range from bottom up (inherently fast division rate, high N assimilation under high nitrate conditions, high growth rate in relatively low light conditions, ability to utilize bicarbonate) or a top down view (silica cell wall is better at resisting predation and/or buoyancy regulation allows avoidance of bottom dwelling filter feeders in shallow estuarine conditions).

In contrast there is a more constant crop of small cell picoplankton composed primarily of cyanobacteria and small eukaryotes *(Nannochloropsis sp., Teleaulax amphioxeia, Plagioselmis prolonga*) that occur across a wide range of salinities and seasonal conditions (Ning et al., 2000; Cloern and Dufford, 2005). Picoplankton make up <15% of the Bay biomass and <2% during blooms (Ning et al., 2000; Cloern and Dufford, 2005) and 11% of the total measured spatially and temporally averaged results for the whole North and South Bay combined. In relation to the possibility of using phytoplankton community structure as an ecological response indicator, some phytoplankton taxa (*Prorocentrum aporum, Coscinodiscus marginatus, Protoperidinium depressum, Eucampia zodiacus*) have not been seen since 1996 while others (*Protoperidinum bipes, Pseudo-nitzschia delicatissima, Scrippsiella trochoidea, Thalassiosira nodulolineata*) have appeared perhaps attributable to the Pacific Decadal Oscillation (PDO) (Cloern and Dufford, 2005).

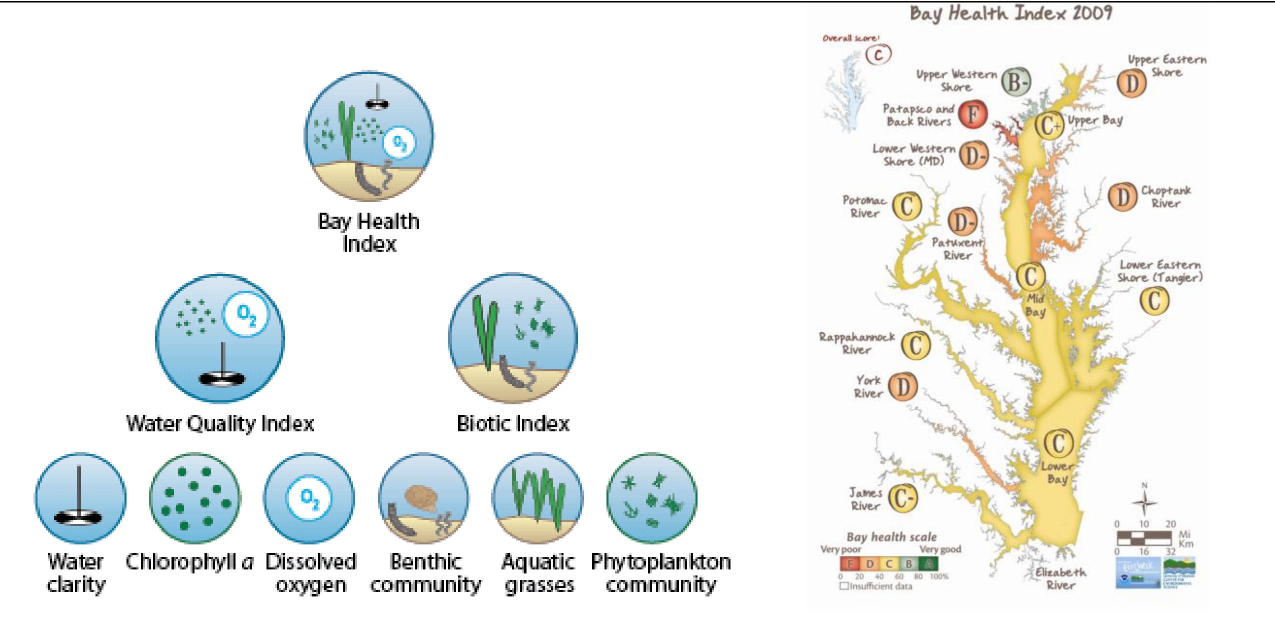
One use of data on phytoplankton community structure is to combine it into an index of biological integrity (IBI). IBIs are becoming more common for assessment of estuarine ecological condition and management focus in the face of physical and chemical transformation, habitat destruction, and changes in biodiversity (Borja et al., 2008). An IBI describes the biological condition of an assemblage of plants or animals, typically based on the diversity and relative abundance of species or the presence or absence of pollution tolerant species. A key element of developing an IBI is the ability to describe the community response of the assemblage (e.g., benthic invertebrates, phytoplankton, etc.) along gradient of physical or chemical stress from minimally disturbed or “reference state” to highly disturbed. IBIs are most commonly used in stream bioassessment, but several examples exist for estuarine environments as well including submersed aquatic vegetation (Dennison et al., 1993; Corbett et al., 2005), benthic macroinvertebrates (Weisberg et al., 1997; Graves et al., 2005), fish populations (Deegan et al., 1997;

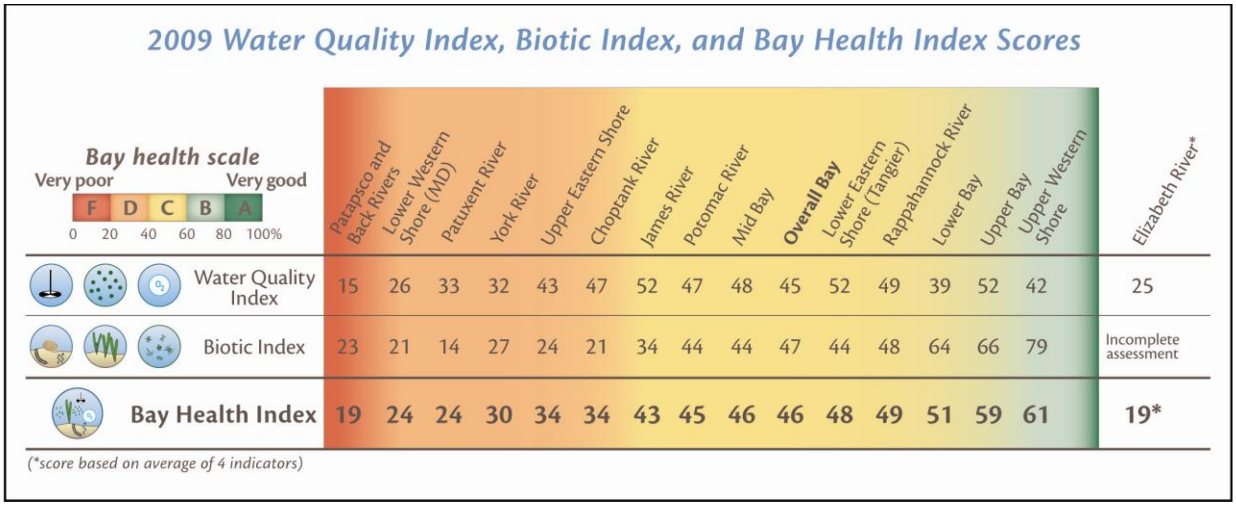
Bortone et al., 2005), zooplankton (Carpenter et al., 2006), micro-algae (Paerl et al., 2005) and phytoplankton (Lacouture et al., 2006).

IBIs developed and used in Chesapeake Bay present an example of how phytoplankton community structure data can be synthesized to provide information about the ecological health of the Estuary and about the ability to support specific beneficial uses. A Phytoplankton Index of Biotic Integrity (P-IBI) was developed in Chesapeake Bay using an 18 year data set (Lacouture et al., 2006). Thirty-eight phytoplankton metrics were tested for their ability to discriminate between impaired and least-impaired habitat conditions. Twelve discriminatory metrics were chosen from a tested set of 38 to discriminate between impaired and least-impaired habitat conditions. Combinations of these twelve metrics were scored and used to create phytoplankton community indexes for spring and summer in the four salinity regimes. The P-IBI, thus developed, combined the scores of pollution-sensitive, biologically important metrics of the phytoplankton community into a single index. Like other multi-metric indexes, the P-IBI is more sensitive to habitat conditions than its component metrics, which include chlorophyll a, the abundances of several potentially harmful species, and various indicators of cell function and species composition (Lacouture et al., 2006).

Following on from the work of Dennison et al. (1993) on the use of submerged aquatic vegetation (SAV), Carpenter et al. (2006), who developed an IBI for zooplankton, and Lacouture et al. (2006) on the development and testing of a P-IBI for the Chesapeake, a Bay Health Index (BHI) that combined three water quality and three biological measures was developed to assess the ecological effects of nutrient and sediment loading in Chesapeake Bay (Williams et al., 2009). A Water Quality Index (WQI) was generated by averaging concentrations of chlorophyll a, dissolved oxygen, and Secchi depth. A P-IBI and B-IBI was developed from the biological measures of the phytoplankton and benthic community composition and combined with the area of SAV to create the Biotic Index (BI). The WQI and BI were then averaged to give a BHI for the growing season (March–October) (Figure 5.2.6; http://www.eco-check.org/reportcard/chesapeake/2009/). Least impaired regions of Chesapeake Bay exhibited low chlorophyll a, high dissolved oxygen, greater transparency, higher phytoplankton and benthic indices relative to ecological health-based thresholds, and greater SAV area. All three indexes were significantly correlated with nitrogen (N), phosphorus (P) and sediment loads and the sum of developed and agricultural land use. The BHI is used annually to track progress as part of the annual environmental report card.

The development of multi-metric indexes of estuarine quality are not without challenges which include the formation of multidisciplinary scientific teams and stakeholder groups that are committed to the outcome more than representation of their individual interest, long term multi-parameter data sets on a wide range of biotic and abiotic indicator species, co-factors, and stressors, and empirically demonstrated and perhaps modeled cause and effect relationships that can demonstrate trends with a high signal to noise ratio. Following from the example set in Chesapeake Bay (Carpenter et al., 2006; Lacouture et al., 2006; Williams et al., 2009; Williams et al., 2010); it would seem that SF Bay, with its rich multi-parameter long term data sets, may be a suitable living laboratory to develop such an index.





2009 Water Quality Index, Biotic Index, and Bay Health Index Scores

Bay health scale
Very poor — Very good
F D C B A
0 20 40 60 80 100%

	Patapsco and Back Rivers	Lower Western Shore (MD)	Patuxent River	York River	Upper Eastern Shore	Choptank River	James River	Potomac River	Mid Bay	**Overall Bay**	Lower Eastern Shore (Tangier)	Rappahannock River	Lower Bay	Upper Bay	Upper Western Shore	Elizabeth River*
Water Quality Index	15	26	33	32	43	47	52	47	48	45	52	49	39	52	42	25
Biotic Index	23	21	14	27	24	21	34	44	44	47	44	48	64	66	79	Incomplete assessment
Bay Health Index	**19**	**24**	**24**	**30**	**34**	**34**	**43**	**45**	**46**	**46**	**48**	**49**	**51**	**59**	**61**	**19***

(*score based on average of 4 indicators)

Figure 5.2.6. The Chesapeake Bay Report Card. The report card aims to provide a transparent, timely, and geographically detailed annual assessment of Chesapeake Bay health. (See Chesapeake EcoCheck: Assessing and Forecasting Ecosystem Status. http://www.eco-check.org/reportcard/chesapeake/2009/).

Harmful Algal Blooms and Toxins

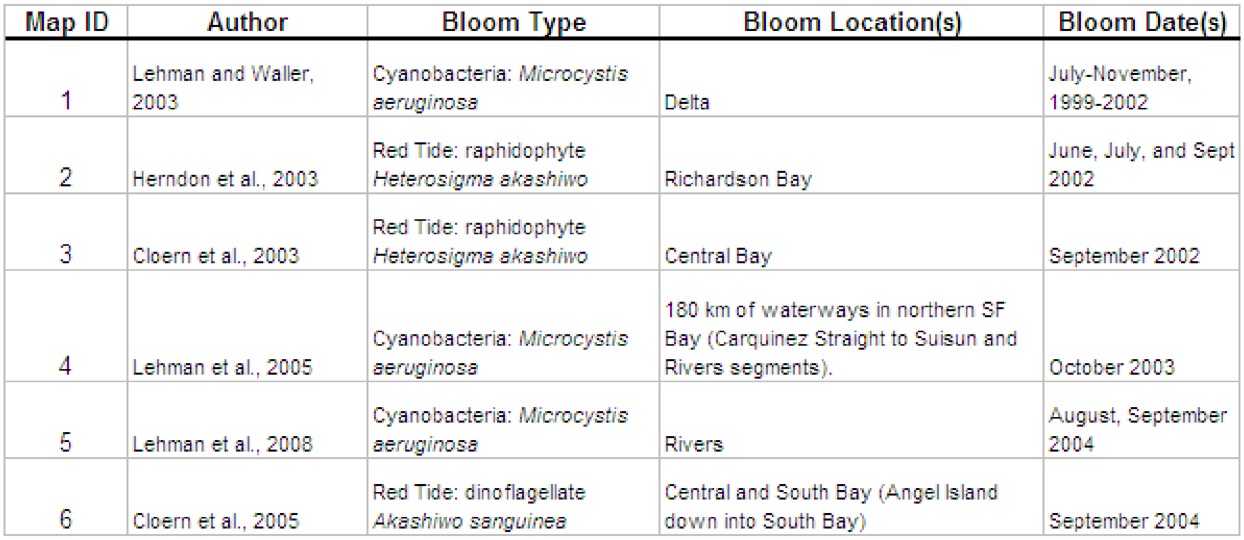
Harmful algal blooms are blooms of phytoplankton, algae, or cyanobacteria that can produce potent toxins, nuisance levels of biomass, and suppress oxygen causing harm to humans, fisheries resources, and coastal ecosystems. While increased anthropogenic nutrients increase the potential for HAB development, the conversion of nutrients into biomass is dependent on other factors including clarity, temperature, stratification, and seed populations (Cloern et al., 2005b).

Despite the persistent nutrient enriched status of San Francisco Bay, few harmful algal blooms (HABs) have been reported recently in San Francisco Bay, apparently because nutrient enriched turbid conditions in the estuary favor larger celled diatoms associated with new production as opposed to nutrient regeneration (Cloern, 1996; Ning et al., 2000). A lack of monitoring may also play a role, given the large number of potentially harmful algae present in San Francisco Bay (Cloern and Dufford, 2005). However, there have been historical occurrences (see Cloern et al., 1994 referenced in Cloern, 1996), and recently cyanobacteria and dinoflagellate blooms have been documented. For example, blooms of the cyanobacteria *Microcystis aeruginosa* have been occurring in the late summer/autumn in the northern reaches of the Bay since 1999 (Lehman et al., 2005), the raphidophyte *Heterosigma akashiwo* created a red tide in the Central Bay in summer 2002 (Herndon et al., 2003), and the dinoflagellate *Akashiwo sanguinea* caused a red tide in the Central and South Bay areas during September 2004 (Cloern et al., 2005a; Table 5.2.2; Figure 5.2.7). The conditions under which these blooms occurred are presented in greater detail below.

Microcystis aeruginosa blooms have occurred in the Delta and the North Bay during July through November of each year since 1999. The colonial form of *M. aeruginosa* is the first recorded toxic phytoplankton bloom in the northern reach of SF Bay and may have been recently introduced because it was not recorded in historic samples taken between 1975 and 1982 (Lehman and Smith, 1991 in Lehman et al., 2005). *M. aeruginosa* can form surface scums and is a nuisance to recreational users, reduce aesthetics and oxygen and can produce microcystin, a hepatoxin to humans and wildlife (Lehman and Walker, 2003; Lehman et al., 2005; Lehman et al., 2008). Concentrations found at Benicia, in Suisun Bay, and at Chips Island were low relative to upstream locations (Lehman et al., 2005) perhaps because of dilution or cell death at higher salinities (Lehman et al., 2008). Blooms occurred at salinities less than 18 ppt, although growth was probably limited to <7 ppt (Lehman and Walker, 2003; Lehman et al., 2005; Lehman et al., 2008).

Several surveys of *M. aeruginosa* blooms have documented that the blooms can be widespread, often with microcystin concentrations that exceed World Health Organization guidelines for risks to humans and wildlife (e.g., Lehman and Walker, 2003; Lehman et al. 2005; Lehman et al., 2008). For example, Lehman et al. (2005) documented that an extensive *M. aeruginosa* bloom was found to extend 180 km from Benicia to near Rio Vista on the Sacramento River to 20 km downstream from Tracy on the San Joaquin River side of the Delta, with toxicity exhibited at all stations. Concentrations of microcystin were measured in greater concentrations in zooplankton and clam tissue relative to algal tissue although concentrations were not greater than lethal limits known to cause acute death (Lehman et al., 2005; Lehman et al., 2008). This appears to support the hypothesis that microcystin are transferred or perhaps biomagnified in the food web, the exceptions being clams which appear to be able to depurate toxins from their tissue rapidly (Lehman et al., 2008). However, concentrations they found may be chronically obstructive to food quality, feeding ability, growth, and fecundity in zooplankton (Lehman et al., 2008). Given *M. aeruginosa* seems to prefer high light and warm shallow water eutrophic conditions, any change in the management of the flows from the Sacramento River that leads to increased or more persistent but steady flow rate and improved salinity stratification may expand the population in the late summer/autumn. Given the potential threats to humans and wildlife, Lehman et al. (2005) recommended annual monitoring and further assessment of the causes and controls on this species.

Table 5.2.2. Reported harmful algal blooms in San Francisco Bay since 1995 (See Figure 5.2.5 for approximate locations and extent of blooms).



Map ID	Author	Bloom Type	Bloom Location(s)	Bloom Date(s)
1	Lehman and Waller, 2003	Cyanobacteria: *Microcystis aeruginosa*	Delta	July-November, 1999-2002
2	Herndon et al., 2003	Red Tide: raphidophyte *Heterosigma akashiwo*	Richardson Bay	June, July, and Sept 2002
3	Cloern et al., 2003	Red Tide: raphidophyte *Heterosigma akashiwo*	Central Bay	September 2002
4	Lehman et al., 2005	Cyanobacteria: *Microcystis aeruginosa*	180 km of waterways in northern SF Bay (Carquinez Straight to Suisun and Rivers segments).	October 2003
5	Lehman et al., 2008	Cyanobacteria: *Microcystis aeruginosa*	Rivers	August, September 2004
6	Cloern et al., 2005	Red Tide: dinoflagellate *Akashiwo sanguinea*	Central and South Bay (Angel Island down into South Bay)	September 2004

Red tides associated with a bloom of *Heterosigma akashiwo* have occurred in Richardson Bay (Herndon et al., 2003). Three bloom events were observed in northern Richardson Bay during the summer and autumn of 2002 and all coincided with clear skies, warm air temperatures >25ºC, and calm and warm (>20°C) waters (Herndon et al., 2003). The blooms were a near monoculture with other species comprising <7% of the samples (by cell count) (Herndon et al., 2003). A fourth bloom occurred between September 1 and 12 and covered a wider geographic area including most of the coastline of Tiburon Peninsular over to the Berkeley frontage (Herndon et al., 2003). That same year it was identified by O'Halloran et al., (2006) at the Berkeley pier in April and September. This harmful alga is a new occurrence and has been associated with fish kills in other coastal ecosystems (Cloern et al., 2003). In this case it was widespread outside of the Golden Gate with similar reports at Stinson Beach and in Bodega Bay. Although there was some evidence that the bloom was seeded from the near-field ocean, it is not clear what other factors including nutrients supplied from terrestrial sources, turbulence, and temperature played in bloom sustenance and degradation.

Figure 5.2.7. Harmful algal bloom (HAB) occurrences reported in the literature since 1995. Large segments show locations of HABs as reported in the literature (usually from a diagram) and small segments indicate general location of HAB in which more descriptive location information was not provided in the literature. Locations are approximated based on location description in the referenced journal publication.

Red tides associated with a bloom of the dinoflagellate species *Akashiwo sanguinea* have been observed in the southern area of Central Bay (September 2004). This species is not normally described as a HAB but can be potentially disruptive if biomass is high and is indicative of recent increases in red tides on coastal California (Kudela et al., 2008). An advantage of this and other dinoflagellates is their ability to move lower in the water column to feed on nutrients during the night hours and reside within the photic zone during daylight. The bloom, which had chlorophyll a concentrations approaching 200 µg L^{-1}, coincided with unusually weak neap tides, calm winds, and four consecutive high air temperature days creating a shallow (<3 m) surface layer above a thermocline that persisted long enough for the motile dinoflagellate species to proliferate (Cloern et al., 2005a,b). This bloom reduced ammonium and then nitrate concentrations to some of the lowest concentrations measured (Cloern et al., 2005a,b; Wilkerson et al., 2006). While climatic conditions were intimated as the key factor in bloom development, the bloom followed a summer of weak coastal upwelling and high dinoflagellate biomass in nearfield coastal waters, apparently providing seed organisms (Cloern et al., 2005b).

A common theme emerging from observations of all recent HAB blooms have been there occurrence in the summer and autumn months, perhaps associated with the decline of the spring and summer diatom blooms and consumption of regenerated nutrients. If blooms become more common and magnitude increases, the occurrences of hypoxia/anoxia may also rise in relation of higher punctuated organic matter loading and resulting biological oxygen demand (BOD).

5.2.5 Utility of Phytoplankton Biomass, Productivity, and Community Composition as an NNE Indicator for San Francisco Bay

Clear Linkage to Beneficial Uses

Phytoplankton has a well-documented linkage to beneficial uses of SF Bay. Phytoplankton are the dominant primary producer in SF Bay, and therefore the ultimate source of carbon for the entire food web (Cole and Cloern 1982). Food supply is smaller than in many other estuaries (largely because SF Bay is turbid) and, as a result, consumers such as zooplankton, mysid shrimp, and clams are limited by carbon productivity (Cloern et al., 2003). Only during blooms is the rate of carbon production sufficient in SF Bay to keep pace with consumption (Cloern, 1996). Additionally there is compelling evidence of the linkage between phytoplankton and the pelagic foodweb. For example, there was a remarkable change in phytoplankton post 1986 in Suisun Bay when the *Potamocorbula amurensis* (now called *Corbula amurensis*; Coan, 2002, referenced in Wilkerson et al., 2006) was introduced. The summer bloom was decimated and primary production decreased around 2.5-fold from 106 g C m^{-2} to just 39 g C m^{-2} (Alpine and Cloern, 1992) arguably directly caused by increased consumption faster than phytoplankton reproduction by the invasive clam and perhaps leading to massive failures in several competing pelagic organisms, the copepod *Eurytrmora affinis* and the native mysid shrimp *Neomysis mercedis* (Orsi and Mecum, 1996). Note that Wilkerson et al. (2006) more recently argued that grazing could not be the dominant cause of low phytoplankton in Suisun Bay because surface growth rates are an order of magnitude less than clam pumping rates, the similarity of NH_4 uptake rates between Central, San Pablo, and Suisun Bays despite differences in clam populations, and the fact that the clam population is depressed during the spring bloom period. There is also evidence that phytoplankton biomass is linked to water clarity (May et al., 2003). Recently, Schoellhamer (2009) provided evidence that all regions of the Bay are showing decreasing trends in turbidity mainly associated with declines in suspended sediment loads (McKee et al., 2006). This is likely one factor that is contributing to increasing trends in primary productivity.

Although there is clearly complexity, these studies provide a broad base of evidence that phytoplankton have a direct linkage to important SF Bay beneficial uses, including food web support for marine and estuarine aquatic organisms (EST, MAR) including the commercial and sport fisheries (COMM), shellfish such as clams, oysters and mussels (SHELL and AQUA), migratory (MIGR) birds and fish, support for fish nursery habitat (SPAWN). Harmful algal blooms can adversely affect the health of humans (REC-1) by irritation and injury to recreational swimmers, sailboarders, and boaters (Lehman et al., 2005). In addition, elevated phytoplankton biomass could impact estuarine and wildlife habitat by shading and degrading eelgrass habitat and impact aesthetics (REC-2) through nuisance buildup and smell during decay.

Predictive Relationships to Causal Factors

Use of phytoplankton as an NNE indicator for SF Bay requires the ability to develop a predictive model that links phytoplankton response variables back to nutrient loads and other causal factors. Specifically, this requires, at minimum, the development of models that establish the relationship between nutrient loads and phytoplankton response (biomass, productivity, or assemblage). These models can be empirical or computer spreadsheet or dynamic simulation models.

There has been some success in relating empirical phytoplankton to both external nutrient loads and *in situ* nutrient concentrations in some estuaries, particularly when data are averaged over annual time periods. Table 5.2.3 shows relatively high correlation coefficients published by various authors for both phytoplankton biomass and production. In general, variations in N loading rates are reflected in concentrations of N in receiving water bodies, particularly when the residence time of that water body is long (on the order of weeks). Although many processes act at various rates to modify nutrient concentrations, mean total nitrogen (TN) concentrations are significantly correlated to TN loading for 5 sub-systems of Chesapeake Bay averaged over a decadal period (Boynton and Kemp 2008). Conley et al. (2000) reported that on an annual basis about 70% on the variation in TN concentration could be explained by variation in TN loads in a large sample of Danish estuaries. Madden et al. (2010) found a strong correlation between SEAWIFS remotely sensed chlorophyll a and TN loading for 108 estuaries in the United States. A survey of the fundamental nutrient forms and processes in several major estuaries was performed by Smith (2006) using data from 92 estuarine and coastal sites worldwide. It demonstrated a strong correspondence between log transformed annual mean concentrations of total P and N and standing stock of chlorophyll a.

Table 5.2.3. Modeled relationships between nutrient loading and phytoplankton response in world estuaries. (From Boynton and Kemp, 2008).

Location	Variable, X (units)	Variable, Y	r^2 / n	Reference
Multiple estuaries	TN-loading (g N m^{-2} y^{-1})	Phytoplankton Production	0.60 / 14	Boynton et al. 1982
SF Bay and other estuaries	Composite parameter X = f(B, Z_p, I_0)		0.82 / 211	Cole and Cloern 1987
Narragansett Bay	Composite parameter X = f(B, Z_p, I_0)		0.82 / 1010	Keller 1988
Multiple estuaries	DIN-loading (mol N m^{-2} y^{-1})		0.93 / 19	Nixon et al. 1996
Multiple estuaries	TN-loading (g N m^{-2} y^{-1})		0.36 / 51	Borum and Sand-Jensen 1996
Boston Harbor	Composite parameter X = f(B, Z_p, I_0)		0.66 / 12	Kelly and Doering 1997
Waquoit Bay system	Annual average DIN conc (μM)		0.61 / 12	Valiela et al. 2001
Chesapeake Bay	TN(x_1), TP(x_2) load (kg mo^{-1})		0.67 / 11	Harding et al. 2002
Multiple estuaries	DIN (mM m^{-3}); tidal range (m)	Phytoplankton Biomass	na / 163	Monbet 1992
Multiple systems / MERL	DIN input (mmol m^{-3} y^{-1})		na / 34	Nixon 1992
Ches Bay mesohaline	River flow (m^3 d^{-1}); proxy for N-load		0.70 / 34	Harding et al. 1992
Maryland lagoons	TN load (g N m^{-2} y^{-1})		0.96 / 9	Boynton et al. 1996
Danish coastal waters	TN concentration (ug l^{-1})		0.64 / 168	Borum 1996
Canadian estuaries	TN concentration (ug l^{-1})		0.72 / 15	Meeuwig 1999
Ches Bay and Tributaries	TN Load; (mg N m^{-2} yr^{-1}) (R_{time}, yrs)$^{-1}$		0.82 / 17	Boynton and Kemp 2000
Danish estuaries	TN concentration (ug N l^{-1})		0.30 / 1347	Nielsen et al. 2002

However, San Francisco Bay has long been recognized as an estuary in which phytoplankton biomass and pelagic primary productivity is not driven by simple nutrient-limitation, due to a variety of co-factors that modulate primary producer response to nutrients (Figure 5.2.8, Cloern and Dugdale, 2010). Substantial effort has gone into the development of empirical relationships between phytoplankton and causal indicators in SF Bay (Cloern and Cheng, 1981; Lucas et al., 1998; Lucas, et al., 1999a,b; Lucas and Cloern, 2002; May et al., 2003; Thompson et al., 2008; Lucas et al., 2009). Typically the basis of the models has been temperature, light (surface irradiance and photic depth), stratification, predation, and senescence. The premise of Cole and Cloern (1987) that 80% of the spatial and temporal variability in productivity is correlated with variations in three easily measured parameters (phytoplankton chlorophyll a, photic depth, and surface irradiance) seems to largely hold true. In addition to these main factors, modelers have explored other cofactors such as turbidity and transport. In no single case have the authors used nutrient concentrations or external loads in the bloom initiation components of the models, however, the limitation of phytoplankton biomass by nutrients deserves more study (Thompson et al., 2008).

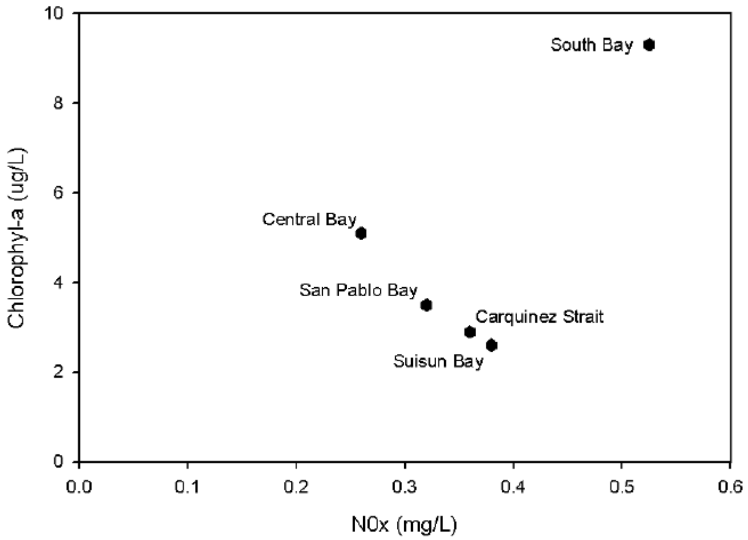


Figure 5.2.8. Mean chlorophyll a concentrations as a function of mean nitrate+nitrite (NOx) concentrations in San Francisco Bay for the period January 1999 to February 2009 (Source: James Cloern, USGS: http://sfbay.wr.usgs.gov/access/wqdata).

From nearly four decades of research, a number of common themes have emerged about the relationships between phytoplankton production and causal or limiting factors: 1) Light limitation plays a role in bloom production and magnitude in all reaches of the Bay; if turbidity continues to decrease, overall production will likely increase; 2) in all areas of the Bay, nitrate and phosphate never limit the initiation of spring bloom phytoplankton production; 3) in all areas of the Bay, predation by bottom dwelling filter feeders limit the maximum bloom magnitude as can depletion of nitrate more occasionally; 4) in the North Bay, the ratio of ammonium to nitrate along with salinity based stratification and the magnitude of freshwater outflow appear to control initiation of spring blooms whereas in the South Bay, the timing and magnitude of blooms appears to be controlled by the rate of recruitment of macro-invertebrates in the late winter and spring; and 5) greater winter runoff (and nutrient supply) can increase the size of spring blooms in both the North and South Bays, however, the occurrence is controlled by other factors; this implies that if at some future time blooms cause impairment of the Bay, nutrient load controls may only have the potential to bring the Bay back to its current “2010” condition. Another general theme that has emerged is the role of macro-invertebrates as both response variable as a secondary consumer that bloom in response to availability of planktonic food resources, and as a cofactor causal variable where the presence or absence of winter populations can strongly influence bloom initiation.

Northern SF Bay (North Bay) has undergone radical changes in ecosystem structure, from nutrients to phytoplankton and up the food chain to zooplankton and fish. Although much of these changes have been ascribed to the appearance of invasive species, e.g., of zooplankton and bivalves, a new analysis of the long term time series for Suisun Bay has provided a bottom-up view of the causes of these changes based on principles of ecological stoichiometry (Glibert, 2010). Glibert describes an initial change from a diatom-based foodweb that began in 1982 when the Sacramento Regional Sewage Treatment plant came on line and converted to secondary treatment, releasing NH4 into the Sacramento River, eventually discharging 15 tons N daily by 2002 (Jassby, 2002). Phytoplankton blooms became rare in Suisun Bay after 1987 coincident with the arrival of the Asian clam, Corbula amurensis. However, the diatom population had been declining for the previous 5 years, now believed to be the result of increasing NH4 input (Wilkerson et al., 2006; Dugdale et al., 2007; Glibert, 2010). The composition of the zooplankton also changed as the phytoplankton community became dominated by Chryptophytes and Flagellates and the growth rate of the delta smelt declined. In this scenario, the causal agent was the conversion of the Sacramento River from a nitrate-based diatom phytoplankton system, to an NH4 – based, small phytoplankton, small zooplankton (*Eurytemora, Pseudodiatomus, Limnothiona*, and the introduction of the invasive clam which came to dominant phytoplankton blooms in some times of the year (not in spring when clam populations are seasonally low). The next major perturbation in the northern estuary nutrient regime was the decline in phosphate, another anthropogenic effect as phosphate was removed from detergents on the basis of fear of eutrophication. The northern estuary then became an enriched NH4-N, low P ecosystem, which now favored the development of Cyanobacteria, in particular Microcystis blooms began to occur, as that group is able to grow well at low P concentrations and compete with the Cryptophyte and Flagellate functional groups, diatoms having been eliminated by the blocking of access to nitrate by high NH_4 concentrations. These three stages in the degradation of the Suisun Bay ecosystem are diagrammed (Figure 5.2.8).

Although the changes described above are well documented, there is no consensus among the scientific community that these changes can be attributed to a single factor, such as wastewater inputs of ammonium. For example, recent analyses of population declines of pelagic fish and their food resources in the upper estuary show strong associations with changes in water clarity, export flows, and salinity distribution measured as X2 (Mac Nally et al., 2010; Thomson et al., 2010). The conclusion of Dugdale et al., 2007, that the northern SF estuary is locked most of the time in an NH_4-based low-primary productivity condition (due to NH_4 control of access to the larger N pool, NO_3) is not incompatible with changes in productivity driven by other factors, e.g., in the NH_4 based system, an increase in transparency will result in an increase in the depth integrated primary productivity, and other modulating factors will also come into play to increase or decrease productivity. Therefore, the SF technical advisory team views that much remains to be learned about the ecological consequences of ammonium enrichment in the context of other drivers of population declines, and recommends formation of a working group to review the relevant data and identify and evaluate potential ammonium endpoints.

With respect to South Bay, most recently, Cloern et al. (2010) presented an analysis of the influence of the Pacific Decadal Oscillation (PDO) on biomass trends in SF Bay. They found that populations of demersal fish, crabs, and shrimp covary with PDO and the North Pacific Gyre Oscillation (NPGO) accounting for upwards of two thirds of the variability. They argue in this and their previous paper

(Cloern et al., 2007) that population decreases in these predators after 1999 were followed by declines in bivalve suspension feeders, and increasing abundance of phytoplankton biomass. It appears that at the scales of decades, the supply of nutrients may not be strongest driver of productivity in an estuary where nutrients are not limiting; the existence of a unique and long term phytoplankton time series for SF Bay appear to suggest responses on the decadal scale to climate variability.

In order to capture the complexity of these relationships, the consensus among the SF Bay NNE Technical Team is that computer models are required to predict watershed, airshed loadings and oceanic exchange with SF Bay and models that simulate response of the Bay to nutrient loads and other factors. Dynamic simulation models are mathematical representations of the real world that estimate environmental events and conditions. Models can be used to predict pollutant delivery as well as simulate how various changes or pollution-reduction actions could affect a waterbody's beneficial uses, especially with respect to water quality, aquatic life, and wildlife. Because estuaries and their watersheds are typically complex, scientists and managers can rely on computer models to synthesize information about the ecosystem's characteristics and the effects of various environmental actions to reduce pollution. To-date, no comprehensive predictive calibrated model exists for the Bay or the watershed, airshed or ocean that is able to couple forcing factors, co-factors, and biological response. Instead, these models have tended to support empirical observations

The conceptual approach to development of models for the SF Bay estuary could be similar to that done for the Chesapeake Bay Estuary (http://archive.chesapeakebay.net/pubs/backgrounder_CBP_Models.pdf), in which models were developed and refined through a 30 years of collaboration by federal, state, academic and private partners. Initially, two types of models would need to be developed:

1) Watershed, airshed, and oceanic exchange model, which incorporates information about loadings or exchanges from land use, fertilizer applications, wastewater plant discharges, septic systems, wet and dry air deposition, exchange with the coastal ocean, weather and other variables to estimate the amount of nutrients and sediment reaching the SF Bay estuary and where these pollutants originate. The watershed model would include three components:

    - A hydrologic sub-model that uses rainfall, evaporation and meteorological data to calculate runoff and sub-surface flow for all land uses.
    - A non-point source sub-model, which simulates soil erosion and nutrient loads from the land to rivers, driven by the hydrologic sub-model
    - A river sub-model which routes flow and associated nutrient loads from the land through lakes, rivers and reservoirs to the Estuary.
    - An ocean exchange submodel can be used to force exchange of flow, chemical and biological constituents across the estuarine-oceanic boundary.

2) Estuary water quality model, which simulates the ecosystem response to pollutant loads, which would consist of two sub-models

- A hydrodynamic sub-model that will simulates the exchange with rivers, oceans, mixing of waters in the Estuary and its tidal tributaries.
- A water quality sub-model that simulates the Estuary's biological, chemical and physical dynamics in response to nutrient loads and other factors (light, temperature, grazing, etc.).

The models would be used to establish load allocations of nutrients that the SF Bay estuary can sustainably assimilate. It would also be used to generate simulations of the past, present or future state of the Estuary, ocean, watershed, and airshed (e.g., population growth, climate change, etc.) to explore potential effects of management actions and evaluate alternatives. Thus these models would be a key component of a strategy to adaptively manage SF Bay.

Ideally sufficient data and knowledge of SF Bay should exist to support the development of system wide dynamic simulation models to predict phytoplankton biomass/community response and relationships to models of secondary productivity. At the macro scale, the relationship between phytoplankton based primary production is a given; phytoplankton need nutrients to grow; nutrient loads to each Bay compartment, the standing nutrient mass, and speciation exert a strong control on primary production and are primary causal variables. However, as shown in Figure 5.2.8, the relationship for the Bay is complex. Empirical models as well as any subsequently developed dynamic simulation models will need to capture this complexity in order to be of use in a management context.

Unfortunately, there are some data sets that are less well developed that may also hamper the development of predictive models and a fuller understanding of the causes of change. For example, we have no reliable annual estimates of nutrient loads from either the Central Valley or local tributaries and current annual average loads can only be considered 1st order estimates (See section 4 of this report). We presently have only limited understanding of sediment loads entering the Bay from local tributaries although recent evidence suggests that this source may now be larger than Central Valley loads (Lewicki and McKee, 2009). Data on HABs for SF Bay are presently weak due mainly to limited recent occurrences and few research programs aimed at such ephemeral events. There is a lack of data on the causes, timing, extent of toxin production (influences and concentrations), nutrient consumption during blooms, and processes leading to bloom termination and predictive models that link HAB production to nutrient loads are lacking. There is presently no systematic monitoring program for either zooplankton benthic macro-invertebrates or eelgrass and other submerged aquatic vegetation. Given the rate of change of these populations in recent decades (see later sections); we suggest that comprehensive surveys of these system components should be completed about every five years. In addition, Cloern et al. (2006) point out that our understanding of key processes of change is limited by the lack of systematic measurement of phytoplankton growth and transport rates, and there is no systematic measurement of phytoplankton biomass outside the Golden Gate. There is presently no systematic collection of urea concentrations and there has been no research done to-date on the role of ammonium or urea limitation on phytoplankton growth rates in the Central and South Bays.

To-date no system-wide dynamic simulation model has been developed, but given the spatial and temporal data richness of the system and the basis provided by existing sub-system models, of any system, SF Bay hold great promise for the development of such a unified model. Wilkerson et al., 2006 suggest in their closing statement that: “The effect of water management changes, i.e., changing DIN loading, can now be modeled using these nitrogen productivity data as a framework to understand the importance of different nutrient concentrations in the development of phytoplankton blooms in the northern SFB.” In addition, given that the majority, if not all the parameters that supported the Chesapeake Bay, Bay Health Index (BHI), have been collected in SF Bay, we can’t help but conclude that further effort to develop such an index for SF Bay may yield a useful tool for tracking and predicting nutrient related water quality changes in SF Bay.

Sound and Practical Measurement

Phytoplankton is among the best studied assemblages of estuarine organisms, with over six decades of research. As a consequence, there are a variety of sound and scientifically well-vetted means of measuring phytoplankton biomass, productivity, community composition, and growth efficiency (Table 5.2.4). See Sutula (2011) review of phytoplankton indicators for the estuarine NNE for additional detail. In addition, San Francisco Bay has the advantage of an established long term USGS research program on phytoplankton that began in 1977 and currently spans 29 years and that includes >11,000 discrete laboratory measurements of the chlorophyll a in water samples and 156,610 estimates of chlorophyll a made from a linear relationship between fluorometer voltage and discrete lab measurements. Thus, a long-term data set exists to support decisions on regulatory endpoints as well as for use in developing a load-response model. It is important to note that the USGS research program is not mandated. There is the critical need for a commitment to support regular sampling to measure and understand future changes in water quality, including those related to nutrient enrichment.

Table 5.2.4. Summary of methods for measuring phytoplankton biomass and community structure (from Sutula, 2011).

Group	Indicator	Methods	Information
Water Clarity	Light Attenuation	Grab samples or Continuously deployed Instrumentation (e.g., data sondes)	Attenuation of light reaching bottom
	Turbidity or TSS		Turbidity or TSS
	Secchi Depth	Field survey	Coarse measure of water clarity
Phytoplankton Biomass	Discrete water column chlorophyll a	Grab samples with laboratory analysis	Precise measure of water column chlorophyll-a
	Chlorophyll a fluorescence	*In situ* probes and flow through instrumentation	Chlorophyll a fluorescence, which must be calibrated to grab samples
	Remote sensing of color	Satellite (SeaWiFS, MERIS, MODIS) or wide variety of multispectral and hyperspectral airborne sensors	Water color as a proxy for chlorophyll a
Phytoplankton Productivity	Photosynthesis versus irradiance curves	Modeled production	Rate of carbon fixation per unit time per square meter (areal) or cubic meter (volumetric)
	Isotope	Direct measure of gross productivity, respiration, and net productivity	
Phytoplankton Community Structure	Number of species and relative abundance	Taxonomy and cell counts	Dominant species and presence/absence of rare or pollutant tolerant taxa
	Chemotaxonomic phaeopigments	HPLC	Relative composition of broad taxonomic group composition by determining chlorophyll and carotenoid presence phaeopigments (e.g., Chlorophytes, Cryptomonads, diatoms, dinoflagellates and zeaxanthin)
HAB species and toxin concentrations	HAB species abundance	Taxonomy/cell counts or Q-PCR	Abundance of HAB species
	Toxin concentrations	HPLC or Elisa Assay	Concentration of toxins associated with water column or sediment

Acceptable Signal to Noise Ratio

Phytoplankton are subject to a high degree of spatial and temporal variability (see previous sections summarizing this variability), due to a number of physical, chemical and biological co-factors. However, given the long-term data set available for SF Bay, it has been possible to determine statistically-significant trends with respect to phytoplankton biomass at the decadal time scale (e.g., Cloern et al., 2006). Thus our ability to use phytoplankton as an NNE indicator in SF Bay is possible because of this 39-yr data set. However, this may be too long of a time scale to be useful for determining the cause of more ephemeral system responses such as HABs. Smetacek and Cloern (2008) comment that because phytoplankton species populations appear and disappear within weeks, assessing change on shorter timescales may require higher resolution monitoring of annual cycles over many years. It should be noted that while high spatial and temporal variability is characteristic of all biological indicators, these indicators tend to integrate better over time and space than stressors, such as nutrient concentrations. Ultimately, our understanding and the various hypotheses about controls on spatial and temporal variability in phytoplankton biomass, productivity and community structure and linkages to consumers can be tested and refined through predictive models.

The use of any particular indicator of phytoplankton (biomass, productivity, assemblage) alone to assess eutrophication is not recommended. Each of these indicators has strengths and limitations which, when measured and used as multiple lines of evidence provide a more holistic assessment of adverse effects with an acceptable signal:noise ratio. By contrast, use of any single indicator may produce a false negative or positive assessment of adverse effects without supporting information.

For example, phytoplankton productivity is the measure of the rate of biomass production and is in fact a more immediate measure of the influence of nutrients on autotrophic production and potential eutrophication than biomass alone. Cole and Cloern (1984) showed that regions of the Bay which may have higher chlorophyll a do not necessarily provide a net transfer of carbon to herbivores because, in some cases, respiration exceeds gross production in deeper or more turbid areas. Therefore, high chlorophyll a alone is not necessarily indicative of trophic status if that high chlorophyll a is the depth of the photic zone is limited to a fraction of total water depth. In addition, productivity and assemblage information (e.g. % diatoms) have a much stronger linkage to beneficial use than biomass per se.

However, though the rate of productivity may be a good indicator of nutrient concentration, the ultimate disposition of the production may vary across estuaries or even within an estuary based on several factors. High productivity in deep and well-mixed waters may not result in problematic levels of phytoplankton biomass as the biomass produced can be mixed throughout the water column, and the balance of productivity to respiration (P:R) within the entire water column constrains the production within acceptable limits. Moreover, even in shallow estuaries where biomass may accumulate in the euphotic zone, if grazer or filter feeding communities are present, the biomass may be efficiently removed, contributing to a healthy and productive estuary, without causing negative impacts. Second, direct measures of productivity are relatively difficult and time consuming, so gathering data over a large and representative spatial area is not typically widely conducted in monitoring programs for coastal waters (Anderson et al. 2006).

Phytoplankton indicators can be used in tandem to provide information not only about the accumulation of organic matter in the system, but also information about the health of the phytoplankton community and factors that may lead to trophic level changes that underpin key estuarine beneficial uses. For example, the ratio of productivity: biomass is an index of growth potential and is a meaningful indicator of the physiological state of phytoplankton from ammonium or other toxic contaminant (Yoshiyama and Sharp, 2006).Thus a combination of measures of phytoplankton biomass, productivity, and assemblage are needed in order to make a more robust assessment of adverse effects of nutrient over-enrichment or eutrophication.

Approaches to Setting Numeric Endpoints Based on Phytoplankton

Paradigms for establishment of estuarine numeric endpoints based on phytoplankton typically separate seagrass from subtidal unvegetated habitats. For seagrass, precedents for establishment of numeric endpoints exist based on biomass, based on light limitation for photosynthesis of seagrass beds (e.g., Janicki et al., 2000; Kemp et al., 2004; Brown et al., 2004; Sutula, 2011). Turbidity, total suspended solids (TSS), chlorophyll a, and dissolved organic matter are measured to determine light available in the water column that reaches the seagrass bed (Biber et al., 2008). For example, In the mid-Atlantic, environmental conditions that allow adequate light penetration for SAV survival are total suspended solids (TSS) less than 15 mg L^{-1} and chlorophyll a less than 15μg L^{-1} (Kemp et al., 2004). Bio-optical models predicting light attenuation under various environmental conditions have been calibrated for the Chesapeake Bay (Gallegos, 2001), Indian River Lagoon in Florida (Gallegos and Kenworthy, 1996), and North River in North Carolina (Biber et al., 2008), Yakina Estuary in Oregon (Brown et al., 2007), and Tampa Bay in Florida (Janicki et al., 2000). Explicit studies are needed to understand the precise light requirements of seagrass in SF Bay. This information can be used to develop a bio-optical model that could be used to establish a combination of chlorophyll a thresholds and turbidity to establish levels of light attenuation that will be protective of SF Bay seagrass beds (see Section 5.4 for further discussion).

For unvegetated subtidal habitats, some precedent for setting chlorophyll a endpoints for biomass (e.g., Bricker et al., 2003; Soucho et al., 2000; Ferreira et al., 2006; Zalidvar et al., 2008) and phytoplankton productivity (Devlin et al., 2007) to assess eutrophication exists, though use of phytoplankton for regulatory purposes is not widespread. Ultimately, confidence in setting NNE endpoints based on biomass, productivity and/or community structure is more easily accomplished with long-term data sets that describe the range in variability in these indicators and relationship to consumer communities linked to beneficial uses. In SF Bay, this would be done by convening a workshop of experts to synthesize data that could be used to establish thresholds based on biomass, productivity and community structure.

With respect to HAB species abundance and toxin concentrations, experience with establishing numeric thresholds is more evident for freshwater cyanobacteria species such as *Microsystis spp*. A summary underway of suggested action levels for adverse health effects of anatoxin-a, cylindrospermospin, and four microcystins (LA, LR, RR, and YR) by California EPA Office of Environmental Health Hazard Assessment will be an excellent starting point for consideration of numeric endpoints for cyanobacteria. That report is currently in peer review. These thresholds are most applicable for oligohaline environments, where cyanobacteria are most prevalent, but should also be considered for downstream impacts to polyhaline or euhaline habitats, as cyanobacteria toxins such as microcystin can accumulate in marine invertebrates and thus adversely affect marine mammals. As an example of this, Miller et al. (2010) found that microcystin poisoning was the likely cause of death in sea otters in Monterey Bay Marine Sanctuary.

For estuarine or marine HAB species typically found in California, there is a lack of understanding on the controls of relative abundance and toxins production that limit our ability to use these as NNE indicators at this time. Additional research is needed to understand controls on marine HAB frequency and occurrence and controls on toxin production. Additional work is required to understand chemical controls on community structure (ammonium, trace elements, and micronutrients).

5.2.6 Summary: Use of Phytoplankton as an NNE Indicator

Overall, phytoplankton appears to satisfy the four evaluation criteria to be considered as an NNE indicator for SF Bay. However, several key data gaps and recommended next steps are required in order to further pursue its use for this purpose. The steps are:

1. Select the precise indicator and numeric endpoints

   The following indicators are recommended for use in the SF Bay NNE: 1) phytoplankton biomass, 2) productivity, 3) phytoplankton assemblage and in particular, HAB species abundance and toxin concentrations. The SF Bay TAT recommends a series of expert workshops to synthesize data, identify data gaps and create a phytoplankton assessment framework that would be used by policy makers to set numeric endpoints based on these indicators.

2. Scope the development of a series of dynamic watershed, atmospheric, and oceanic loading and SF Bay hydrodynamic and water quality models to simulate the ecological response of the Bay to nutrient loads and other factors. This would be done through a series of workshops to develop a modeling strategy for SF Bay. The product of this effort would be the identification of the appropriate models, a phased workplan, timeline and budget to develop these models, and identification of and coordination among key institutions, programs and respectively roles. This scoping must include three elements:
   1) *Conceptual Model Development*. There is a need to develop conceptual models that explicitly show linkage between watershed, airshed, ocean and estuarine hydrology, nutrient loads, ecological response indicators, and “co-factors” that control ecological response to eutrophication or oligotrophication. The conceptual model would identify key sources, sinks and

processes of transformation that would need to be incorporated into the models. Areas of disagreement on causal mechanisms should be synthesized as alternative hypotheses than can be tested through experiments, field studies and model sensitivity analyses.

2) *Model Selection*. The next step in the scoping of model development is to select the appropriate models. This should be done by reviewing available loading and receiving waterbody models and present an analysis of the advantages and disadvantages of their use for modeling eutrophication and other adverse responses to nutrients, based on the explicit conceptual models.
3) *Data Needs Assessment*. Based on explicit conceptual models and the modeling platform selected, the next step would be to identify data required to support model development, calibration and validation.

This information could be synthesized into a workplan to develop the loading and estuary water quality models and a preliminary timeline and budget for Phase I of the effort.

5.3 Macroalgae

Macroalgae are an ancient group of single to multicellular primary producers found in all aquatic ecosystems. They provide the same ecological functions as vascular plants in terrestrial ecosystems, but lack the structural tissues characteristic of plants. Marine macroalgae form an important component of productive and highly diverse ecosystems in estuaries worldwide and in moderate abundances provide vital ecosystem services. They are important primary producers in intertidal and shallow subtidal estuaries, providing food and refuge for invertebrates, juvenile fish, crabs and other species. However, some species of macroalgae thrive in nutrient-enriched waters, out-competing other primary producers (Sutula, 2011). For this reason, macroalgae have been proven to be useful indicators of eutrophication in estuaries. Estuarine ecosystems have been subjected to increased frequencies and magnitudes of harmful macroalgal blooms, outcompeting seagrasses and other primary producers and resulting in hypoxia, reduced biodiversity, fish and invertebrate mortality, altered food webs and energy flow, and disruption of biogeochemical cycling (Sfriso et al., 1987; Valiela et al., 1992, 1997; Coon, 1998; Young et al., 1998; Raffaelli et al., 1989; Bolam et al., 2000). Fong, Green and Kennison provide a detailed review of the utility of macroalgae as an NNE indicator in estuaries (Chapter 5, Sutula, 2011). This section provides a brief synopsis of that work and presents literature directly relevant to known abundance, distribution and variability of macroalgae in SF Bay estuary.

5.3.1 Applicable Habitat Types

As an NNE indicator, macroalgae are most applicable to intertidal or shallow subtidal habitat, including seagrass beds. Macroalgae are also applicable to diked Baylands and salt ponds.

5.3.2 Available Data on Macroalgae in San Francisco Bay

No regular program of monitoring of macroalgal abundance exists in SF Bay. A survey of macroalgal abundance in seagrass beds is slated for completion in Spring 2011.

5.3.3 Macroalgal Relationship to Nutrients and Water Quality

Macroalgae are important members of the primary producer community in rocky and shallow soft-sediment systems worldwide where light penetrates to large areas of the benthos. They are present in all estuarine geoforms, but their relative abundance is, at least in part, proportional to the amount of suitable habitat and nutrient supply. In oligotrophic systems, macroalgae are a component of the primary producer community, but are generally not dominant (Figure 5.3.1). Rather, in shallow subtidal and intertidal portions of these estuaries, benthic communities may be dominated by the microphytobenthos (MBP), an assemblage of diatoms, dinoflagellates, cyanobacteria, and sporling green macroalgae living on the sediment surface that can contribute up to 50% of the primary production in an estuary (Underwood and Kromkamp 1999). In larger, well-flushed California estuaries, shallow subtidal portions are often dominated by seagrasses.

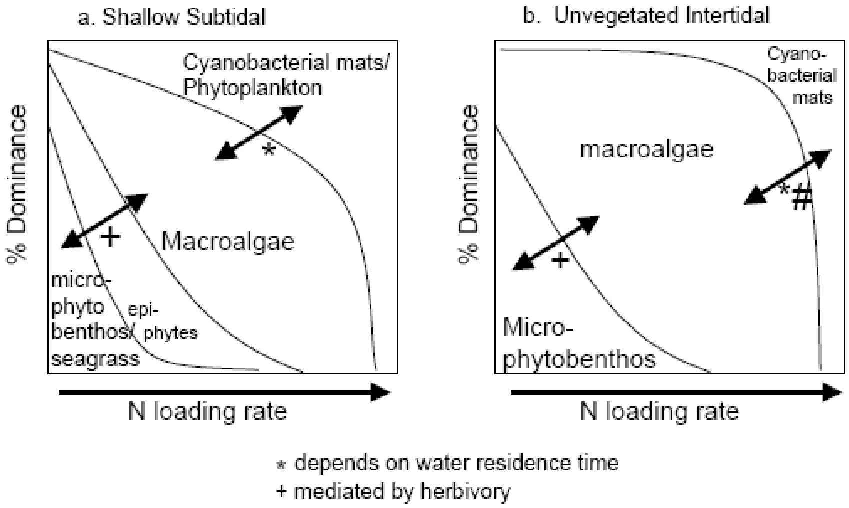


Figure 5.3.1. Conceptual model of the relationships between N loading rate and the community composition of primary producers in a) shallow subtidal and b) unvegetated intertidal habitat in California estuaries.

As nutrient availability increases, it has been well-documented in many parts of the world that blooms of green or red macroalgae become dominant in shallow subtidal and intertidal estuaries and lagoons, replacing seagrass or MPB (Figure 3.2; e.g., Sfriso et al., 1987, 1992; Raffaelli et al., 1989,; Valiela et al., 1992, 1997; Peckol et al., 1994; Marcomini et al., 1995; Hernández et al., 1997; Hauxwell et al., 1998; Kamer et al., 2001). Macroalgal carbon moves more easily through microbial and consumer food webs than carbon derived from seagrasses due to the more labile nature of its carbon (Valiela et al., 1997). In shallow estuaries, macroalgae can dominantly influence the O_2 profile of the water column, further affecting the biogeochemistry of the sediments and exchange of nutrient to and from sediments. Macroalgal blooms affect the abundance of macrofauna through more frequent hypoxia/anoxia events (caused by metabolism of organic matter that depletes the benthos of dissolved oxygen) and habitat changes (Valiela et al., 1997; Cloern, 2001). Thus blooms of opportunistic macroalgae can cause in loss of critical habitat, hypoxia, reduced biodiversity, fish and invertebrate mortality, altered food webs and energy flow, and disruption of biogeochemical cycling (Sfriso et al., 1987; Valiela et al., 1992, 1997; Coon, 1998; Young et al., 1998; Raffaelli et al., 1989; Bolam et al., 2000).

Macroalgae have a well-documented linkage to nutrients and water quality. Common bloom-forming species of Ulva have been used as biological indicators of nutrient supplies in estuaries. They are good indicators because of their ability to rapidly take up large pulses of inorganic nitrogen (Fujita, 1985; Pedersen and Borum, 1997; Lotze and Schramm, 2000; Runcie et al., 2003) and store it for future growth (Fujita, 1985; Bjornsater and Wheeler, 1990; Fong et al., 1994; Pedersen and Borum, 1997; Lotze and Schramm, 2000; Naldi and Viaroli, 2002). Thus, tissue nutrients in macroalgae integrate nutrient supplies over time (Wilson, 1994). This is especially important in Mediterranean systems, where nutrient supply and availability can be variable due to pulses of nutrients that are delivered by runoff from seasonal storms in the wet season as well as during periodic discharges of sewage and agricultural waste in both the wet and dry seasons (Zedler, 1996). In addition, in eutrophic estuaries with organically –enriched sediments, macroalgal biomass accumulation can be partially or wholly supported by benthic regeneration of nutrients; macroalgae have in fact been shown to increases the magnitude of benthic fluxes by increasing the concentration gradient between surface waters and sediment pore waters. Therefore, traditional water column nutrient sampling methods may miss pulsed nutrient signals, and not provide an accurate estimate of nutrient enrichment. With the combination of a high affinity for nitrogen and ability to store nutrients, macroalgal tissue nutrient status can be used as a biological indicator (Harrocks et al., 1995; Fong et al., 1998; Costanzo et al., 2000; Huntington and Boyer, 2008b) to determine nutrient availability.

The effects of nutrient loading rate on macroalgal distribution and biomass accumulation are heavily influenced by the hydrological connection to the ocean of each estuary. Due to the ability of most bloom species to shift habitat usage from benthic to floating stages, macroalgae are able to occupy all estuarine habitats by rafting in surface waters or depositing on subtidal or intertidal sediments. Biomass accumulation, however, is linked to nutrient supply. Thus, low abundances of macroalgae may co-occur in low nutrient systems with subtidal and intertidal seagrasses and the microphytobenthos (benthic microalgal community). It is only as nutrient loads increase that proliferation of macroalgae has negative impacts on other producer groups across all estuarine classes (see Sutula, 2011 for a full review).

5.3.4 Species Composition of Macroalgae in San Francisco Bay

In the SF Bay 162 species of macroalgae have been identified, the most common species are *Ulva clathrata, U. intestinalis, U. linza, U. angusta, U. lactuca* (commonly known as sea lettuce), *Cladophora sericea, Antithamnion kylinii* and *Polysiphonia denudate* (Figure 5.3.2). Besides the last two which are red and brown algae respectively, all of those species are green algae. Other common macroalgal species with smaller spatial distributions include Fucus distichus spp. edentatus, *Gracilaria verrucosa, Bryopsis hynoides, Grateloupia doryphora, Gigartina exasperata, Cryptopleura violacea,* and *Gelidium couheri* (Josselyn and West, 1985). The kelp (*Laminaria*) has been identified in Raccoon Straight between Tiburon and Angel Island (Josselyn and West, 1985; BCDC, 2010).

5.3.5 Trends and Factors Effecting Temporal and Spatial Variation of Macroalgae

There is very little previous literature regarding the historical extent of macroalgae in the SF Bay. Josselyn and West (1985) describe the spatial extent of macroalgae, but macroalgal distribution is not described quantitatively. According to Josselyn and West (1985), the SF Bay has experienced some long term changes in macroalgal species. Several species have been accidentally introduced since the 1970s including *Codium fragile*, *Ascophyllum nodosum ecad scorpioides, Sargassum muticum, Polysiphonia denudata,* and *Callithamnion byssoides*.

As with most estuarine organisms, there are number of complex and interrelated factors that influence the spatial and temporal variation of macroalgae. Spatial and temporal variations in estuarine nutrients and relationships to macroalgae have been studied extensively at various locations, mostly because of their relative importance to primary production (Valiela et al., 1992; Peckol and Rivers, 1995; Pihl et al., 1999; Boyle et al., 2004; Krause-Jensen et al., 2007). The supply of nutrients to an estuary is a primary control on macroalgal abundance (Josselyn and West, 1985; Mackas and Harrison, 1997; Boyle et al., 2004). According to Cloern (1985), continual periods of low discharge allow some marine-estuarine macroalgal species to migrate upstream, and that maximum biomass and diversity of macroalgae is reached in the summer. Variation in salinity, temperature, and available light (Josselyn and West, 1985), as well as the abundance of grazers and differences in water residence time (Valiela et al., 1997), are factors that lead to spatial gradients between different areas of the SF Bay and temporal variation on a scale from days to years.

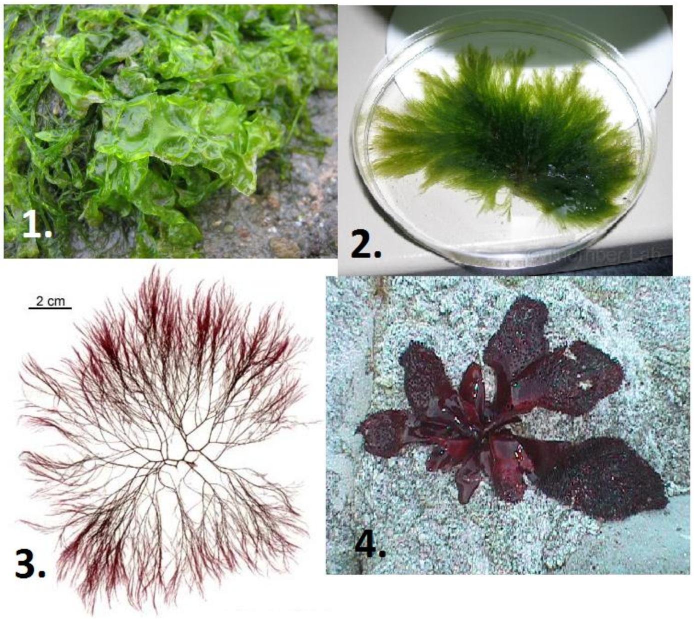


Figure 5.3.2. Examples of macroalgae found in the San Francisco Bay: 1. *Ulva lactuca* (commonly known as sea lettuce), 2. *Cladophora sericea*, 3. *Polysiphonia denudata,* and 4. *Gigartina exasperata.*

Spatial Variability

Macroalgae are most commonly found growing on hard substrate such as rock outcroppings, coarse sediment, and man-made structures but they are also found in lesser diversity on mud and salt flats. Hard substrate suitable for macroalgal growth in the intertidal zone is more common on the shores closest to the ocean and less common in San Pablo Bay, Suisun Bay, and the southern branch of SF Bay (Josselyn and West, 1985). Calm, protected areas often provide ideal locations for certain macroalgal species to grow and accumulate (Josselyn and West, 1985) however at present, macroalgae beds are less dominant by area in SF Bay than submerged aquatic vegetation (BCDC, 2010) Drifting macroalgae also can accumulate in thick mats, detached from substrate, and the current and tides can carry them away from the intertidal zone to deeper areas of the estuary (Nichols and Patamat, 1988). However, at present, there is no recent data on the distribution of macroalgae in the SF Bay.

Temporal Variability

Seasonal variability of macroalgae in SF Bay appears to be strongly influenced by temperature, salinity and light availability (Josselyn and West, 1985). Water temperature varies over the year with the highest temperatures in the summer and the lowest in the winter. There is little difference in water temperature between different areas of the Bay. Salinity drops significantly in the winter to below 10ppt and there is a reduction in macroalgal species number (Cloern and Nichols, 1985; Josselyn and West, 1985). Light attenuation, measured as the light-extinction coefficient (m^{-1}), fluctuates seasonally with the highest levels in the late-winter and spring and the lowest levels in the summer and early-fall. There are significant differences in levels of light attenuation between seaward sites (lower levels) and landward sites (higher levels) in the SF Bay. Figure 5.3.3 compares these previously listed physical factors over an annual cycle. The greatest abundance of macroalgae, measured as percent cover, occurs during May-September. Green algae contribute the most percent cover, red algae are present all year, and brown algae are present all year, but are only abundant during the summer.

Relationships between more frequent daytime exposure of mudflats and an increase in macroalgae, particularly *Ulva clathrata*, have been observed in the SF Bay (Shellem and Josselyn and West, 1982). This greater daytime exposure occurs from late spring to early summer due to the increased frequency of daytime low tides (Josselyn and West, 1985). An increase in macroalgal abundance during the summer has been found to coincide with peak periods of benthic efflux of ammonium and phosphate (McLaughlin et al., 2011). Previous studies have suggested that macroalgae can drive an increased efflux of dissolved inorganic nutrients from sediments by drawing down surface water concentration, thereby increasing the concentration gradient (Tyler et al., 2003). As these nutrients are trapped as biomass, macroalgae become an effective mechanism to retain and recycle nutrients within an estuary, diverting losses such as denitrification or tidal outflow.

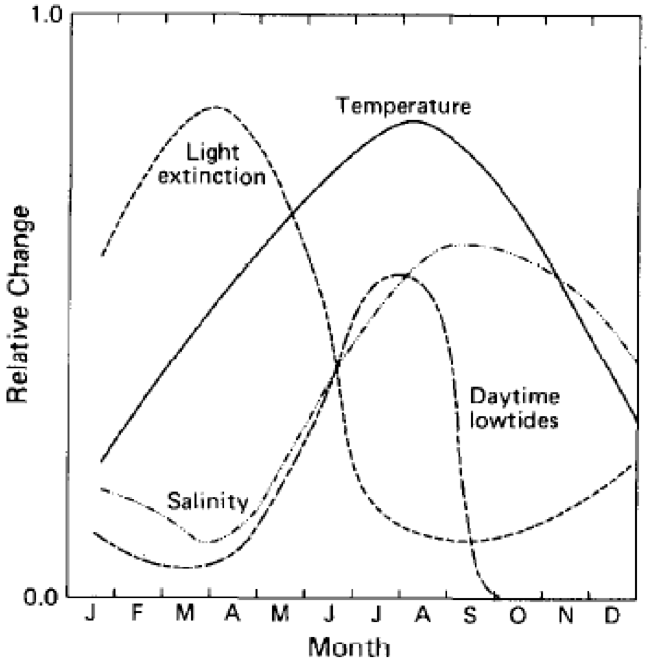


Figure 5.3.3. Relative relationships between physical factors that affect macroalgal growth in San Pablo Bay over the course of a year (adopted from Josselyn and West, 1985).

5.3.6 Utility of Macroalgae as an Eutrophication Indicator for San Francisco Bay

Clear Linkage to Beneficial Uses

Excessive macroalgal blooms have a variety of negative effects on estuarine beneficial uses including: 1) increasing frequency of water column and sediment hypoxia and heightening heterotrophic bacterial activity, resulting in poor water quality and increased frequency of diseases, 2) alteration of biogeochemical cycling, more rapid nutrient regeneration (Tubbs and Tubbs, 1980; Raffaelli et al., 1991; Wennhage and Pihl, 1994; Bolam et al., 2000), 3) shading or smothering of seagrass, shellfish beds and other important habitats (Nelson, 2009; Young 2009), 4) decreased recruitment and survival of benthic invertebrates and reduced carrying capacities for fishes and shorebirds (e.g.Raffaelli, 1999; Thomsen and McGlathery, 2006; Nezlin et al., 2009), 5) poor aesthetics and an increase in odors relating to the decomposition of organic matter and increased sulfide production, and 6) subsequent changes in both trophic and community structure of invertebrates, birds and fishes (Raffaelli et al., 1989, 1991; Bolam et al., 2000). Cumulatively, these adverse effects result in a reduction in recreational use of estuarine waters (REC1 and REC2), poor water column and benthic habitat quality for estuarine (EST) and marine (MAR) aquatic species, direct impacts to populations of threatened and endangered (RARE), migratory (MIGR) and spawning (SPAWN) birds, fish and mammals, and reduction in the economic value of commercial and sports fisheries, aquaculture, and shellfish harvesting (COMM, AQUA).

Fish and invertebrate kills as a result of lowered dissolved oxygen may occur when estuaries are stressed by mats of macroalgae, especially in conjunction with high temperatures and high cloud cover (D'Avanzo and Kremer, 1994). In addition to oxygen stress by respiring macroalgae, fish may be physically affected by drift mats. For example, cod foraging efficiency was drastically reduced with *Ulva intestinalis* cover of 70-80% (Pihl et al., 1995). Macroalgae may also affect piscine recruitment. Wennhage & Pihl (1994) found that juvenile flat fish preferred to recruit to bare sand compared to plots with dense macroalgal cover. They speculated that macroalgae invokes stress in juvenile fish through declines in dissolved oxygen and also restricts prey availability. However, drifting macroalgae is not the only form of algae that may result in the loss of fish and invertebrates. Increases in epiphytic algae on seagrass can result in dramatic reductions in the abundance and biomass of epibenthic invertebrates and fish (Isaksson and Pihl, 1992). Thus ephemeral macroalgae may cause deleterious declines in both recreational and commercial fish stocks (Raffaelli, 1999).

Field studies show that inverse correlations exist between the density of macroalgae and numbers of shorebirds. Cabral et al. (1999) made the claim, "In a long-term perspective, an increase of dense and contiguous macroalgal mats, covering large areas of the intertidal flats, may affect directly or indirectly all wader species in the Mondego estuary." Green (2010) showed that the presence of macroalgae changed foraging behavior from pecking to probing in Sandpipers and Marbled Godwits, suggesting that macroalgae hindered foraging by obscuring visual cues or physically interfering with foraging effort. Further, if macroalgal blooms reduce prey intake rates by shorebirds, then shorebirds with less flexible diets may be more negatively affected than generalist foragers that feed on a wide range of prey species. Green (2010) found avoidance of mats based on foraging ecology. For example, shorebirds that are generalist foragers, such as Least terns and Western Sandpipers and Willets, foraged on macroalgal mats and bare sediment equally. In contrast, shorebirds with more specific dietary requirements such as Marbled Godwits often avoided mats while foraging. In another study, Cabral et al. (1999) observed that Dunlin, a bird species with more restricted diets, tended to avoid dense mats. These studies suggest that as macroalgal blooms become more prevalent in estuaries, specialist species of birds may suffer losses in population numbers.

Predictive Relationships to Causal Factors

There is overwhelming evidence that blooms of macroalgae are stimulated by high nutrient loading, particularly of nitrogen (N) and phosphorus (P) (e.g., Raffaelli et al., 1989; Valiela et al., 1992; Peckol and River 1995; Pihl et al 1999; Krause-Jensen et al., 2007). Recent studies have shown that estimates of nutrient loading that include all possible sources as well as physical removal (flushing) and biological processes are accurate and generalizable predictors of macroalgal biomass. In one of the best examples of this approach, Fox et al. (2008) compared three sub-estuaries of Waquoit Bay, Massachusetts, with different nitrogen loads and found the magnitude of macroalgal standing stock was predicted by total nitrogen load over a six-year period. Notably, this level of detail of the relationship between nutrient loading and producer biomass has been quantified in only a few systems (Valiela et al., 1992, 1997; Hauxwell et al., 1998; Conley et al., 2000; Fox et al., 2008), as it is an expensive and time-consuming process. Measures of nutrient removal to the ocean via tidal flushing are also a key factor that may help to ensure accurate predictions. For example, in Mugu Lagoon (Calleguas Creek arm), southern California, water column nutrient concentrations were always high but algal biomass always low due to low hydraulic residence time and scouring of sediments (Kennison, 2008). Finally, biological processing such as nutrient uptake and algal growth, internal nutrient cycling, and grazing (see Schramm, 1999 for review) must be taken into account to improve the predictive capability of any model. For example, longer residence times that allow more biological uptake and resultant macroalgal growth may result in lower water column nutrient concentrations and more proliferation of macroalgae as found in Mugu (West arm), Tijuana Estuary, and Upper Newport Bay in southern California (Kennison, 2008). Additional work is needed to understand conditions in which phase shifts from microphytobenthos to macroalgae occur, including quantifying rates of uptake and release of nutrients from macroalgae and seasonal storage and release of inorganic nutrients in sediments, in order to parameterize dynamic simulation models of estuarine water quality and biological response to nutrient loads.

Sound and Practical Measurement

A suite of methods to assess the extent and impact of macroalgae in estuaries is well vetted and considered to be sound and practical. These methods are centered on measures of macroalgal taxonomic composition and abundance (biomass and cover). Of these, taxonomic composition is not a particularly useful indicator of eutrophication because the taxonomic composition of macroalgae in California estuaries is limited and the presence of these species alone does not indicate eutrophication.

Overall, macroalgal abundance, as measured by biomass and percent cover is a widely used and accepted method of measurement. Measures of biomass typically require field sampling, collection of samples on mudflats or shallow subtidal habitat in randomly placed transects, and processing of biomass in the laboratory and thus are labor-intensive. Percent cover is usually collected along the same transect as biomass and provide an ability to better capture spatial heterogeneity of macroalgal mats. Measures of both biomass and percent cover are essential to characterize macroalgal response to nutrient loadings. For example, 100% cover of a visible yet thin and still attached mat of algae that may weigh only grams per square meter while 100% cover of dense macroalgal bloom may weigh 12 kg per square meter. Thus the combination of both percent cover and biomass are used to integrate the magnitude and spatial heterogeneity of a macroalgal bloom.

Acceptable Signal to Noise Ratio

Macroalgal blooms are known to be spatial patchy and temporally variable. Mechanisms that control net production of macroalgae are the same as for other primary producers: geographic limits for growth are set by temperature and light and for removal by grazing and physical disturbance. Within these geographical limits, biomass accumulation is controlled by many interacting biotic and abiotic factors including light quantity and quality, water motion, intra- and inter-specific competition, herbivory, and physical disturbance. However, in most places where macroalgae proliferate and dominate estuarine communities in temperate zones, this dominance is a function of nutrient, usually nitrogen (N), availability (for reviews see Howarth and Marino, 2006; Valiela et al., 1997, Vitousek et al., 1997; Downing et al., 1999). For this reason, macroalgae have been successfully used to detect a trend towards decreasing or increasing eutrophication (Scanlan et al., 2007).

Approaches to Setting Numeric Endpoints Based on Macroalgae

Paradigms for establishment of estuarine numeric endpoints based on macroalgal biomass and cover must separate effects for three types of habitats: 1) seagrass 2) polyhaline and euhaline intertidal and shallow subtidal unvegetated habitats, and 3) oligohaline and mesohaline intertidal and subtidal habitat (vegetated and unvegetated; see Figure 5.3.4). A wealth of literature exists documenting the adverse effects of macroalgae on benthic infauna in intertidal and shallow subtidal habitat (Sutula, 2011). Two studies have been conducted that provide data useful for "range-finding" of endpoints. However, more definitive studies need to be conducted to develop better characterization of the relationship between macroalgal biomass, duration of bloom, and effects benthic infauna in intertidal flats across the diversity of intertidal habitats encountered in California (See Sutula, 2011). Some preliminary studies are now underway under the Estuarine NNE project.

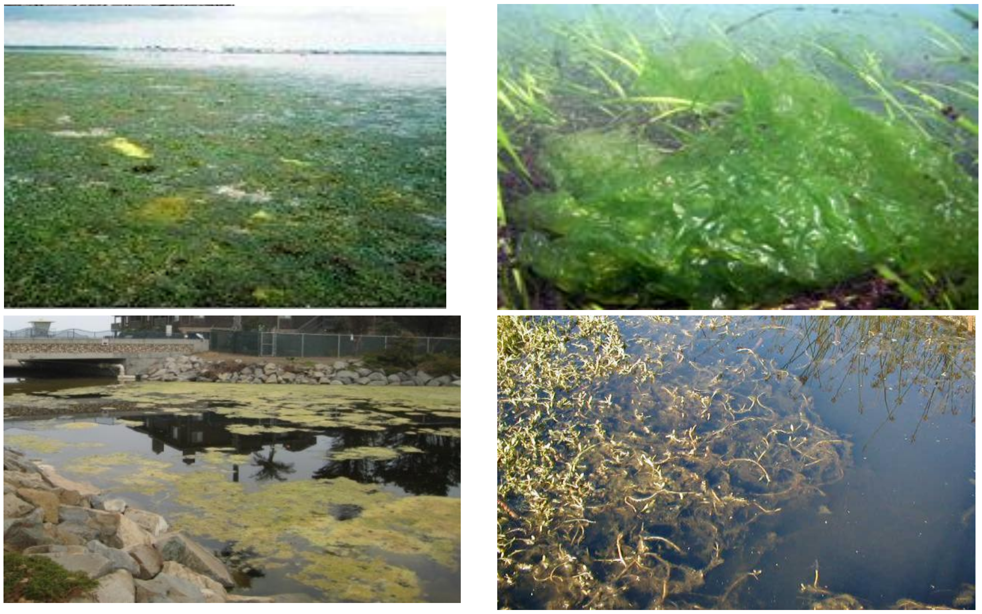
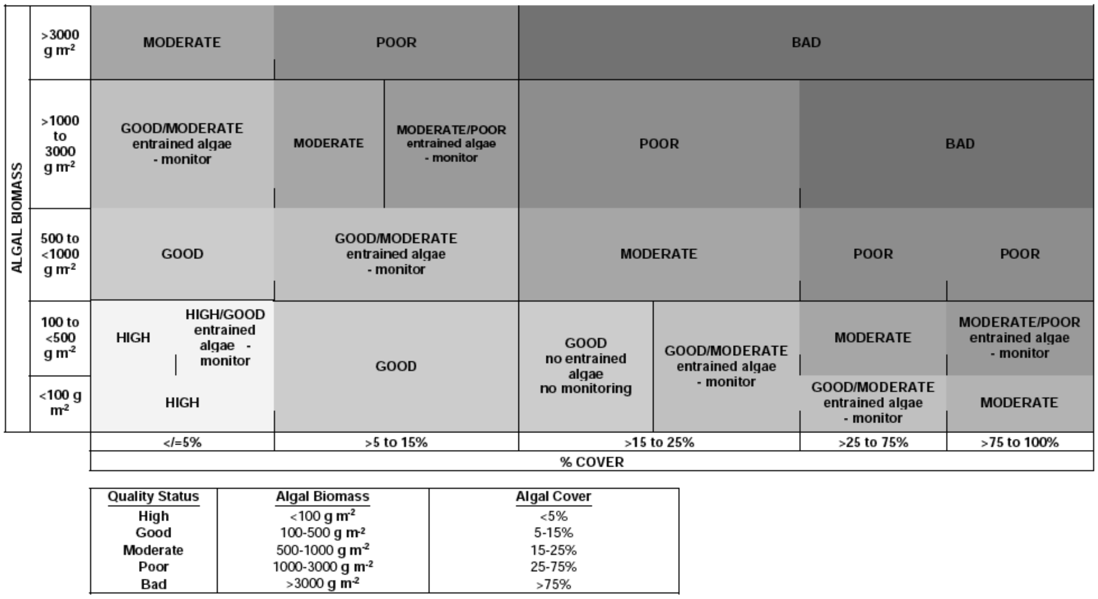


Figure 5.3.4. Examples of types of habitats in which macroalgae can occupy a dominant role among other primary producers: mats on intertidal shallow subtidal flats in polyhaline to euhaline environments (upper left), rafting mats on seagrass (upper right), and floating/rafting mats in oligohaline unvegetated (lower left) and intercalated with *Ruppia* in oligohaline environments (lower right).

Studies documenting effects of macroalgae on seagrass bed density have likewise been published (see Sutula et al., 2011 for full review), but few of these studies are useful specifically for establishing endpoints. As with macroalgae on intertidal flats, studies are likewise needed to document the effects of rafting mat biomass and duration on seagrass bed density in California estuaries. Recently, there is anecdotal evidence of macroalgal blooms found in seagrass habitat in SF Bay (Katharyn. Boyer , SFSU, personal communication, March 2011), though a comprehensive survey is needed to better document the cover and biomass found in SF Bay seagrass beds. Little documentation is available on the effects of rafting algae on microphytobenthos and brackish SAV oligohaline environments. Conceptual models from applicable habitats could be applied, but few field studies have been conducted to illustrate the effects of rafting mats on dissolved oxygen, other pelagic or benthic food webs. These conceptual models may be applicable to the Northern reaches of SF Bay.

Some precedent exists for the use of macroalgae to assess eutrophication. With the adoption of the Water Framework Directive (WDF, 2000), the European Union has been working to assess the ecological condition of its waterbodies. Scanlan et al. (2007) has proposed an assessment framework to diagnose eutrophication. This framework is moving towards adoption within the WFD (Zalidvar et al., 2008). The Scanlan et al. (2007) assessment framework utilizes both macroalgal cover and biomass in a multiple lines of evidence approach (Figure 5.3.5). Both biomass and cover are required to make a diagnosis, because the measurement of just one indicator in isolation could be misleading. For example, an estuary may have low biomass (a positive indicator for estuarine health) but high macroalgal cover (a negative indicator for estuarine health) resulting in a moderate impact to the ecosystem. On the other hand, high macroalgal biomass may be recorded locally, but be mediated by low percent cover over the whole estuary.



ALGAL BIOMASS	</=5%	>5 to 15%	>15 to 25%	>25 to 75%	>75 to 100%
>3000 g m²	MODERATE	POOR	BAD	BAD	BAD
>1000 to 3000 g m²	GOOD/MODERATE entrained algae - monitor	MODERATE MODERATE/POOR entrained algae - monitor	POOR	BAD	BAD
500 to <1000 g m²	GOOD	GOOD/MODERATE entrained algae - monitor	MODERATE	POOR	POOR
100 to <500 g m²	HIGH HIGH/GOOD entrained algae - monitor	GOOD	GOOD no entrained algae no monitoring GOOD/MODERATE entrained algae - monitor	MODERATE	MODERATE/POOR entrained algae - monitor
<100 g m²	HIGH	GOOD	GOOD no entrained algae no monitoring GOOD/MODERATE entrained algae - monitor	GOOD/MODERATE entrained algae - monitor	MODERATE

% COVER

Quality Status	Algal Biomass	Algal Cover
High	<100 g m⁻²	<5%
Good	100-500 g m⁻²	5-15%
Moderate	500-1000 g m⁻²	15-25%
Poor	1000-3000 g m⁻²	25-75%
Bad	>3000 g m⁻²	>75%

Figure 5.3.5. Proposed assessment framework to diagnose eutrophication using macroalgae for macroalgae in intertidal and shallow subtidal habitat for the European Water Directive Framework (from Scanlan et al., 2007). Biomass is in wet weight.

The framework uses biomass and percent cover to classify an area within an estuary into one of five categories: High, Good, Moderate, Poor, and Bad. Each of these categories was defined as a deviation from a reference or pristine condition. They used a combination of data and expert opinion to generate their categories and assign threshold values between categories, emphasizing that more work was needed, especially to differentiate between moderate, poor, and bad conditions. Scanlan et al. (2007) emphasized that the proposed threshold values must be validated by examining multiple ecological indicators across the eutrophication gradient.

The Scanlan et al. (2007) assessment framework provides a good conceptual model for how to incorporate both biomass and cover into a diagnostic tool and as such is a good starting point for California, in general, and SF Bay, in particular. However, several caveats should be considered. First, the assessment framework does not explicitly incorporate duration of mat presence into the framework, a factor that has been determined to be important through in situ experiments and published literature (Hull, 1987; Balducci et al., 2001; Osterling and Pihl, 2001; Bolam and Fernandes, 2002) and is likely important for SF Bay. Second, Scanlan et al. (2007) did not clearly specify the geographic scope of these specific thresholds for macroalgal biomass and percent cover. Countries within the European Union span the range from Artic to Mediterranean climates and it is unreasonable to think that, given differences in water temperatures across large area, that some differences in the thresholds for biomass and cover are not warranted. Third, while reasonable, the thresholds are based on best professional judgment with little citation of the actual data used to derive the thresholds. Additional work would need to be conducted to develop an appropriate macroalgal assessment framework for SF Bay.

5.3.7 Summary: Use of Macroalgae as an NNE Indicator

Overall, macroalgae appear to satisfy the four evaluation criteria to be considered as an NNE indicator for SF Bay. However, limited data exist on the distribution and variability in macroalgae in SF Bay in seagrass and intertidal flat habitat as well in the tidally muted portions of the Bay. There are a number of data gaps that would need to be filled in order to develop macroalgae as an NNE indicator:

1. Conduct a comprehensive assessment of macroalgal biomass and cover in SF Bay habitats.

2. Develop an assessment framework for macroalgae, including data to support the development of numeric endpoints. This assessment framework ultimately needs to address effects of macroalgae on seagrass habitat, intertidal flats, and oligohaline subtidal environments. Some of this work has already begun for “other” California estuaries (see Sutula, 2011). The SF Bay TAT recommends evaluating the findings of these planned studies in tandem with better information on macroalgal biomass and cover and revisiting how macroalgae could be incorporated into the SF Bay NNE framework at that time.

3. Scope the development of a macroalgal component within SF Bay water quality models (see Section 5.2.6 for additional details on loading and SF Bay water quality models). If macroalgae is to be included, then scoping should include conceptual model development, understanding of how models under consideration can simulate macroalgae, and a data needs assessment to conduct this work. As with the development of an assessment framework for macroalgae, the SF Bay recommends that this work be considered pending the findings of planned studies for macroalgae being conducted for other California estuaries.

5.4 Submerged Aquatic Vegetation (SAV)

Rooted submerged aquatic vegetation (SAV) encompass a large diversity of species that range from obligate halophytes (e.g., seagrasses, *Zostera marina L., Z. japonica*) to mesohaline and oligohaline species (e.g., *Ruppia maritima L., Vallisneria spp., Stukenia pectinatus*) to freshwater obligates (e.g., *Elodea canadensis, Nuphar spp*.). The primary features distinguishing between groups of SAV are salinity tolerance and pollination vectors. Throughout the course of this review, the term "seagrass" will be applied exclusively to genera that are obligate halophytes, exhibit hydrophylious (underwater) pollination and form meadows; this includes but is not limited to, *Zostera, Phyllospadix, Halodule, Thalassia, Halophil*a, etc. "Brackish SAV" or "aquatic beds" will be applied to genera that are euryhaline species, exhibit aerial or surface pollination and tend to form canopies; this includes but is not limited to *Ruppia, Stukenia, Zannichellia, Myriophyllum,* etc. Seagrass and SAV can form extensive beds (Figure 5.4.1), and can also be found as solitary patches much smaller in size (Merkel & Assoc., 2004a).

Seagrass and SAV have a variety of characteristics that make them good candidates to be "end-points of concern" for eutrophication or "bio-indicators". First, many of these species, especially the seagrasses, are perennial and form persistent rhizomes; consequently they act as "long term integrators" responding to environmental change (Burkholder et al., 2007). Second, as rooted organisms, they are not mobile and cannot move in response to changing environmental drivers. Third, for a number of key seagrass and SAV species (including *Zostera marina L*. and *R. maritima*) the biological and physiological requirements are known well enough to develop models of how the plants respond to stressors. Finally a number of very well designed monitoring programs currently use seagrasses as bio-indicators (Fonseca et al., 2001; Foden and Brazier, 2007; Madden et al., 2009) including government organizations such as Washington State Dept. of Natural Resources[4] and non-governmental organizations (http://seagrassnet.org). Although many estuarine systems do support SAV it is important to recognize that not all systems would be expected to support these plant communities based on the morphology and hydrology of the system.

Kaldy and Sutula provide a detailed review of the utility of seagrass and brackish SAV as an NNE indicator in estuaries (see Chapter 6 in Sutula, 2011). They conclude that there are three types of primary indicators relevant to the assessment of eutrophication in seagrass habitats: 1) macroalgal biomass and cover, 2) phytoplankton biomass and light attenuation, and 3) epiphyte load. Seagrass areal distribution and density can be considered supporting indicators, but are known to respond to a wider number of stressors than just eutrophication per se (e.g., excessive sedimentation, temperature stress, etc.). This section provides a brief synopsis of that work and presents literature directly relevant to known abundance, distribution and variability of seagrass and brackish SAV in San Francisco Bay estuary.

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[4] www.dnr.wa.gov/ResearchScience/Topics/AquaticHabitats/Pages/aqr_nrsh_eelgrass_monitoring.aspx

Figure 5.4.1. Southern End of Eelgrass Bed, between Point San Pablo and Point Pinole (from Merkel & Assoc., 2004a).

5.4.1 Data Available on Seagrass and Brackish SAV

In general there is still little is known about abundance and distribution of SAV in SF Bay. One study exists on four types of SAV communities *Phyllospadix scouleri* (surfgrass), *Zostera marina* (eelgrass), *Ruppia maritima* (widgeongrass), and *Potamogeton pectinatus* (sago pondweed) (Schaeffer et al., 2007). The data set on eelgrass is more comprehensive made possible by the Bay Bridge construction and the associated mitigation related surveys (Merkel & Assoc., 2003, 2010). Surveys of areal distribution are available for the years 1987, 2003, and 2009 (Merkel & Assoc., 2004, 2010) and will continue for at least another 5 years. Aspects of physiology, recruitment, and growth have been studied for eelgrass (Zimmerman et al., 1995; Boyer et al., 2008).

5.4.2 SAV Relationship to Nutrients and Water Quality

Submerged aquatic vegetation (SAV) is an important component of estuarine and coastal nearshore ecosystems and form structural habitats for a diversity of plant and animal species, affect rates of sedimentation and erosion, and influence the structure of inshore benthic communities (see reviews in Zimmerman et al., 1991, 1995). SAV usually grow in unconsolidated anoxic sediments in shallow, calm nearshore areas and are therefore extremely vulnerable to human encroachment from stormwater outfalls, marinas, piers, wharfs, swimming beaches, dredging, trash and other forms of urban pollution. The depth distribution and plant density of SAV is strongly correlated to light availability, temperature, salinity, and tidal forces. Light, however, appears to be the factor that most often controls the depth distribution, density, and productivity of SAV (see review in Zimmerman et al., 1991, 1995). Unfortunately, in SF Bay, high turbidity renders much of the Bay light limited; the euphotic zone (depth where irradiance falls to 1% of surface irradiance) is <1 m in many locations (Alpine and Cloern, 1988; Zimmerman et al., 1991). Observations made by Zimmerman et al. (1991) provided evidence for the hypotheses that 1. *Zostera* populations in the Bay are adapted to low light availability, and 2. *Zostera* in SF Bay may be more controlled by short lived low light pulses of turbid water during high runoff or high wind periods rather than by average annual light conditions. Therefore, it should not be surprising that the majority of SAV beds, dominated by the seagrass *Zostera* spp., found in SF Bay occur in the higher salinity lower turbidity areas in the Central Bay at depths <2 m mean lower low water (MLLW) and at some sites at depths only shallower than 1 m MLLW (Zimmerman et al., 1991).

The response of eelgrass (*Zostera marina*) to increased nutrient loading and eutrophication has been a major research focus over the past few decades; however, the majority of this work has occurred on the east coast (Nixon et al., 2001; Burkholder et al., 2007). It has been shown that increases in nutrient loading causes degradation in *Z. marina* through the stimulation of algal production (micro- and macroalgae) and shading out seagrass (Havens et al., 2001). Algae are stronger competitors than seagrass, and when nutrients are increased, a phase shift from seagrass dominance to either phytoplankton or macroalgae dominance can occur (Burkholder et al., 2007). However, Nixon et al. (2001) did not find a predictable relationship between algal type (phytoplankton vs. microphytobenthos vs. macroalgae) and nutrient levels; a number of biotic and abiotic factors contribute to this complex relationship. More often, the impact of nutrient enrichment on seagrass is usually observed through indirect effects, although there have been direct effects reported. Burkholder et al. (2002) performed mesocosm experiments in North Carolina and found that increased nitrate levels in the water column led to declines of eelgrass, independent of macroalgal shading. They suggested that the impact was due to direct physiological effects associated with internal imbalances in nutrient ratios from sustained nitrate intake through the leaves. Nutrient issues for eelgrass in San Francisco Bay have been reviewed by Boyer and Wyllie-Echeverria (2010). They suggested there is evidence for a lack of N-limitation except in a few locations but more work needs to be done. Recently Carr et al. (2011), presented a new analysis which included some data epiphyte loads (as chl a) on eelgrass in SF Bay. Overall, there is consensus that turbidity is the most limiting factor for eelgrass in San Francisco Bay. However, recent and sustained downward trends in suspended sediment concentrations might mean that factors like phytoplankton might become the leading factors in the future (Katharyn Boyer, personal communication, March 2011). Overall, the effects of nutrients of eelgrass are not well established for the Bay. In a recent study in Tomales Bay, macroalgae was found in abundances that were determined to have an adverse effect on eelgrass density and growth rates (Huntington and Boyer 2009). In SF Bay, Boyer and Wyllie-Echeverria (2010) found biomass of macroalgae generally low in eelgrass beds, but occasionally levels approached biomass found to be detrimental to eelgrass in Tomales Bay; further work is recommended. Presently there are a number of ongoing studies being overseen by Katharyn Boyer. These include chlorophyll a data from the water column, surface sediments, and ammonium and nitrate data from four eelgrass beds and about a year of continuous dissolved oxygen data from two seagrass beds (Katharyn Boyer, personal communication, March 2011).

Given the habitat benefits associated with eelgrass beds, there are efforts to restore areas formally thought to have supported eelgrass. However, efforts, thus far, to restore or increase the area of existing meadows have been met with mixed success. In a study of eelgrass (*Zostera marina L.*) transplants in SF Bay, Zimmerman et al. (1995) found that despite the period of favorable light levels due to low Sacramento River runoff in 1989 and 1990, 40% of the transplants were lost within the depth range of the native populations (-0.5 to -1.0 m depth). Despite these losses, self-sustaining beds were established and observed over a four year period and the authors concluded that transplanting should be viable given sufficient plant C reserves and light availability (Zimmerman et al., 1995). Using new seeding techniques, Buoy-Deployed Seeding, Boyer et al. (2008) were able to recruit and establish *Zostera marina* at two restoration sites in the SF Bay and they believe that seedling recruitment will continue to contribute to eelgrass cover in the next several years. They observed fish and amphipods in the restored beds which signify that restored patches were beginning to serve as habitat for native and eelgrass dependent species (Boyer et al., 2008).

5.4.3 Species Composition

Although there is developing information on SAV in SF Bay, in general there is still little is known about abundance and distribution. There are four types of SAV communities found in the SF Bay (Figure 5.4.5: *Phyllospadix scouleri* (surfgrass), *Zostera marina* (eelgrass), *Ruppia maritima* (widgeongrass), and *Stuckenia pectinata* (sago pondweed)) (Schaeffer et al., 2007). Based on the recent surveys of SAV in the Bay, eelgrass is the most widely distributed and most abundant (Merkel & Assoc., 2003, 2010; Schaeffer et al., 2007). Eelgrass is the most commonly studied form of SAV in the SF Bay. Sago pondweed has long rhizomes and runners which allow it to better tolerate strong currents and wave action. Eelgrass has all of its life cycle stages occur underwater, including seed germination, flowering, and pollination. Widgeongrass can grow in both freshwater and high salinity environments (Schaeffer et al., 2007).

5.4.4 Trends and Factors Effecting Temporal and Spatial Variation of SAV

Zostera marina beds vary greatly during a single season in shoot density and between years in aerial extent. Several factors have been found to influence this variation including temperature and light conditions in relation to variations in water clarity (turbidity) (Merkel & Assoc., 2010). Given turbidity is gradually decreasing in the Bay (Schoellhamer, 2009) associated with decreasing sediment loads entering the Bay from the Central Valley (McKee et al., 2006); it is possible that expansion will continue as an overall trend. However, annual variations in light conditions will continue to cause inter-annual fluctuations.

Spatial Variability

Eelgrass beds are currently found in euhaline to polyhaline environments in the SF Bay which is mainly along the eastern shores of San Pablo and Central Bays. The majority of eelgrass in the Bay and the largest eelgrass bed is found between Point Pinole and Bayfarm Island (Figure 5.4.2) (Merkel & Assoc., 2010). In another example, the eelgrass patch on the southern side of the Richmond shipping channel is also large but has decreased in size by 11.9 ac (4.8 ha) over a two year period. During the same time frame, the patch at Keller Beach increased in size by 10.4 ac (4.2 ha) (Merkel, 1999 cited in Wyllie-Echeverria and Rutten, 1989).

Using data from the 2003 and 2009 surveys performed by Merkel & Assoc., the depth distribution of eelgrass was mapped (Figure 5.4.3). The depth distribution of eelgrass is narrow and it also shows the turbid nature of the SF Bay (Merkel & Assoc., 2010). The likely factor controlling the spatial distribution of eelgrass in the Bay is water clarity and turbidity. In the North and South Bay there are low light levels due to the large input of sediment per year (on average 1 million mt per year: McKee et al., 2006) from freshwater rivers. Additional suspended sediment occurs from tidal currents and wind driven waves (Merkel & Assoc., 2010).

Temporal Variability

Eelgrass distribution varies greatly within and between years in association with light and temperature conditions. For example, at the control site of the Richmond training wall transplanting experiment study, shoot density was almost 10 times greater in April 1985 than that observed in July 1985. Shoot densities for September 1985 and May 1986 were 30% and 48 % of the April densities respectively (Fredette et al. 1987 cited in Wyllie-Echeverria and Rutten, 1989). According to Keith Merkel (personal communication, 2011) monitoring performed in 2010 suggests a slight increase in eelgrass from 1,500 ha to approximately 1,522 ha in the SF Bay. He also anticipates that there will be a substantial drop in eelgrass abundance in the next year due to his anticipation of relatively high sediment loading from the 2010/11 wet weather period. In general, for a few years after a large sediment loading event, sediment resuspension will cause a decline in eelgrass, especially where wind driven resuspension and limited flushing occurs. This appears to be supported by a preliminary relationship between suspended sediment loads entering the Bay from the Central Valley via the Delta (McKee et al., 2006), and eelgrass extent data collected on three occasions beginning 1987 (Figure 5.4.4). There are of course flaws in this very simple model; it suggests a linear relationship, whereas more likely maximum possible eelgrass extent is greater than 4920 acres (possibly as high as 20,000 acres (8,100 ha) (Merkel & Assoc. 2004), and it is unlikely that total extirpation would occur if a future 3-year averaged sediment load exceeds 0.7 million mt, since there has been many times in the past 30 years when that has likely happened and 7 times since 1997 when we know it has happened (McKee et al., 2006).

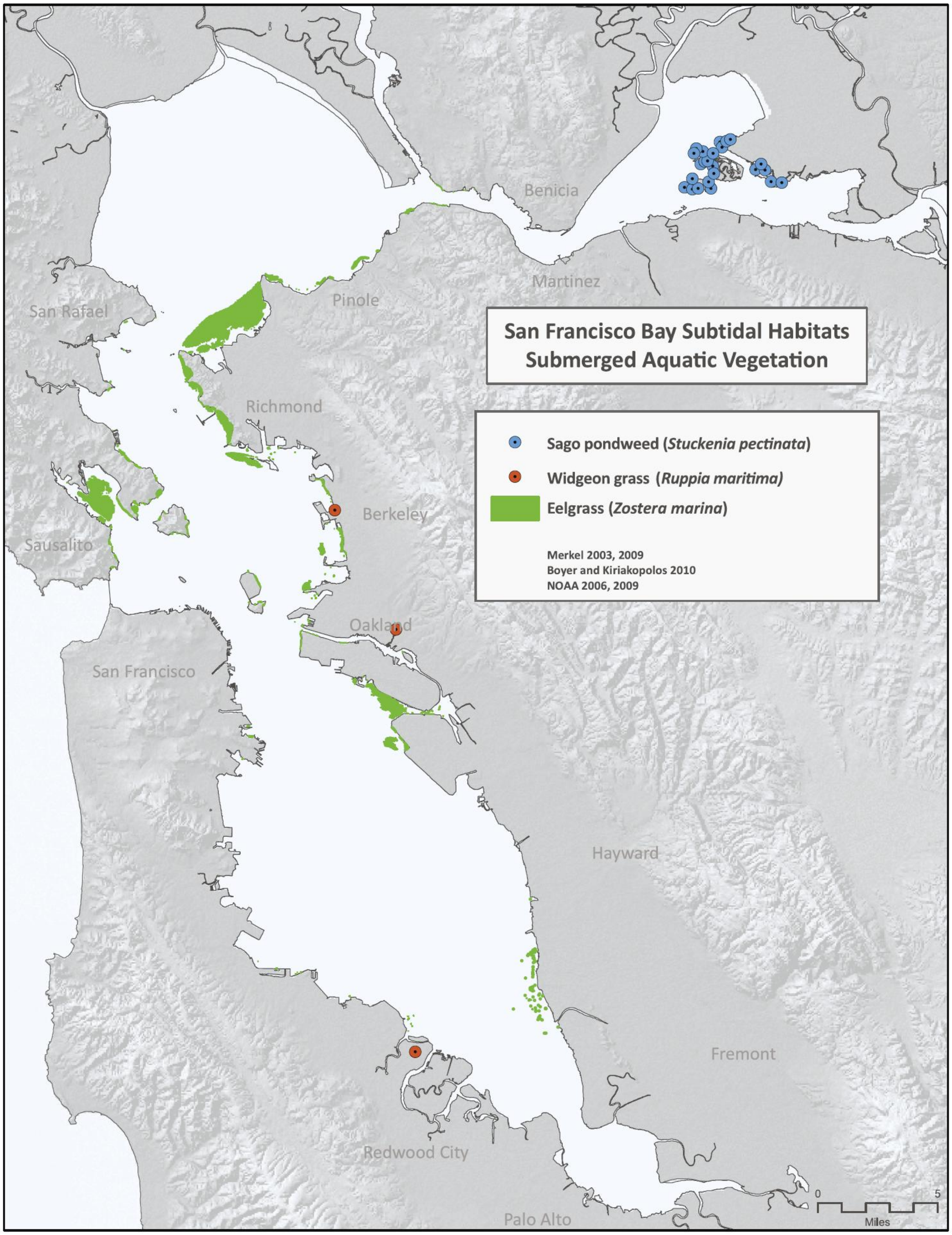


Figure 5.4.2. The distribution of three different SAV communities in San Francisco Bay (SSC, 2010).

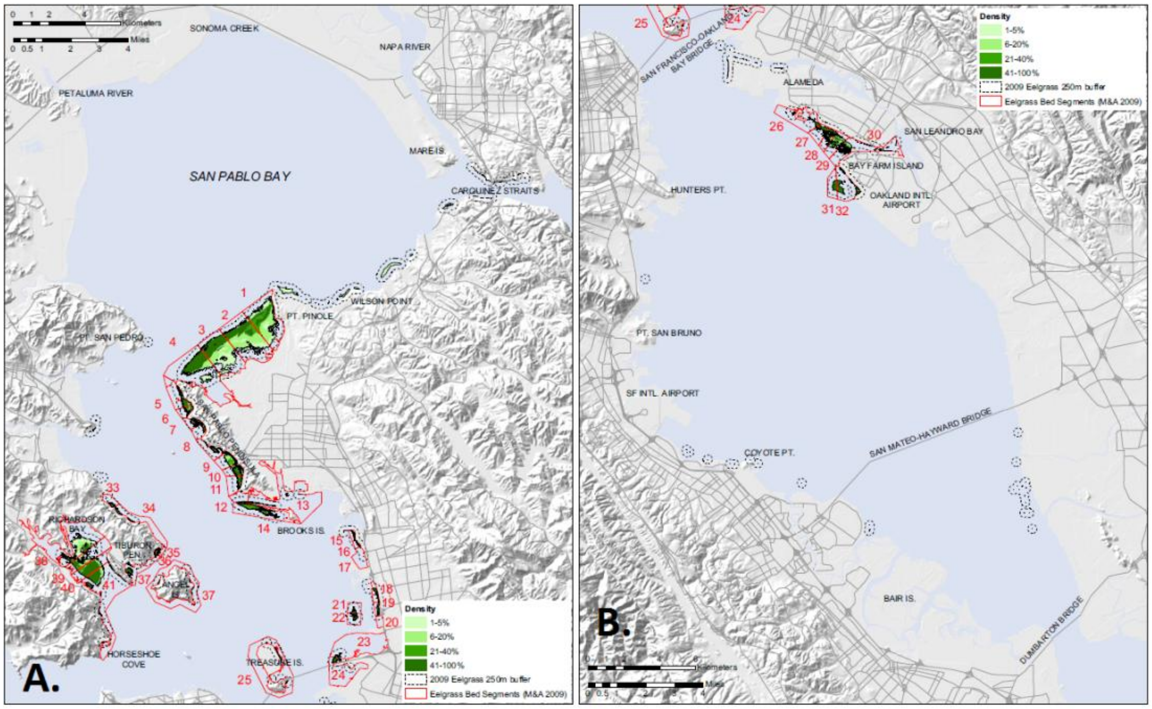


Figure 5.4.3. 2009 distribution of eelgrass meadows (*Zostera marina L.*) in A. North and Central San Francisco Bay and B. South San Francisco Bay (Source Merkel & Assoc. 2010). Note – the numbers on the Figure refer to the numbers and text in Table 5.4.2.

SAV Trends

The first formal recorded study of eelgrass in SF Bay was that of Setchell (1922) who studied *Zostera marina* patches in Keil and Paradise coves on the eastern coastline of Tiburon. Unfortunately, that we can determine, no Bay wide estimates of total bed area were published at the time. However, given the massive fluxes of sediment that were coming into the Bay in the early 1900s in response to landscape disturbances during the gold rush and as the Central Valley was opening up for agriculture (McKee et al., 2006; Jaffe et al., 2007; Ganju et al., 2008), that any eelgrass was observed at all is perhaps miraculous. A long hiatus for some 50 years occurred before research interests picked up in relation to managing and monitoring diminishing beds, spawning habitat value for Pacific herring, and ongoing deepening and widening projects in relation to shipping (Wyllie-Echeverria and Fonseca, 2003). In 1989, a study revealed that eelgrass populations were discontinuous and found on the southern shorelines of San Pablo Bay and northern reaches of central Bay and in the northern reach of South SF Bay in 23 locations ranging in size from 0.5 ac (0.2 ha) to 124 ac (50 ha) with a total estimate of 316 ac (128 ha) (Wyllie-Echeverria and Rutten, 1989). In 1993, these aerial distributions were combined with estimates of carbon production from literature (300 g $m^{-2}y^{-1}$) to estimate a total baywide carbon production from eelgrass of 384 mt (Jassby et al., 1993) or <0.2% of total autochthonous productivity at that time.

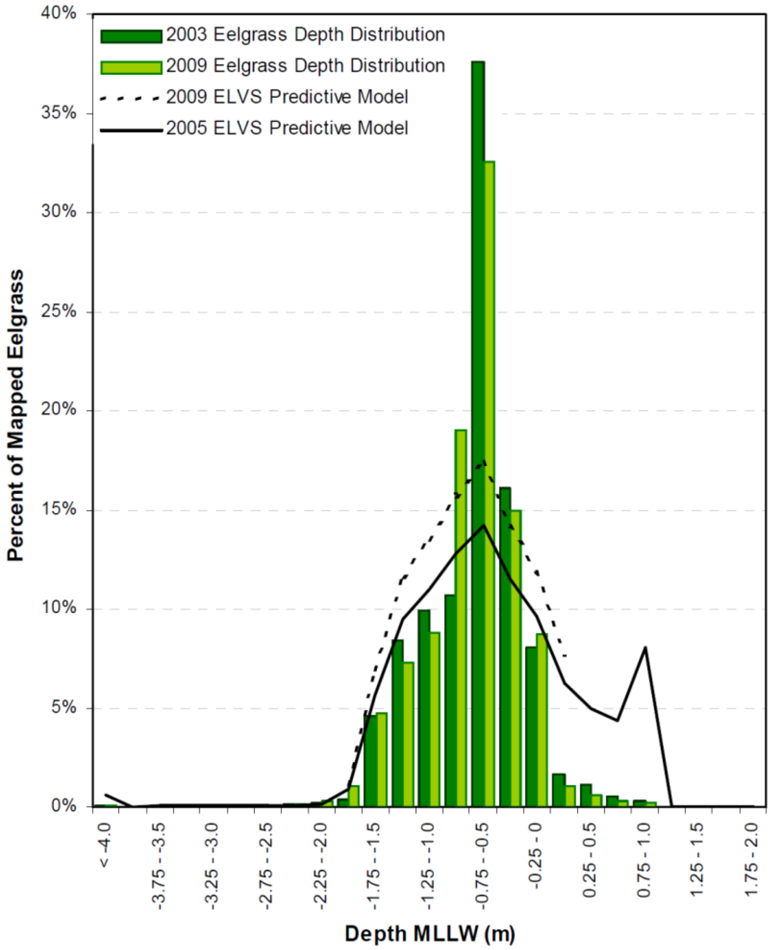


Figure 5.4.4. Eelgrass depth distribution in San Francisco Bay (from Merkel & Assoc., 2010).

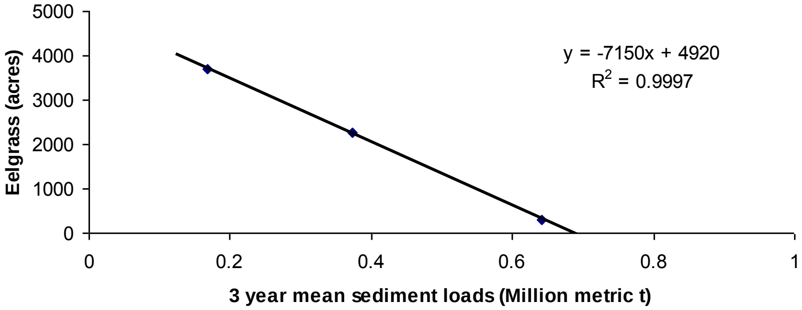


Figure 5.4.5. The quasi-relationship between eelgrass area and three- year mean sediment loads in the San Francisco Bay using estimated and measured sediment loads from the Central Valley to the Bay and the equation from McKee et al., 2006.

More recently eelgrass has been surveyed extensively in relation to the construction of the new eastern span of the Bay Bridge. To reduce time and resources needed to accurately map eelgrass in SF Bay, a prescreening model and aerial survey was used by Merkel & Associates (2003) to reduce the area of detailed survey from ~100,000 ha of sub- and inter-tidal habitat to 20,000 acres (8,100 ha) of possible habitat which was subsequently surveyed using vessel-deployed acoustic techniques. Based on this program, new estimates of eelgrass habitat are now a little less than 1,200 ha (Merkel & Assoc., 2004) (Tables 5.4.1 and 5.4.2). While their methods are not absolute, diver surveys are done to confirm there is eelgrass at a given site (as opposed to macroalgae or *Ruppia*). It is possible that small patches are missed and some deeper patches are over estimated but over all there is good confidence in the data (Katharyn Boyer, Romberg Tiburon Center for Environmental Studies, San Francisco State University personal communication, January 15th 2011).

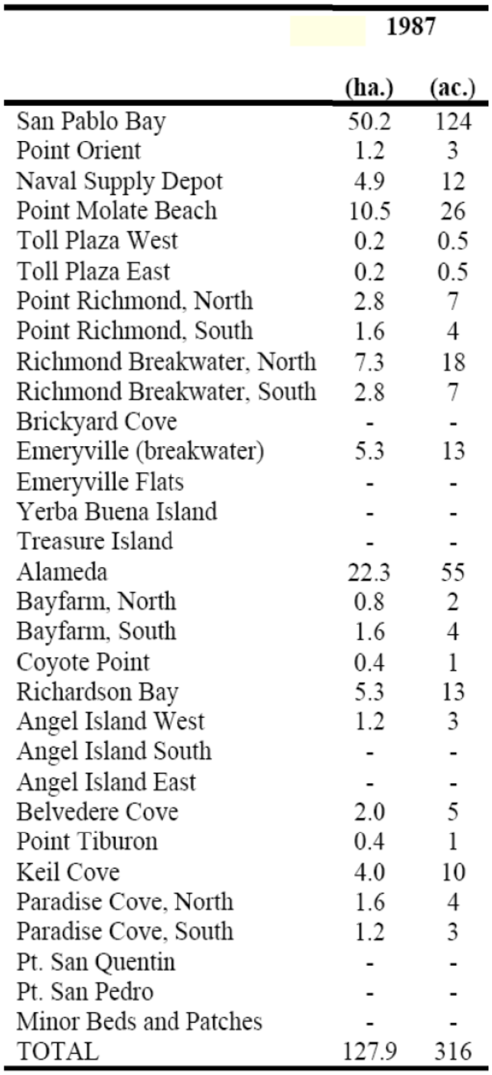
The survey completed in 2009 by Merkel & Associates provides estimates of the eelgrass habitat area of 1,500 ha, a 28.7% increase in overall eelgrass coverage since the 2003 survey (Merkel & Assoc., 2004) and nearly a 1,200% increase since the 1987 survey (Tables 5.4.1 and 5.4.2). An expansion of eelgrass northward into the Carquinez straights and an increase along both the shoreward (shallower) and bayward (deeper) edges of existing eelgrass beds are major differences between the 2003 and 2009 surveys. These increases were not observed in the southern reaches of the Bay, where eelgrass remained comparably or even less abundant than the 2003 surveys had shown.

5.4.5 Utility of Seagrass and Brackish SAV as an Eutrophication Indicator for San Francisco Bay

Clear Linkage to Beneficial Uses

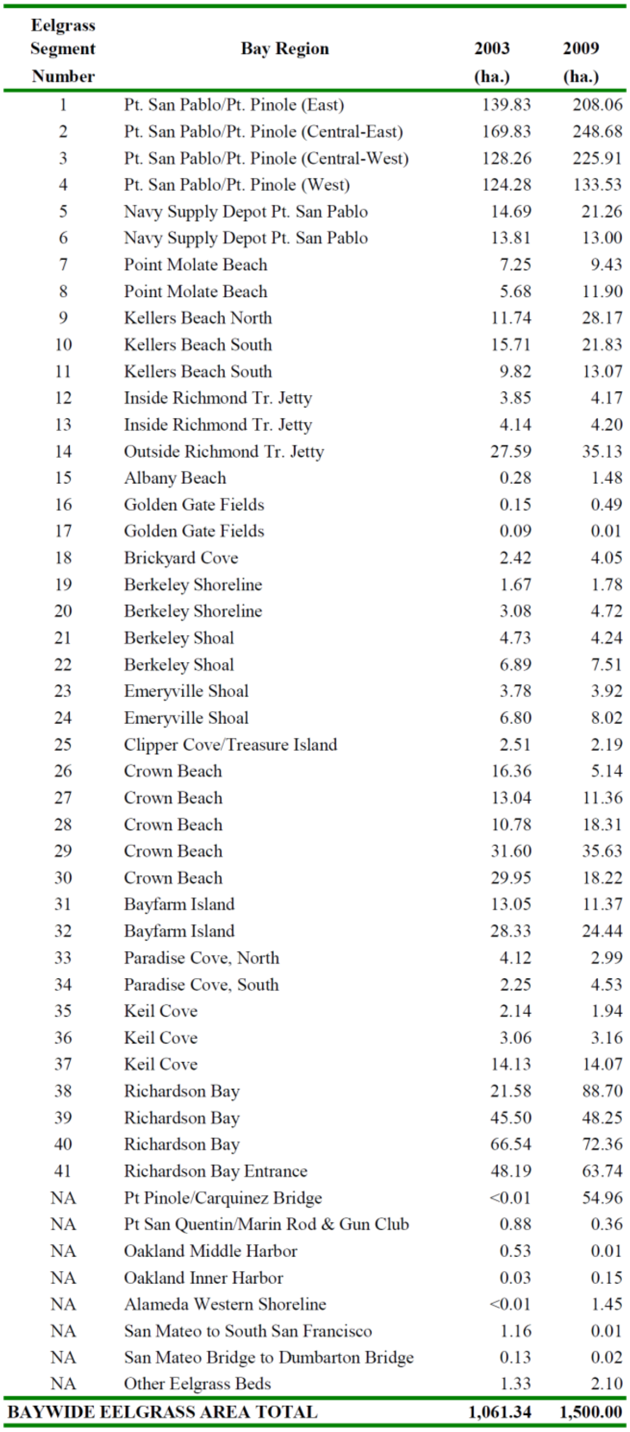
Seagrasses and other SAV are considered to be community structuring plant that forms expansive meadows or smaller beds. As a result, they are considered to be “habitat forming” species that creates unique biological, physical, and chemical environments when it occurs in the forms of submerged or intertidal aquatic beds or larger meadows. Eelgrass beds are important ecological communities of shallow bays and estuaries because of the multiple ecological services they sustain (Orth et al., 2006). Seagrass and SAV are designated marine and/or estuarine habitat that have an obligate requirement for seawater (MAR and EST beneficial uses). They are also wildlife habitat particularly waterfowl and shorebirds (WILD beneficial use). Seagrass and SAV beds function as habitat and nursery areas for commercially and recreationally important open ocean marine and estuarine fish and invertebrates, and provide critical structural environments for resident bay and estuarine species. Many commercially and recreationally (COMM beneficial use) important fisheries species have a life-history stage that is estuarine dependent and many of them utilize seagrass beds; examples include, salmonids, herring, Dungeness crab, shellfish (Blackmon et al., 2006). Seagrass also support shellfisheries (SHELL beneficial use), as a variety of bivalves used for human consumption and bait occur in seagrass beds. Presence of seagrass can influence the population structure and growth rates of clams (Peterson et al., 1984); additionally seagrass patch size and structural characteristics affect bivalve survivorship (Irlandi, 1997). Peterson and Heck (2001) suggest that bivalves and seagrass have positive interactions resulting in a facultative mutualism.

Table 5.4.1. Trends in the aerial extent of eelgrass meadows (*Zostera marina L.*) in San Francisco Bay between 1987 and 2003 (Merkel & Assoc., 2004).



	1987	
	(ha.)	(ac.)
San Pablo Bay	50.2	124
Point Orient	1.2	3
Naval Supply Depot	4.9	12
Point Molate Beach	10.5	26
Toll Plaza West	0.2	0.5
Toll Plaza East	0.2	0.5
Point Richmond, North	2.8	7
Point Richmond, South	1.6	4
Richmond Breakwater, North	7.3	18
Richmond Breakwater, South	2.8	7
Brickyard Cove	-	-
Emeryville (breakwater)	5.3	13
Emeryville Flats	-	-
Yerba Buena Island	-	-
Treasure Island	-	-
Alameda	22.3	55
Bayfarm, North	0.8	2
Bayfarm, South	1.6	4
Coyote Point	0.4	1
Richardson Bay	5.3	13
Angel Island West	1.2	3
Angel Island South	-	-
Angel Island East	-	-
Belvedere Cove	2.0	5
Point Tiburon	0.4	1
Keil Cove	4.0	10
Paradise Cove, North	1.6	4
Paradise Cove, South	1.2	3
Pt. San Quentin	-	-
Pt. San Pedro	-	-
Minor Beds and Patches	-	-
TOTAL	127.9	316

Table 5.4.2. Trends in the extent of eelgrass (*Zostera marina*) in the San Francisco Bay during the 2003 (Merkel & Assoc., 2004) and the 2009 surveys (Merkel & Assoc., 2010).



Eelgrass Segment Number	Bay Region	2003 (ha.)	2009 (ha.)
1	Pt. San Pablo/Pt. Pinole (East)	139.83	208.06
2	Pt. San Pablo/Pt. Pinole (Central-East)	169.83	248.68
3	Pt. San Pablo/Pt. Pinole (Central-West)	128.26	225.91
4	Pt. San Pablo/Pt. Pinole (West)	124.28	133.53
5	Navy Supply Depot Pt. San Pablo	14.69	21.26
6	Navy Supply Depot Pt. San Pablo	13.81	13.00
7	Point Molate Beach	7.25	9.43
8	Point Molate Beach	5.68	11.90
9	Kellers Beach North	11.74	28.17
10	Kellers Beach South	15.71	21.83
11	Kellers Beach South	9.82	13.07
12	Inside Richmond Tr. Jetty	3.85	4.17
13	Inside Richmond Tr. Jetty	4.14	4.20
14	Outside Richmond Tr. Jetty	27.59	35.13
15	Albany Beach	0.28	1.48
16	Golden Gate Fields	0.15	0.49
17	Golden Gate Fields	0.09	0.01
18	Brickyard Cove	2.42	4.05
19	Berkeley Shoreline	1.67	1.78
20	Berkeley Shoreline	3.08	4.72
21	Berkeley Shoal	4.73	4.24
22	Berkeley Shoal	6.89	7.51
23	Emeryville Shoal	3.78	3.92
24	Emeryville Shoal	6.80	8.02
25	Clipper Cove/Treasure Island	2.51	2.19
26	Crown Beach	16.36	5.14
27	Crown Beach	13.04	11.36
28	Crown Beach	10.78	18.31
29	Crown Beach	31.60	35.63
30	Crown Beach	29.95	18.22
31	Bayfarm Island	13.05	11.37
32	Bayfarm Island	28.33	24.44
33	Paradise Cove, North	4.12	2.99
34	Paradise Cove, South	2.25	4.53
35	Keil Cove	2.14	1.94
36	Keil Cove	3.06	3.16
37	Keil Cove	14.13	14.07
38	Richardson Bay	21.58	88.70
39	Richardson Bay	45.50	48.25
40	Richardson Bay	66.54	72.36
41	Richardson Bay Entrance	48.19	63.74
NA	Pt Pinole/Carquinez Bridge	<0.01	54.96
NA	Pt San Quentin/Marin Rod & Gun Club	0.88	0.36
NA	Oakland Middle Harbor	0.53	0.01
NA	Oakland Inner Harbor	0.03	0.15
NA	Alameda Western Shoreline	<0.01	1.45
NA	San Mateo to South San Francisco	1.16	0.01
NA	San Mateo Bridge to Dumbarton Bridge	0.13	0.02
NA	Other Eelgrass Beds	1.33	2.10
BAYWIDE EELGRASS AREA TOTAL		**1,061.34**	**1,500.00**

Besides providing important habitat for fish, seagrass and SAV are considered to be an important resource supporting migratory birds and spawning fish during critical life stages. Bortolus et al. (1998) found that *Ruppia maritima* was an important food sources for a variety of waterfowl species in Argentina; including swans and ducks. Along the Pacific flyway, both *Ruppia maritima* and *Z. marina* are food resources for Black Brant geese (Ward, 1983; Derksen and Ward, 1993; Moore et al., 2004). Seagrass and SAV meet the spawning beneficial uses as they provide a refuge for anadromous fish (salmonids) particularly during the transition from freshwater to seawater (see reviews in Kennedy, 1982 and Blackmon et al., 2006). Seagrass and SAV habitat provide a direct food source for migrating waterfowl (Moore et al., 2004) as well as an acclimation refuge for anadromous fish species (Blackmon et al., 2006), thus linking to MIGR, SPWN, and RARE beneficial uses. Healthy Seagrass and SAV support REC-2 beneficial uses in a number of ways. These habitats are prime areas for recreational crabbing and fishing as well as kayaking and waterfowl hunting and a focus of marine studies programs in the SF Bay as well as elsewhere in California.

Predictive Relationships to Causal Factors

Under oligotrophic conditions, increased nutrient loads may initially be beneficial to seagrass communities by stimulating primary production, leading to greater secondary production by consumers. However, under continued high nutrient loads, algae are superior competitors and their increased abundance can be deleterious to seagrass. Initial indications of eutrophication issues include decreased bed density and increased abundance of the algal flora. Under very high nutrient loading, the system can become dominated by algal competitors (phytoplankton, epiphytes or macroalgae) resulting in the degradation or loss of the seagrass community (Figure 5.4.6). The primary mechanism of seagrass loss is through light reduction caused by shading or smothering from algal competitors. Reduced light coupled with increased delivery of labile organic detritus (senescent algae and seagrass) to the sediments can lead to additional biogeochemical stressors (hypoxia/anoxia, sulfide toxicity, etc.) that further exacerbate the problem. Consequently, there tends to be a positive feedback loop between nutrient enrichment and expression of eutrophic or dystrophic conditions. Degraded seagrass beds tend to be sparse or patchy, heavily epiphytized with macroalgae and experience large diurnal swings in dissolved oxygen concentrations.

Response of seagrass to nutrient loading and eutrophication has been a major research focus over the last couple decades (Nixon et al., 2001; Nielsen et al., 2004a; Burkholder et al., 2007; and many others). Most seagrass eutrophication studies have examined the community level response in experimental systems ranging from aquaria to mesocosms to the natural environment (Table 6-1). For *Z. marina* much of this work has been conducted along the East Coast of North America and has resulted in a general theory of seagrass response. Specifically, that enhanced nutrient loading leads to a degradation of *Z. marina* habitat (Figure 5.4.6) by stimulating algal production (micro- and macroalgae) and shading out seagrass (Short et al., 1991, 1995; McGlathery, 2001; Havens et al., 2001).

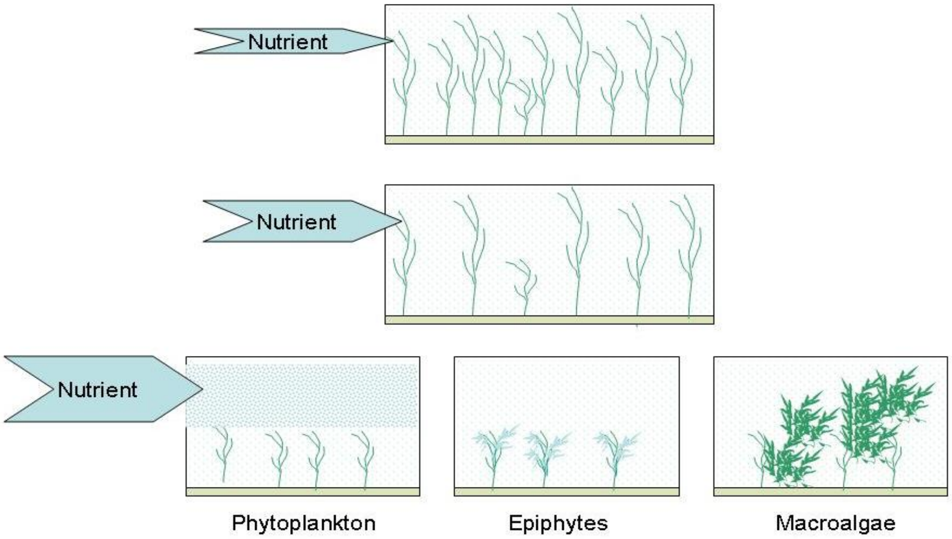


Figure 5.4.6. Conceptual model of how seagrass and some SAV communities respond to increased nutrient loading and resulting eutrophication. From Kaldy and Sutula (Chapter 5, Sutula, 2011).

A number of dose response experiments have been undertaken with *Z. marina*; however, most studies have been monocultures in experimental mesocosm experiments and this work has been primarily conducted with plants from the North Atlantic populations (Burkholder et al., 2007 and references therein). The experimental mesocosm and load response experiments clearly show that the taxonomy of the community shifts from seagrass dominance to either macroalgae or phytoplankton dominance (Burkholder et al., 2007). Field based sampling along eutrophication gradients (e.g., Waquoit Bay, MA) and field manipulations of water column nutrients exhibit similar patterns (Burkholder et al., 2007). Thus the seagrass declines through indirect effects on the seagrass (e.g., shading, increased hypoxia, increased respiration, etc.). However, there does not appear to be a predictable trajectory of development between nutrient input and the algal type (epiphyte vs. macroalgae vs. phytoplankton) that can adversely affect seagrass (Nixon et al., 2001). That is for any given load of nutrients there is no capacity to predict if the system will become dominated by macroalgae, epiphytes (e.g., microphytobenthos) or phytoplankton.

The ultimate primary producer dominance is determined by initial starting condition of the system (spore or propagules availability) interacting with various biotic and abiotic cofactors. Additionally, since seagrass occur in shallow subtidal habitats, characterization of external loads versus internal recycling of nutrients through the sediments is important for achieving a predictable load-response. These data gaps would effects the accuracy and precision of dynamic simulation models that attempt to capture the relationship between seagrass community structure and nutrient loads and other co-factors. In general then, an understanding of the relationship between nutrient loading and algal primary producers within seagrass habitats is a data gap that requires additional research.

Dynamic simulation models (as ecosystem or stress-response) have been used in many instances to develop and evaluate nutrient and other water quality criteria and restoration potential. The Chesapeake Bay approach has been to define the water quality parameters (e.g., light, temperature, salinity, nutrients) associated with SAV declines and to develop a suite of environmental characteristics that are protective of the resource and allow restoration of SAV habitat (Batuik et al., 1992, 2000). As part of this approach, light attenuation by epiphyte load, in addition to light attenuation by the water column (k_d) from water column suspended sediment and phytoplankton, is considered in efforts to evaluate SAV light requirements. These relationships have been based on extensive published and unpublished data sets developed over more than 30 years of research on a single, albeit large estuary. A dynamic simulation model is then used to model the relationship between the seagrass habitat and factors that control light availability, and other stressors that require management (nutrient loads, sediment inputs, etc.). Similarly, the USEPA Western Ecology Division has developed and used a seagrass stress response model to evaluate the impact of proposed nutrient criteria on eelgrass in a Pacific Northwest estuary (Brown et al., 2007). Potential nutrient criteria were developed using the *in situ* observations as a basis for the "Estuarine Reference Condition" using cumulative distribution functions. Proposed criteria were then incorporated into the *Zostera* stress response model to determine whether particular percentile values would adequately protect *Zostera* resources within the estuary (Brown et al., 2007). Based on this data analysis and modeling exercise, median values of most water quality parameters were protective of seagrass habitat.

Some work has been done on the direct effects of nutrients on seagrasses. Tenant (2006) conducted an *in situ* fertilization experiment in Humboldt Bay and concluded that phosphate toxicity explained field observations. The study suffers from a poor design and lack of data to evaluate the sediment nutrient pools as well as the tissue nutrient pools. Claims of phosphate toxicity are premature and not supported by the primary literature. Work from North Carolina and Europe suggests that some populations of *Z. marina* may exhibit declines in response to low level concentrations of NO_3 or NH_4 (Burkholder et al., 1992, 1994; van Katwijk et al., 1997). However, these conclusions may be confounded by other factors (e.g., high temperatures and reduced light). Oregon populations of *Z. marina* are regularly exposed to >30 µM NO_3 from coastal upwelling (Kaldy and Lee, 2007; Brown and Ozretich, 2009) and have shown no declines associated with nitrate toxicity. Furthermore, exposure to 1000 µM NO_3 for two weeks in a laboratory experiment (temp 8°C, 12:12 L:D with saturating irradiance) did not produce mortality or evidence of stress (J. Kaldy, USEPA, unpublished data). Additionally, this seagrass-nutrient loading paradigm from east coast systems may not be directly transferable to California or the West Coast in general.

There are a variety of local and regional processes that effect nutrient dynamics on the west coast that are not as prevalent on the east coast. For example, seasonally closed estuaries and large scale upwelling are not primary features in East coast systems and as a result are not considered in many of the conceptual models that develop from research in that region. Other factors include land use patterns and the fact that in some portions of *Zostera's* range "natural" nutrient loads far exceed those calculated for even the most eutrophic systems on the East Coast (e.g., Waquoit Bay, MA).

There have been relatively few studies of *Zostera marina* nutrient interactions on the West Coast or in California. In a field study from Padilla Bay, WA, Williams and Ruckelshaus (1993) demonstrated that eelgrass growth was influenced by both sediment nutrient availability and higher order effects of epiphytes and isopod grazer interactions. They conclude that "Consideration of sediment nitrogen, epiphytes or herbivores alone is unlikely to yield a predictable understanding of the control of eelgrass productivity in nature, particularly given the complexity of the eelgrass habitat with respect to its dual nutrient sources" (Williams and Ruckelshaus, 1993). This conclusion suggests that external nutrient loads and internal recycling alone do not control eelgrass growth and production, but that integrated water quality, biological and environmental factors play a considerable role (Koch, 2001). Therefore, the use of seagrass health as a primary indicator of eutrophication is problematic in that other stressors (temperature, excessive sedimentation, climate change) may be adversely impacting health.

Estuarine macrophyte community response to nutrient loading occurs primarily as a shift in dominant primary producers. Macroalgae interact with *Zostera* spp. in a variety of ways. An overabundance of macroalgae can cause degradation of *Zostera* habitat through two related mechanisms. First, mats or rafts of algae can develop over-topping or among seagrass shoots that effectively limit the amount of light available to seagrass. However, some systems can exhibit large accumulations of macroalgae growing among *Z. marina* shoots with no apparent decline in seagrass condition driven by seasonal upwelling of nutrients (J. Kaldy, USEPA, pers. observation). Secondly, mats or rafts of algae that settle on top of the seagrass and effectively smother the plants, cutting off light and oxygen leading to anaerobic conditions with a build of toxic metabolites (e.g., sulfides).

Work conducted in European estuaries indicates that the brackish SAV species *Ruppia* spp. acts as a seagrass analog and is susceptible to degradation based on the same types of interactions (shading, smothering, biogeochemical stressors, etc.). There are several examples from Europe that examine how the systems respond to losses of *Ruppia* associated with eutrophication or other anthropogenic activities (Bachelet et al., 2000; Lenzi et al., 2003; Pergent et al., 2006; Shili et al., 2007). Bachelet et al. (2000) investigated an eutrophication gradient along the coast of France; the intermediate site was characterized by *Ruppia* with a constant biomass with sporadic spring blooms of macroalgae. In contrast, the eutrophic site was dominated by macroalgae and had low biomass and abundance of macrozoobenthos (Bachelet et al., 2000). In the Orbetello lagoon (Italy), eutrophication abatement measures (macroalgal harvesting, increased circulation and waste water phytotreatment) resulted in reductions of algal biomass and increased seagrass (Giusti and Marsili-Libelli, 2005). More recent macroalgal blooms appear to be a "legacy effect" of sediment nutrient release (Lenzi et al., 2003). At Biguglia lagoon in Corsica, a healthy bed of *R. cirrhosa* was replaced by Ulvoid algae between 1997 and 1998 with re-appearance of *R. cirrhosa* in 1999. Pergent et al. (2006) attribute these shifts to nutrient availability related to agricultural runoff and wastewater discharge. Several studies have investigated the response of *Ruppia* spp. nutrient loading; however, these studies use a gradient approach where there is little control over or quantification of the loading to the system. Thus, for European *Ruppia*, well defined load -response experiments do not appear to exist. The USEPA Chesapeake Bay Program explicitly assumes that all SAV species follow the same conceptual model where nutrients increase light attenuation by phytoplankton and epiphytes leading to declines of SAV (Batuik et al., 2000).

Manipulative experiments in Maryland concluded that epiphytes, stimulated by nutrient additions, caused declines in *Potamogeton perfoliatus* (Staver, 1984).

In the Delta and in southern and central California lagoons, very dense and apparently healthy brackish SAV populations appear to persist under very eutrophic conditions (high nutrient loading, high organic loading to the sediments, fish kills, large diurnal dissolved oxygen swings, etc.). The presence of dense brackish SAV beds has been observed primarily in Southern California ICOLL (Sutula & McLaughlin, SCCWRP, unpublished data), but also occurs in the Klamath River in Northern CA (Lee and Brown, USEPA, unpublished data). It is not clear if these beds are adapted to and thrive under high nutrient conditions or if these populations are an expression of eutrophication symptoms. The beds tend to be seasonal and it is unknown what triggers the reduction of biomass and subsequent decline of these apparently annual populations. Alternatively, the presence of these dense, ephemeral California populations may be an expression of the natural life-cycle of this species. In Chesapeake Bay, the growth form of seagrass and SAV are classified as “canopy forming” and “meadow forming,” respectively (Batuik et al., 2000). Brackish SAV species tend to be “canopy formers” with biomass concentrated in the top half of the water column and exhibit rapid growth toward the surface early in the growing season. Canopy formation results in shading of older portions and the sloughing of lower leaves. Epiphytes accumulate on the older portions of the leaves and continued growth results in epiphyte free apical leaves near the surface of the water that actively photosynthesize. In contrast, “meadow forming” species concentrate biomass in the lower portion of the water column and new leaf production occurs near the base of the plant. Older leaf tissue near the surface may be heavily epiphytized but rapid leaf turn-over rates allow the plants to maintain positive carbon balance. Additionally, it should be noted, that changes in the distribution of *Ruppia*, and probably other brackish SAV as well, can be related to factors other than nutrients and eutrophication. In the Ichkeul lagoon (Tunisia) rapid changes (1993-1998) in the species composition and distribution of SAV, including *Ruppia cirrhosa* were linked primarily to water management activities (e.g., dams) coupled with drought and not eutrophication (Shili et al., 2007). In San Diego, California, a shift in community dominance from *Z. marina* to *R. maritima* in San Diego Bay were likely related to increased water temperature associated with the 1997-1998 El Niño event (Johnson et al., 2003).

In general, a better understanding of the response of brackish SAV to alterations in nutrient loading requires substantial research before it could be used as an indicator. Key research questions that need to be addressed before brackish SAV will be useful indicators of eutrophication include: First, the basic physiological requirements (salinity tolerances, temperature tolerances, nutrient requirements, minimum light requirements, etc.) of brackish SAV species need to be defined for California. Second, the environmental triggers to seasonal cycles of biomass (temp, salinity, day length, etc.) of both meadow and canopy forming SAV need to be elucidated. Third, nutrient dose-response relationships need to be determined with emphasis on how the response is manifested (e.g., epiphyte loads, light reduction, self-shading from canopy development, etc.).

Sound and Practical Measurement

A suite of indicators are generally used to assess seagrass health and effects from stressors. For the purposes of this review, these indicators can be grouped into three categories:

1) Indicators of seagrass and SAV community structure (taxonomy, biomass, aerial distribution, density)
2) Factors that affect seagrass health through reduced light availability to the plant (e.g., water column light attenuation, total suspended solids, phytoplankton biomass, epiphyte load, macroalgal biomass or cover)
3) Other indicators (environmental or water quality)

Of these, three groups, Sutula (2011) conclude that those indicators that affect seagrass through reduced light availability are candidates for the California Estuarine NNE framework. Indicators of seagrass community structure (taxonomy, biomass and aerial distribution) are important collateral data to track overall trends in the condition of this key habitat type, but do not uniquely respond to stressors.

The primary mechanism of seagrass loss from eutrophication is through the reduction in available light to plant leaves caused by shading or smothering from algal competitors. Reduced light coupled with increased delivery of labile organic detritus (senescent algae and seagrass) to the sediments can lead to additional biogeochemical stressors (hypoxia/anoxia, sulfide toxicity, etc.) that further exacerbate the problem. Seagrass and SAV beds adversely affected by eutrophication tend to be sparse or patchy, heavily epiphytized with microalgae, and/or shaded with phytoplankton or macroalgal blooms. Thus epiphyte load, water column light attenuation (from attendant phytoplankton biomass and turbidity), and macroalgal biomass are indicators of eutrophication that directly affect light availability to seagrass. Canopy forming SAV are even more complicated because in addition to all of the other factors that attenuates irradiance the canopy formers also self-shade. That is, by having most of their biomass at the surface of the water, the plant absorbs and attenuates light before it can reach the deeper leaves.

Water column light penetration is a dominant factor controlling the growth and distribution of seagrass and SAV. Although, water column light attenuation cannot be directly related to nutrient loading, monitoring of underwater light is likely to be a critical component of evaluating eutrophication because all of the algal groups that respond to nutrients influence the underwater light field. Water column turbidity is generally not related to nutrient loading except in circumstances such as river dominated portions of estuaries and is likely to not be a useful indicator of eutrophication, although it does contribute to water column light attenuation. Water column chlorophyll a (chl a), which is a surrogate measure for phytoplankton, responds to nutrient loading and influences underwater light availability for seagrasses and SAV. Consequently, monitoring of chl a may be a strong indicator of eutrophication under some conditions. Table 5.4.3 provides an overview of available methods to measure reduced light availability to seagrass. Overall, sound and practical methods exist to characterize light attenuation to seagrass beds (see Sutula (2011) for detailed review).

Table 5.4.3. Summary of literature reviews for candidate SAV and seagrass related indicators for E-NNE. Excerpted from Sutula (2011).

Group	Indicator	Methods	Information	Summary of Review
Light Attenuation	Epiphyte Cover or Load	Visual rapid assessment, empirical sampling	Relative abundance of competing primary producers	There is a relationship between nutrient loading and epiphyte biomass but epiphyte load is confounded with a variety of other parameters and is unlikely to have a good "signal to noise ratio". As a result it is not likely to be a good stand-alone tool for detecting eutrophication. A field assessment may work in conjunction with additional metrics. **Recommend to pursue in conjunction with other metrics associated with light attenuation.**
	Light Attenuation	Grab samples or Continuously deployed Instrumentation (e.g., data sondes)	Attenuation of light reaching seagrass or SAV bed	Science exists, but assessment framework needs to be refined for California use. Due to species specific requirements and location specific characteristics application of this metric will require additional research and validation. **Recommend to pursue in conjunction with other metrics associated with light attenuation (epiphyte load, chlorophyll a, turbidity), possibly as a rapid assessment to determine whether additional intensive diagnosis is warranted.**
	Chlorophyll a, Turbidity or TSS		Surface water Chl a biomass and turbidity or TSS	Science exists, but assessment framework needs to be refined for use in California. **Recommend to pursue in conjunction with other metrics associated with light attenuation (epiphyte load, chlorophyll a, turbidity, macroalgal cover/biomass), possibly as a rapid assessment to determine whether additional intensive diagnosis is warranted.**
	Macroalgae	Field survey	Biomass or cover	Science exists but data is required to develop an assessment framework for California estuaries (see Sutula (2011). **Recommend to pursue in conjunction with other metrics associated with light attenuation.**

*parameters may include enzyme assays, photosynthetic characteristics, carbohydrate content, etc.

Acceptable Signal to Noise Ratio

Sutula (2011) reviewed indicators of eutrophication that affect seagrass habitat. The primary indicators, epiphyte load, macroalgal cover/biomass, and light attenuation (monitored through a combination of light attenuation, water column chlorophyll a, and turbidity) are known to have considerable spatial and temporal variability. As noted for phytoplankton, high spatial and temporal variability is characteristic of all biological indicators, these indicators tend to integrate better over time and space than stressors, such as nutrient concentrations. Ultimately, our understanding and the various hypotheses about controls on spatial and temporal variability in seagrass aerial distribution vis-à-vis nutrient loads and other stressors can be refined through predictive models.

Approaches to Setting Numeric Endpoints Based on Rafting Macroalgae, Epiphytes, and Water Column Light Attenuation

In order to develop NNEs that are protective of seagrass and SAV habitat, an assessment framework is needed to integrate the effects of rafting macroalgae, epiphyte load, and water column light attenuation on seagrass health. The fundamental step in developing such a framework is to assess the availability of studies which document the effects of these stressors, either as a single or multiple effects, on seagrass.

Precedent for establishment of numeric endpoints for light attenuation to seagrass beds from phytoplankton biomass and turbidity or water clarity, based on light limitation for photosynthesis of seagrass (e.g., Janicki et al., 2000, Kemp et al., 2009, Brown et al., 2004, Sutula, 2011). Turbidity, total suspended solids (TSS), chlorophyll a, and dissolved organic matter are measured to determine light available in the water column that reaches the seagrass bed (Biber et al., 2008). For example, in the mid-Atlantic, environmental conditions that allow adequate light penetration for SAV survival are TSS less than 15 mg L^{-1} and chlorophyll a less than 15μg L^{-1} (Kemp et al., 2004). Bio-optical models predicting light attenuation under various environmental conditions have been calibrated for the Chesapeake Bay (Gallegos, 2001), Indian River Lagoon in Florida (Gallegos and Kenworthy, 1996), and North River in North Carolina (Biber et al., 2008), Yakina Estuary in Oregon (Brown et al., 2007), and Tampa Bay in Florida (Janicki et al., 2000). Explicit studies are needed to understand the precise light requirements of seagrass in SF Bay. This information can be used to develop a biooptical model that could be used to establish a combination of chlorophyll a thresholds and turbidity to establish levels of light availability required to support a healthy seagrass bed.

A strong relationship exists between epiphyte load and light reduction (Boese et al., 2009). However, epiphyte load is generally not quantified in most seagrass or SAV monitoring programs or is quantified using relative abundance. Epiphyte load and subsequent light reduction are highly variable both spatially and temporally, even at the scale of individual plants. There are differences in epiphyte load between wet and dry seasons, location in the estuary and between younger inner leaves and older outer leaves (Boese et al., 2009). One approach to develop an assessment framework to diagnose eutrophication in seagrass beds may involve quantifying effects of rafting macroalgae or water column light attenuation with categories (high, medium and low) epiphyte loading. Another approach, used by the Chesapeake Bay Program, utilizes light attenuation by epiphyte load, in addition to light attenuation by the water column (k_d) from water column suspended sediment and phytoplankton, in efforts to evaluate SAV light requirements. An epiphyte attenuation coefficient is also calculated (k_e) and used with epiphyte biomass (B_e) to predict the percent light reaching the leaf surface (PLL) as described by Batuik et al. (2000).

$$PLL = \left(e^{-(k_d)(z)}\right)\left(e^{-(k_e)(b_e)}\right) * 100$$

This is based on site-specific empirical measurements made over 30 years throughout Chesapeake Bay. Exporting this concept to other estuaries is problematic and would need to be developed for specific estuaries in California.

Studies documenting effects of rafting macroalgae on seagrass bed density have likewise been published (see Sutula (2011) for full review), but few of these studies are useful specifically for establishing endpoints. As with macroalgae on intertidal flats, studies are likewise needed to document the effects of rafting mat biomass and duration on seagrass bed density in California estuaries. These studies are ongoing in other California estuaries as part of the larger estuarine NNE research program. The outcome of these studies should be evaluated for applicability to SF Bay seagrass habitats.

Little documentation is available on the effects of rafting algae on brackish SAV environments. Conceptual models from applicable habitats could be applied, but few field studies have been conducted to illustrate the effects of rafting mats on dissolved oxygen, other pelagic or benthic food webs. Thus, foundational studies are needed to better understand the effects of macroalgae on brackish SAV.

5.4.6 Summary: NNE Indicators Protective of Seagrass and Brackish SAV Habitat

Seagrass

Overall, rafting macroalgae, epiphyte loads and the portion of water column light attenuation influenced by phytoplankton biomass appear to satisfy the four evaluation criteria to be considered as an NNE indicator for seagrass habitats in SF Bay. In order to pursue the use of these indicators for diagnosing and managing eutrophication, several key data gaps need to be addressed. These include:

- Studies to establish thresholds of macroalgal biomass, cover and duration that adversely affect seagrass habitat
- Studies that establish light requirements for seagrass beds in different regions of SF Bay and assessment of duration of reduced light/photosynthesis that results in adverse effects to the seagrass bed.
- Determination of thresholds of the frequency, duration and magnitude of phytoplankton biomass which would result in adverse effects of phytoplankton.
- Development and validation of site-specific dynamic simulation models that simulate reduced light availability to seagrass beds from nutrient loads and other co-factors. This modeling could be done in concert or separately from SF Bay water quality models discussed.

Brackish SAV

Though brackish water SAV are an important component of the Delta, little documentation exists on the extent and ecology of these primary producer communities in the North Bay. Studies are funded to characterize the structure and stressors associated with *Stuckenia pectinata* (Sago Pondweed) beds in the North Bay (K. Boyer, personal communication. Literature from intermittently tidal Mediterranean estuaries and the Chesapeake Bay suggests that brackish SAV species decline in response to eutrophication. The mechanism of decline is presumably mediated through light limitation caused by epiphytes, phytoplankton or macroalgal blooms, though there is relatively poor documentation of response of SAV with nutrient loads and other co-factors (temperature, salinity, etc.). This literature contradicts anecdotal observations of brackish water SAV in the Bay Delta and in intermittently tidal estuaries, where very dense and apparently health *Ruppia* populations exist under very eutrophic conditions (high nutrient loading, high organic loading to the sediments, fish kills, large diurnal dissolved oxygen swings, etc. No clear documentation exists of dose-response relationship between elevated biomass of SAV and secondary consumers such as neither water column macroinvertebrates, nor documentation of changes in bed extent, biomass or density as a function of nutrient loading. The following key data gaps should be addressed in order to pursue the use of brackish water SAV for diagnosis of eutrophication in the North Bay.

- Document nutrient load- SAV community response through long-term monitoring at established sites
- Document the relationship between SAV community structure, indicators of light availability (epiphyte load, chlorophyll a biomass, macroalgal cover/biomass, etc.), dissolved oxygen, pH, and indicators of aquatic life use (macroinvertebrates, fish, etc.)
- Document the growth habits of these plants and elucidate mechanisms of water column versus sediment response to nutrient loads. Detailed physiological and autecological studies of brackish SAV species need to be undertaken in order to better understand the habitat requirements of these communities.

It should be noted that this indicator will likely be of interest in the Delta. Therefore, opportunities exist for synergy on research to address this data gap with what could be proposed for the Delta.

5.5 Macroinvertebrates

Macrobenthic fauna or macrobenthos are invertebrates living on and within the sediments of aquatic waterbodies. Macrobenthos are one of the primary tools used to assess the ecological condition of estuaries and coastal nearshore habitat because 1) they live in bottom sediments, where many stressors accumulate; 2) most macrobenthos are sedentary and therefore reflect the quality of their immediate environment (Pearson and Rosenberg, 1978; Dauer,1993; Weisberg et al., 1997); 3) most communities are comprised of a diverse array of species with a variety of tolerances to stress, so the presence or absence of different taxa can provide information about the types of stressors present (Christman and Dauer, 2003; Lenihan et al., 2003); and 4) they serve as food sources for many ecologically and economically important estuarine fish and birds (Virnstein, 1979; Phil et al., 1992; Gillett, 2010). Macrobenthic community-based assessment tools have traditionally been designed to assess overall habitat quality, successfully integrating a variety of anthropogenic stressors (e.g., contaminants, eutrophication, or physical disturbance) while accounting for gradients in natural stressors/environmental conditions (e.g., salinity, sediment type, or depth). Within the macrobenthos, there are a variety of aspects that can be used in environmental assessment, including individual responses (e.g., condition indices, cellular bioindicators, or contaminant loads) (Ringwood and Keppler, 1998; Brylawski, 20089), as well as community-level responses (e.g., abundance of sensitive/tolerant taxa, community composition changes) (Weisberg et al., 1997; Borja et al., 2000; Smith et al., 2001; Llansó et al., 2002). The community structure of macrobenthic infauna has been used as an indicator of ecosystem health and environmental stress for a number of years in a variety of estuarine habitats around the United States, including the USEPA Environmental Monitoring and Assessment Program (EMAP), National Coastal Assessment (NCA), Chesapeake Bay Benthic Monitoring Program, Southern California Bight Regional Monitoring Program, California Sediment Quality Objective, and internationally the European Union Water Framework Directive (WFD).

Three types of macrobenthic indicators have been considered for assessing eutrophication: 1) taxonomic composition, 2) abundance, and 3) biomass. Gillett provides a full review of these indicators for the California estuarine NNE framework (see Chapter 7 in Sutula (2011), and that work is incorporated in this review.

5.5.1 Applicable Habitat Types

Macrobenthic infauna are valuable tools for environmental assessment in estuaries because, in the absence of long-term hypoxic conditions (e.g., main-stem Chesapeake Bay), they can be found throughout all soft sediment habitats found in estuaries; from euhaline sandy sediments through tidal freshwater muds or from deep subtidal waters through the littoral zone. These macrobenthic communities, however, are not uniform across these gradients in physical habitat, with relatively unique communities in each salinity and sediment regime (e.g., Sanders, 1958; Holland et al., 1987; Attrill and Rundle, 2002; Ranasinghe et al., 2010). Consequently, assessment tools developed to work across the entire spectrum of estuarine habitats use a categorical approach to ecological condition assessment. Different aspects of community structure and/or different thresholds of community characters for the different salinity zones – typically following the Venice classification scheme (International Association of Limnology, 1958) – and sediment types – typically either sands or muds – found in an estuary (e.g., Weisberg et al., 1997; Van Dolah et al., 1999; Llansó et al., 2002). The lower salinity (<5 psu) portions of estuaries are notoriously difficult systems for the application of macrobenthic community changes in assessing habitat quality due to the salinity fluctuations and high turbidity, which act as stressors to the community, as well as the pervasive human perturbations typically found there throughout the United States and Europe (Draheim, 1998; Alden et al., 2002; Attrill, 2002; Diaz et al., 2004). Estuarine habitats that encompass a large amount of intertidal area additionally problematic, because at low tide the air exposure can create desiccation and large fluctuations in temperature that can impact community diversity, abundance, and biomass in comparison to adjacent subtidal habitats (Van Dolah et al., 2000; Holland et al., 2004).

5.5.2 Availability of Data on Macrobenthos in San Francisco Bay

Surprisingly, there are no recent summaries of the soft-bottom benthic community of the entire Bay and Schaeffer et al (2007) bemoan the challenge of keeping such a summary up to date given the high rate of non-native species introductions into the system (Schaeffer et al., 2007). The only comprehensive summary of any compartment of the soft bottom benthos was written in 1986

(Nichols and Pamatmat, 1988), just prior to the invasion of *Corbula amurensis*, a bivalve that greatly altered the soft-bottom benthos community structure (Alpine and Cloern, 1992; Schaeffer et al., 2007).

The data used to describe macrofauna in this system are limited to the post-Corbula invasion period due to the dominance of the bivalve *Corbula* in the communities where it resides. The major data sources and their abbreviations include: 1) long-term monitoring data collected monthly from the freshwater Delta to the Richmond San Rafael Bridge by the California Department of Water Resources (DWR); 2) 2 to 3 year bimonthly data collected by the Regional Effects Monitoring Program in 1986 to 1989 (REM); 3) long-term near-monthly data collected by the USGS south of Dumbarton Bridge in Palo Alto (USGSPA); 4) the summary of semi-annual data collected by various agencies as listed in Thompson et al. 2000 (Regional Monitoring Program [RMP], Long-term monitoring program [LMP], and Bay Protection and Toxic Clean-Up Program [BPTCP]); 5) samples taken as part of NOAA’s National Coastal Assessment and the USEPA Environmental Monitoring and Assessment Program (EMAP) West Coast pilot (2000-2001); 6) a monthly study of the bivalves south of San Mateo Bridge collected by the USGS in 1990 to 1996 (Thompson, 2005, 1999); and 7) unpublished rapid assessment survey data from the California Academy of Sciences (C. Brown, Smithsonian Institute, pers. comm.; Schaeffer et al., 2007).

5.5.3 Indicator Relationship to Nutrients and Water Quality

Excessive amounts of nutrients that lead to excessive amounts of primary production and eutrophication typically do not have direct impacts on macrobenthic fauna, with the exception of HABs (e.g., Anderson et al., 2002). Eutrophication primarily affects the macrobenthos via two basic microbially-mediated, indirect paths: 1) water column hypoxia/anoxia or 2) the accumulation of toxic reduced sulfides and ammonium in the sediment.

As heterotrophic microbes consume the organic matter from the primary producers, oxygen is removed by aerobic microbes and reduced compounds are created as metabolic byproducts. Even in natural, non-eutrophic conditions, these processes occur in both muddy and sandy sediment environments and the fauna that live there are adapted to deal with low-oxygen, reducing environments. As the amounts of organic matter produced and accumulated in the system, the low oxygen and reduced conditions begin to either smother or poison the benthic fauna. These processes lead to progressive changes in the abundance, biomass, and composition of the macrobenthic community and eventually lead to azoic conditions. By looking at trajectories and magnitudes of these changes in community, one should be able to distinguish between the effects of eutrophication, as well as changes brought about by other common estuarine stressors (e.g., contaminants, physical disturbance, or salinity fluctuation).

Effects of Hypoxia

Most of the information detailing the response of macrobenthic fauna to eutrophication is related to the effects of low-oxygen (i.e., hypoxia or anoxia) on macrobenthic communities. Benthic sediments in estuaries are naturally low-oxygen environments because of the large amounts of organic matter and large number of heterotrophic microbes there. As a consequence, most benthic fauna have evolved to deal with those conditions (Pearson and Rosenberg, 1978; Hargrave et al., 2008), but hypoxic or anoxic conditions in the overlying water can be an important factor structuring the composition of an ecosystem (e.g., Rosenberg et al., 1991; Diaz and Rosenberg ,1995; Baustin and Rabalais, 2009; Seitz et al., 2009).

The response of the macrobenthic community to hypoxic conditions is primarily negative. Increases in frequency and duration of hypoxic (<2.0 mg O_2 L^{-1}) or anoxic (<0.5mg O_2 L^{-1}) conditions lead to reduced community diversity, biomass, and productivity and eventually complete absence of macrofauna (Gray et al. 2002; Rakocinski 2009; Seitz et al. 2009). The degree of the response in these broad, community attributes and the trajectory of community changes will vary, depending upon the severity and duration of hypoxic conditions. Tolerance to low oxygen conditions varies widely among the taxonomically diverse macrobenthic community, though persistent anoxic conditions will eventually kill all metazoans (e.g., main stem Chesapeake Bay, Gulf of Mexico, coast of Oregon) (Holland et al., 1977; Diaz and Rosenberg, 2008; Rabalais et al., 2010). Among the most common types of estuarine macrofauna, crustaceans and gastropods are typically the most sensitive (LT_{50} anoxia <1 d), annelids the most tolerant (LT_{50} >5 d), and bivalve mollusks in-between, as different species have differing capabilities of sealing themselves off to the environment and waiting for better conditions (Llansó, 1992; Sagasti et al., 2001; Gray et al., 2002; Calle-Delgado, 2007).

Water column hypoxia can also have indirect effects on macrobenthic survival and community structure by altering behavior that increases the risk of being preyed upon. As oxygen concentrations near the bottom decline, many species of infauna will start to move closer the sediment surface in an effort to extend appendages or siphons further up into the water column in search of oxygenated water (Rosenberg et al., 1991; Llansó, 1992; Long et al., 2008). Eventually, continued exposure to low oxygen forces many infaunal species from the sediment entirely and they remain moribund on the sediment surface, which greatly increases their exposure to predation by benthivoric nekton (Nestlerode and Diaz 1992; Pihl et al. 1992; Seitz et al. 2003; Powers et al. 2005).

Effects of Increased Sediment Organic Matter Accumulation

Eutrophic conditions do not always lead to hypoxia and anoxia, but can still have effects on the macrobenthic community of estuaries. Hypoxic conditions are, in part, a function of water column stratification and water residence time (Diaz and Rosenberg, 1995; Hagy et al., 2004; Kemp et al., 2009) and many of California's estuaries that are always connected to the open ocean are not always prone to the formation of chronic hypoxic bottom waters. As such, understanding the effects of non-hypoxic eutrophication on the macrobenthos will be particularly relevant to California's estuaries.

Almost every modern work on the effects of eutrophication and the accumulation of organic matter on benthic fauna is based upon the conceptual model of Pearson and Rosenberg (1978). This paper summarizes one of the central tenets of benthic ecology: that there are relatively consistent and predictable changes in macrobenthic community structure with increasing accumulation of organic matter in marine sediments (Figure 5.5.1). In short, the model proposes that: 1) under normal, non-eutrophic conditions, a benthic community should be composed of a trophically and functionally diverse array of species that span different body sizes and lifespans, as well as live at various depths through the sediment, often extending 10's of cm below the sediment-water interface[5]; 2) as organic matter begins to accumulate in the sediment and there will be changes in the community, shifting towards a less diverse community composed of smaller fauna with relatively short lifespans living near the sediment surface; and 3) eventually the sediments are devoid of macrofauna and are covered in mats of sulfur-oxidizing bacteria (i.e., *Beggiatoa*). The presence of benthic infauna will typically enhance the depth of oxygen penetration due to tube building/ventilating and bioturbation. As a system becomes more eutrophic and organic matter begins to accumulate at greater rates in the sediment, bacterial production is stimulated and the demand for oxygen outstrips the rates of diffusion. This leads to anoxic, reducing processes dominating formally oxygenated sediment, and a variety of bacterial metabolic pathways that produce byproducts (primarily sulfide and ammonium in saline sediments) that are toxic to most metazoans (Pearson and Rosenberg, 1978; Jørgensen, 1996; Gray et al., 2002; Hargrave et al., 2008). These compounds and the reducing environment of the sediments are thought to be the mechanism behind the mortality leading to changes in community structure. Many of the species that are community dominants in disturbed habitats are always present at low densities and presumably at a competitive disadvantage to non-disturbed community dominants. Only when the non-disturbed dominants die off are there available resources that allow tolerant fauna to flourish (e.g., Gillett et al., 2007).

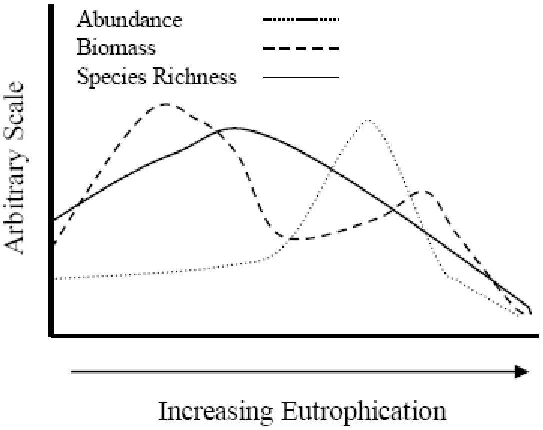


Figure 5.5.1. Conceptual patterns of abundance, biomass and species richness with increasing eutrophication. After Pearson and Rosenberg, 1978.

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[5] In practice, this kind of community should only be expected in relatively high salinity environments (>10-15 psu) with relatively little salinity fluctuation. The premise of community change is still appropriate in lower, more variable salinity environments, but the baseline community will likely be less trophically diverse and more tolerant of environmental stressors than higher salinity communities, as detailed in our subsequent discussion of the so called "estuarine quality paradox".

Most of the historical literature (summarized in Nixon, 1995; Grall and Chauvaud, 2002; Gray et al., 2002; Hyland et al., 2005) focuses on the negative impacts of eutrophication in general and macrobenthic fauna in specific. There has been recent interest, however, in how eutrophication can have both positive and negative effects on the functioning of macrobenthic communities in estuarine ecosystems (e.g., Beukema and Cadée, 1997; Nixon and Buckley, 2002; Rakocinski and Zapf, 2005; Gillett, 2010). By definition, eutrophication typically leads to an increase in the primary production of a system and this represents an increase in food availability for primary consumers, which has been linked to increases in benthic production, as well as fisheries yields (Nixon and Buckley, 2002; Breitburg et al., 2009; Nixon, 2009). Rakocinski and Zapf (2005) put forth a conceptual model of changes in macrobenthic function with increasing eutrophication that incorporates both the positive and negative aspects of eutrophication on benthic communities (Figure5.5.2). In this model, as a system begins to become eutrophic, there is an increase in the rate of macrobenthic function. This increase is related to increases in primary production, which provide a release from food limitation for existing fauna (e.g., Marsh and Tenore, 1990; Sterner et al., 2002; Brylawski, 2008), as well as beginning to alter the sediment biogeochemistry, allowing for the eutrophication-tolerant taxa to increase their proportion within the community before the sensitive taxa are severely impacted.

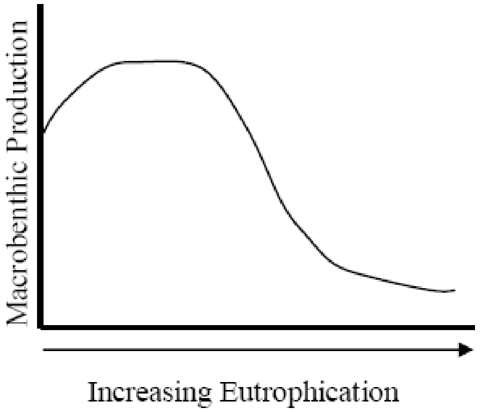

Figure 5.5.2. Conceptual relationship between macrobenthic production and eutrophication. After Gillett, 2010.

As the degree of eutrophication progresses, the model predicts that there will be decline in community function due to the negative aspects of organic matter accumulation (reduced O_2 penetration and the build-up of toxic reduced compounds) outweigh the benefits of additional food and the composition of the macrobenthic community changes, following the classic pattern of habitat degradation described in the preceding paragraphs. Gillett (2010) showed an initial increase in secondary production with increasing eutrophication, followed by a decline in production as eutrophic conditions continued to intensify. There are two aspects of this eutrophication pattern that bear further detail: 1) macrobenthic community composition is important – the positive aspects of eutrophication (i.e., the fertilization effect) most strongly affected filter- and interface-feeding fauna that could directly utilize the increases in phytoplankton production in the water column (Gillett, 2010), whereas other types of benthic fauna remained unchanged until the negative aspects became dominant; and 2) the benthic-pelagic setting is important – the macrobenthos from sandy, non-depositional habitats appear to have a greater buffer to eutrophication and the macrobenthic community experience negative impacts slower than their counterparts from depositional habitats, where the sediments are naturally rich with organic matter. These depositional habitats can be quickly oversaturated with organic matter and therefore habitat quality will start to degrade with only a small increase in eutrophication (Molinaroli et al., 2009).

The concepts of eutrophication having positive benefits to the macrobenthic community, while still representing a change in ecosystem condition from reference, has been incorporated into a small number of environmental monitoring programs that utilize the macrobenthos as their assessment tool (Chesapeake Bay Program – Weisberg et al., 1997; Mid-Atlantic US – Llansó et al., 2002; European Water Framework Directive – Lavesque et al., 2009). In these indices, which are largely built upon the Pearson and Rosenberg (1978) paradigm, macrobenthic abundance and biomass do not have a simple, positive linear relationship with habitat quality. Instead, they have a concave, unimodal relationship to habitat quality, where a sample can be assessed as degraded for having too much or too little biomass/abundance (Weisberg et al., 1997). It should be noted, however, that these indices were developed to assess overall habitat quality or integrity, not individual stressors on the macrobenthic community. However, there has been some work in recent years to use specific aspects of the macrobenthic community to assess multiple stressors impacting ecosystem quality. Christman and Dauer (2003) and Dauer et al. (2000) were able to detect the differential response of the macrobenthic community in Chesapeake Bay to low oxygen stress and chemical contaminant stressors by looking at variation benthic multi-metric index (Chesapeake Bay B-IBI [Weisberg et al., 1997]) scores in relation to environmental conditions. Furthermore, Dauer et al. (2000) were able to relate index score to different types of watershed development (urban, agricultural, and forested) and local water/sediment quality. Lenihan et al. (2003) were able to differentiate macrobenthic community responses to either organic matter enrichment or heavy metal contamination. In this study, they showed positive responses among annelids (i.e., increases in abundance and biomass) with organic matter enrichment, even when combined with increasing concentrations of heavy metals. Conversely, echinoderms had slightly positive responses to organic enrichment, but declined when exposed to heavy metals and arthropods declined with increased exposure to both types of stressor (Lenihan et al., 2003). This type of differential response by separate components of the macrobenthic community to different stressors could be used to delineate eutrophic impacts from the mix of co-occurring stressors typically found in estuarine ecosystems.

5.5.4 Species composition in the San Francisco Bay

Populations of aquatic organisms in the upper portions of SF Bay have undergone significant declines over the past several decades (Jassby et al., 1995; Carlton, 1979). The benthic macroinvertebrate community of the SF Bay is composed of less than 40 species, most of which were introduced in the 19th century when oysters were imported from the eastern coast of the United States and grown in the Bay, and from a lack of regulation of ballast water (Carlton, 1979; Nichols et al., 1986; Nichols and Pamatmat, 1988; Thompson et al., 1999; Cohen and Carlton, 1998). The results of a 10-year study of the invertebrates living on a mudflat at the south end of the Bay (Nichols and Thompson, 1984) show that species composition and relative abundance have remained fairly constant, at least in the second half of the 20th century (Nichols and Pamatmat, 1988). Studies are often separated geographically along a salinity gradient (Figure 5.5.3), as well as by composition of the substrate of the Bay floor (Figure 5.5.4).

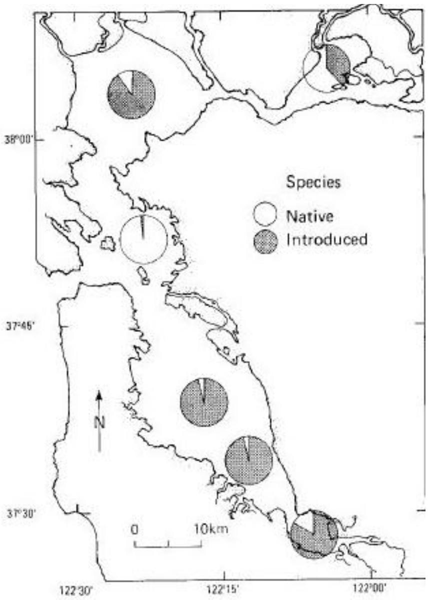

Figure 5.5.3. Proportions of introduced and native species relative to biomass of mollusks in San Francisco Bay (Nichols and Pamatmat, 1988).

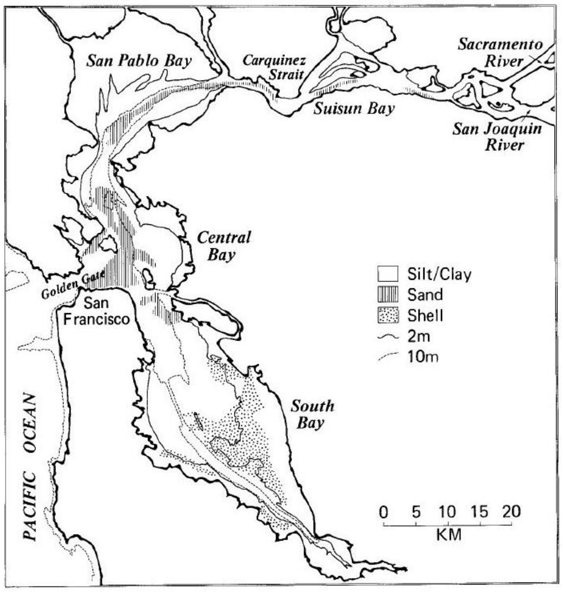

Figure 5.5.4. Generalized distribution of surface sediment composition in San Francisco Bay (Nichols and Pamatmat, 1988).

5.5.5 Factors Effecting Temporal and Spatial Variation of Indicator

The SF Bay experiences natural fluctuations in tide, salinity, nutrient and sediment loads, sediment deposition, water clarity and temperature due to variations in freshwater flows, semidiurnal tides, and seasonal winds as well as historic and recurring anthropogenic influences including nutrient and organic enrichment, and contamination. It follows that the benthic community composition in the SF Estuary responds to these many types of physical, chemical, and biological fluctuations, both spatially and temporally (Nichols, 1985; Nichols and Pamatmat, 1988; Thompson et al., 1999).

Much of the spatial distribution of benthic organisms can be tracked along a salinity gradient between the marine environments of the Central and South Bay, to the northern areas of the San Pablo and Suisun Bays which are characterized by temporally shifting brackish and freshwater landward and seaward boundaries of the estuary (Nichols and Pamatmat, 1988; Jassby et al., 1995). In terms of temporal variability, in the temperate climate of SF Bay, food availability (associated with phytoplankton and microphytobenthos blooms) may be the most important factor in the reproductive cycle and timing of macroinvertebrates (Nichols and Pamatmat, 1988), but as mentioned earlier, predation by shallow feeding birds can also be important some years. Observed seasonal changes in abundance of benthic macrofauna have also been linked to seasonally predictable variations in freshwater inflow, winds and tides leading to water column mixing, microalgal biomass, and sediment erosion and deposition patterns (Nichols and Thompson, 1985; Nichols and Pamatmat, 1988).

Spatial Variability

The SF Bay, except the central portion nearest the Golden Gate, is very shallow with characteristically wide intertidal and shallow subtidal mudflats incised by narrow mid-Bay channels (Thompson and Nichols, 1985; Lucas et al., 2009). However, some conceptualize the Bay as having two estuarine systems, each with a different hydrodynamic and freshwater inflow regime. Historically, North and South Bays have had very different macrobenthic invertebrate communities, whose distribution is most strongly influenced by spatial variations in salinity and composition and stability of sediments (Nichols, 1979, Nichols, 1985; Nichols and Pamatmat, 1988). Suspension-feeding bivalves were found in the 1970s to be less abundant in the northern Bay, potentially because of high-suspended sediment loads and large seasonal variations in salinity (Cloern, 1982). Later, beginning about 1986, an invasive clam Corbula amurensis recruited and is now thought to be one of the dominant controls on algal productivity in the northern reaches of the Bay (Jassby, 2008). The South Bay is a “lagoonal system” (salinity 26-30 PSU) whose major inputs of fresh water and nutrients are more dominated by treated wastewater releases (Nichols et al., 1986; Caffrey, 1995). In the North Bay, salinity ranges from 0- 30 PSU. Although there are some nutrient inputs in treated wastewater (for example the Sacramento Regional Wastewater Treatment Facility) that can be important during lower flow summer and autumn months, the majority of nutrients loads and fresh water are delivered to the North Bay during the winter from the Sacramento and San Joaquin rivers, which converge in the Delta (Caffrey, 1995).

Suisun Bay is made up of brackish water embayments and is inhabited by less than 10 permanent macrobenthic species, and because the region is inundated each winter by freshwater it is considered a mesohaline community (Nichols and Pamatmat, 1988; Schaeffer et al, 2007). Species that survive here include mollusks *Corbicula amurensis*, which dominates, with *Macoma balthica, Mya arenaria*, and *Corbicula fluminea* - though only when the river inflow is particularly high, lowering salinity levels (Nichols and Patatmat, 1988; Schaeffer et al., 2007); the amphipods *Corophium stimsoni*, and *C. spinicorne*; and the annelids *Nereis succinea* and *Limnodrilus hoffmeisteri*. During periods of low river flow, which leads to increased salinity, the populations of some fauna like the polychaete *Streblospio benedicti* and the amphipod *Ampelisca abdita* are shown to expand upstream towards Suisun Bay. Normally however, these two species can only be found west of the Carquinez strait because of their intolerance to freshwater (Nichols and Patatmat, 1988). Suspension-feeding bivalves are less abundant in the northern San Francisco Bay estuary. Cloern (1982) suggested that this is due to high suspended sediment loads and large seasonal variations in salinity in this area.

West of the Carquinez straight, where salinity rarely dips below 5 psu, the macrobenthic community diversity increases. The macrobenthic community of the broad shallow subtidal expanses of the San Pablo Bay (Figure 5.5.5) includes, in addition to the mollusks found further upstream: *Gemma gemma*, *Musculista senhousia*, *Tapes philippinarum* and *Ilyanassa obsolete*; amphipods *Ampelisca abdita*, *Grandideirella japonica,* and *Corophium spp*. The polychaetes *Steblospio bendicti, Hetermastus filiformis, Glycinde* spp., *Polydora* spp., and several other species of oligochaetes (Thompson and Nichols 1985; Nichols, 1988).

The Central Bay is characterized by stronger currents, deeper waters and a more marine environment. The strong tides create a highly dynamic bottom of large sand waves that reverse directions with each tide. The benthic community is dominated by species that are found in sand sediments along the outer coast, demonstrating a more marine influence. Islands and other rock outcrops in the Bay are inhabited by hard-substrate marine organisms, as well as the "cosmopolitan Bay mussel *Mytilus edulis*" (Thompson and Nichols, 1985; Nichols and Pamatmat, 1988).

In the South SF Bay, several of the same species found in San Pablo Bay occur, but in the subtidal mud areas, the large tube-dwelling polychaetes *Asychis elongata* is common in shallow and deep water (Figure 5.5.5). These maidanid polychaetes have burrows that can reach up to a meter deep in sediments, and can occur in very dense patches in South Bay and can influence structure of the entire community (Nichols, 1988; Thompson and Nichols, 1985; Nichols and Pamatmat, 1988; Schaeffer et al., 2007). In the intertidal and shallow subtidal reaches of South Bay, *Gemma gemma*, *Ampelisca abdita* (Figure 5.5.5), and *Streblospio benedicti* tend to dominate (Thompson and Nichols 1985; Nichols and Pamatmat, 1988; Schaeffer et al., 2007). Many of the species in this assemblage are patchy with some, like *A. abdita*, showing very high abundance one year and low abundance the next year (Schaeffer et al., 2007). This is also where introduced macrofauna tend to be most abundant (Nichols et al., 1990).

Thompson and Lowe (2004) conducted assessments of "benthic condition" between 1994 and 1997 using a multimetric Index of Biotic Integrity (IBI) on two major benthic assemblages in the Estuary: the polyhaline assemblage in the Central Bay and the mesohaline assemblage from the moderate salinity portions of the Estuary (Thompson and Lowe, 2004). The same assessment methods were subsequently applied to samples from San Pablo Bay, Napa and Petaluma rivers, and three sites in the Napa-Sonoma Marsh in 2000–2001. They found that elevated TOC and sediment contamination in those areas had more influence on benthic species composition and abundances than did changes in the hydrodynamic regime (e.g., river or marsh channel), or seasonal and tidal differences in salinity, flow, turbidity, or temperature which supports the idea that anthropogenic nutrient enrichment impacts outweigh environmental co-factors (Thompson and Lowe, 2004; Thompson, et al., 2007).

While Nichols and Pamatmat (1988) largely focused on the ecology of the soft bottom benthos, Schaeffer et al. (2007) also discuss the differences along salinity gradients in areas of the Bay with hard bottom surfaces. In the mesohaline regions hard bottoms surfaces include the large filter-feeding mussel *Mytilus trossulus/gallopgrovincialis* and filter-feeding barnacle *Amphibalanus improvisus* and other attached, non-mobile species such as anemone, sponges, and several suspension feeder species. In polyhaline regions, invertebrates are much more diverse than that seen in mesohaline regions (Schaeffer et al., 2007). In euhaline regions, with higher species richness, amphipods are still a major component of the community, as are omnivorous and carnivorous polychaetes, including two species of scale worms. Pacific rock crab (*Cancer antennarius*) and the red rock crab (*C. productus*) inhabit rocky, intertidal and subtidal areas in the Pacific Ocean, and likely use SF Bay as an extension of their coastal habitats (Schaeffer et al., 2007).

Temporal Variability

Estuarine invertebrates are relatively short lived, however the highest densities are normally observed between spring and autumn, peaking in the summer months, reflecting the high reproductive capability and productivity rates of many benthic species. Abundance decline in winter usually indicates species die off after reproducing and lack of endurance for winter conditions, including lack of food availability in shallow waters. However, because of the temperate climate in the SF Bay some species, such as *Ampelisca abdita, Gemma gemm*a, and *Streblospio benedicti*, can reproduce year round (Nichols and Thompson, 1985).

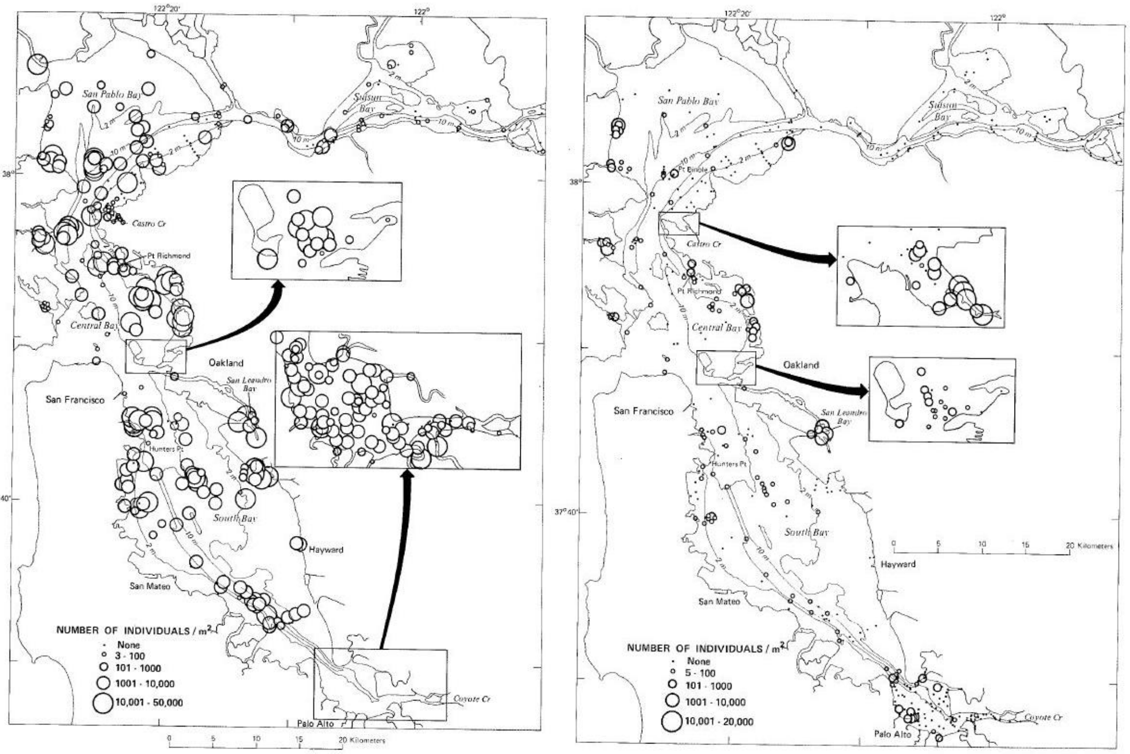

Figure 5.5.5. Distribution and abundance of *Ampelisca abdita* (left) and a *Capitella* polychaete (worm) (right) - an opportunistic species (Nichols, 1988).

Additionally, there is evidence that species interactions can contribute greatly to interannual variations. For example, Nichols and Thompson (1985) observed that Macoma balthica establishes large populations only when the amphipod *A. abdita* is not abundant. Similarly, abundance patterns for the tube-dwelling, surface-deposit feeding polychaete *M. viridis* (peak abundance late spring-early summer, followed by a several month minimum) varies alternately with *Monocorophium alienense*, an amphipod with similar habits and food but which peaks in abundance in late fall/early winter. While Nichols and

Thompsons long term studies have shown that species composition does not change considerably over long periods of times, year-to-year predictability of species abundances is low (Nichols and Thompson, 1985).

Several separate studies of growth of SF Bay benthic invertebrates have shown a strong coincidence in the timing of most rapid growth. Maximum growth in Macoma balthica on a South SF Bay mudflat occurred at the same time as the spring phytoplankton bloom and coincided with the peak in benthic microalgal biomass (Nichols and Thompson, 1985). There is also a strong positive relationship between California bay shrimp annual abundance and freshwater outflow in spring (Schaeffer et al., 2007).

The benthic community in the northern part of the Bay is dominated by *C. amurensis*, which peaks in abundance in fall during the wet and normal water years, and peaks in summer and fall in dry and below normal years. Lowest abundance for this bivalve is in spring or early summer in most years (Schaeffer et al., 2007). The species in the South Bay such as the bivalves *C. amurensis, Mya arenaria, Venerupis japonica, Macoma petalum*, and *Musculista senhousia* show very strong seasonal patterns with decline in abundances to near zero each winter/early spring. The bivalves are therefore mostly annual species in this habitat with peaks in abundance occurring in late spring/early summer. The amphipods (*Corophium heteroceratum* and *Ampelisca abdita*) can show similar annual patterns except during dry years when A. abdita in particular seems to persist through the winter (Schaeffer et al., 2007).

5.5.6 Indicator Trends

Despite the long term scientific and growing management interests in benthic responses to nutrient enhancement, there is considerable debate about the actual impacts of these factors or their potential mechanisms of effect (Posey, 2006). Results from multi-year investigations in the SF Bay estuary show that large inter and intra-annual abundance fluctuations within benthic macroinvertebrate populations, depending on variable predation on adults and on planktonic larvae, variation in the number of larvae in the water column, river inflow from the Sacramento San Joaquin system increasing or decreasing salinity, and changes in the phytoplankton community (Nichols and Pamatmat, 1988; Nichols and Thompson, 1985). After a 10-year study, Thompson et al. (2002) found that the Bay benthic community was composed of non-indigenous, opportunistic species that dominated the community due to their ability to survive the many physical disturbances on the mudflat (Thompson et al., 2002).

Analyses of the benthic community structure over a 28-year period show that changes in the community have occurred concurrent with reduced concentrations of metals in the sediment and in the tissues of the clam *Macoma balthica* (Thompson et al., 2002; Shouse et al., 2003). In addition, two of the opportunistic species (*Ampelisca abdita* and *Streblospio benedicti*) that brood their young and live on the surface of the sediment in tubes have declined in dominance coincident with the decline in metals. *Heteromastus filiformis*, a subsurface polychaete worm that lives in the sediment, consumes sediment and organic particles residing in the sediment has shown an increase in dominance. These changes in species dominance reflect a change in the community from one dominated by surface dwelling, brooding species to one with species with varying life history characteristics, though these trends are directly related to decreases in trace metals, not nutrients (Shouse, 2002).

Multi-year studies reveal that year-to-year variations in species abundances are much larger than would be expected from repetitive cycling of environmental conditions, and have more to do with larger scale shifts in weather patterns, disturbances due to storms and droughts, and anthropogenic influences (Nichols and Thompson, 1985; Cloern et al., 2010). For example, as the PDO has changed signs since 1999, we have seen an uptick in populations of shrimp and crab in the SF Bay. The mean annual catch per unit effort (CPUE) between 1980-2008 for Cancer crabs and caridean shrimp was calculated and reflected 3–6 fold increases. Population increases of these predators after 1999 were followed by population declines of bivalve suspension feeders and increasing phytoplankton biomass that persisted from 2000 through 2008. A strong argument has been put forward that these observations may indicate a dramatic restructuring of biological communities in SF Bay in relation to larger climate driven phenomenon at the decadal scale (Cloern et al., 2010).

5.5.7 Utility of Indicator as an Eutrophication Indicator for San Francisco Bay

Relationships to Beneficial Uses

Macrobenthos play a critical role in the biotic and abiotic functioning of the estuary; thus a diverse, fully functional macrobenthic community is an essential part of maintaining ecosystem services and related estuarine beneficial uses. The State of California has designated six "Estuarine Beneficial Uses" upon which to evaluate the estuarine natural resources (structure) and ecosystem services (function) (Chapter 2). These beneficial uses broadly address biodiversity and threatened/endangered species (rare [RARE], spawning [SPWN], and migratory [MIGR] uses), commercially valuable resources (commercial [COMM], shellfish [SHELL], and aquaculture [AQUA] uses), and the inherent value estuarine habitat for aquatic life (estuarine [EST] and wildlife [WILD] uses). The structure and function of the macrobenthic community encompass: 1) their contribution to estuarine and marine biodiversity; 2.) direct recreational and fisheries harvest; 3) a food resource for a variety of estuarine aquatic life forms, including fish, birds, marine mammals; 4) a critical role in the maintenance of water column and sediment biogeochemical cycling; and 5) the consumption of a variety of organic matter sources and subsequent regeneration of nutrients to the water column.

From the estuarine beneficial use perspective, macrobenthos are part of diversity of aquatic life and as such a direct measure of EST beneficial uses. The State of California has recognized the intrinsic value of macrobenthos and as such, is currently developing a biocriteria program that includes macrobenthos as a primary indicator of aquatic life in streams (J. Bishop, SWRCB, Pers. Comm.). Development of macrobenthic-based assessment tools for California's estuaries will provide the State the same opportunity to establish biocriteria in estuaries.

In terms of commercial value, many species of macrobenthos are directly harvested (e.g., oysters, mussels, clams, shrimp, and lobsters) by humans, which would be classified as COMM and SHELL uses. Within California, commercial shellfish harvest represented approximately $100 million in fisheries landings in 2008 (NMFS pers. comm.), in addition to the creation of jobs and revenue related to harbor infrastructure, seafood processing and distribution, and tourism. The harvest of macrobenthos also provides recreational value. Beyond their direct commercial value, the macrobenthos provide an important source of food for estuarine and marine fish, birds and marine mammals (EST), including migratory fish and marine mammals (MIGR), spawning fish (SPAWN), and threatened/endangered species of fish and birds (RARE). Numerous commercially important nekton (e.g., *Embiotaca jacksoni*, *Umbrina rancador,* or *Hypsopetta guttulata*) from California's estuaries are dependent upon the macrobenthos as a food source and thus provide indirect support for COMM beneficial uses (Allen et al., 2006).

The macrobenthos play a key role in sediment nutrient and contaminant cycling through bioturbation and bioirrigation (the mixing of sediment and advective exchange of sediment pore waters with surface waters) and thus are a key component of maintenance of good estuarine and marine habitat and water quality (EST and MAR). Active burrowing and the building of tubes or galleries in the sediment increases the penetration of oxygen into the sediment and the surface area of oxic/anoxic sediment horizons, which can enhance coupled nitrification/denitrification and ultimately remove nitrogen from the estuary (Aller, 1982; Mayer et al., 1995; Aller and Aller, 1998). As infauna ventilate their burrows and tubes, there is an increase in the flux of pore water through the sediment and the exchange of porewater with overlying waters, which will carry dissolved nutrients and organic matter with it (Michaud et al., 2005, 2006). In estuarine systems where the mixed layer extends to the bottom, filter-feeding benthos will enhance benthic-pelagic coupling by collecting water column production and depositing waste products at or below the sediment surface (Graf, 1992; Gerritsen et al., 1994; Thompson and Schaffner, 2001). Macrobenthos have been shown to be a major control in both the North and South Bay (accepting light limitation and temperature) on phytoplankton populations. Analogously, head-down deposit-feeders (e.g. *Asychis elongata*) feed on bacteria and organic matter centimeters below the sediment surface and depositing waste at the surface, which exposes and recycles organic matter back to the water column (Lopez and Levinton, 1987; Clough and Lopez, 1993; Levin et al., 1997). Autumn (pico) phytoplankton blooms including HABs occur once the peak biomass of water column filter feeders passes in the summer and regenerated nutrients from decaying detritus are able to feed the planktonic foodweb in the later summer and autumn months. Therefore, macrobenthos play an important role in processing organic matter, recycling nutrients, and sequestering contaminants, all of which support healthy estuarine and marine habitat.

Finally, from the biotic, food web perspective, a healthy, well-developed macrobenthic community consists of a diverse array of trophic levels and feeding guilds that utilize the variety of organic matter produced or deposited in the shallow waters of estuaries (e.g., Diaz and Schaffner, 1990; Fauchald and Jumars, 1979; Gaudênci and Cabral, 2007). Much of this production though (e.g., microphytobenthos, bacteria/detritus, phytoplankton) is not directly available to these transient fauna. Macrofauna however, can directly consume most types of bacterial or primary production and via their own somatic growth, accumulate the energy and material in a form that can be consumed by fish or birds (Levin, 1984; Iwamatsu et al., 2007; Neuman et al., 2008). In this respect, the macrobenthos serve as a conduit for the transfer of carbon from bacterial and primary production to higher trophic levels in estuaries, most of which cannot directly consume all of these types of organic matter (Gillett, 2010). Thus macrobenthos play a key role in transfer of energy and carbon to higher trophic levels, a key ecosystem function.

Predictive Relationships with Causal Factors

Water residence time and flushing in estuaries, shallowness and salinity of the estuary, and food availability are key drivers of the relationship between benthic communiy composition and nutrient enrichment. Though retention of nutrients in estuaries is positively correlated with residence time of the water mass, the underlying mechanisms are not well understood and it is conceivable that differential effects of limiting factors other than food may obscure a relationship between nutrient load and benthic biomass production (Martinetto et al., 2006; Josefson and Rasmussen, 2000; Heip et al., 1996). In the North Bay, benthic macroinvertebrate biomass is dominated by C. amurensis and peaks in abundance in fall during the wet and normal water years, and peaks in summer and fall in dry and below normal years. Lowest abundance for this bivalve is in spring or early summer in most years. In the North Bay, where greater than 50% of the carbon budget is allochthonous, there is a weaker relationship between phytoplankton biomass and production rates of secondary macroinvertebrate biomass. In contrast, in the South Bay were the majority of carbon production in autochthonous, benthic biomass is tightly coupled with phytoplankton blooms in the spring; the bivalves C*. amurensis, Mya arenaria, Venerupis japonica, Macoma petalum,* and *Musculista senhousia* show very strong seasonal patterns with decline in abundances to near zero each winter/early spring and peaks in abundance in late spring/early summer. The amphipods (*Corophium heteroceratum* and *Ampelisca abdita*) can show similar annual patterns except during dry years when *A. abdita* can persist through the winter.

This synthesis suggests that accounting for benthic biomass is a critical co-factor in modeling the relationship between nutrient loads and phytoplankton productivity. Development of a model that predicts benthic taxonomic composition as a function of nutrient loads and other co-factors is complicated. It remains difficult to identify a benthic response to eutrophication when contamination commonly covaries with many of these other environmental factors (Nichols, 1979).

Scientifically Sound and Practical Measurement Process

Macrobenthos are relatively easy to quantitatively sample, especially in soft sediments. Samples of sediment and macrobenthos can be collected with a variety of grabs (e.g., Smith-MacIntyre grab, Van Veen grab, Young grab) or cores (e.g., box cores, push cores, or vibrating cores) that can be deployed from various sized vessels, by divers, or by wading in shallow water (see review in Holme and McIntyre ,1984). Once collected, organisms can be separated from the sediment using sieves with a variety of mesh sizes. Macrobenthic fauna are typically collected with a 500-µm sieve, though larger sized meshes can be used to simplify sample processing or to establish size-spectra within the community, while smaller sized meshes are used to sample juvenile macrobenthic fauna (e.g., Edgar, 1990). The selection of sampling gear and sieve size is an important consideration, as they will both influence the characterization of the macrobenthic community. Different gear types sample to different depths in the sediment and larger sample areas will have a greater likelihood of collecting rarer taxa. Different sieve sizes will retain or exclude different size classes of organisms, which will influence abundance and biomass measurements – especially for small fauna like oligochaetes and polychaetes (Gillett et al., 2005). Most macrobenthic monitoring programs in California have refined their protocols to using a Van Veen grab for sample collection and a 1-mm mesh sieve for sample processing to balance community characterization and ease of sample processing (Smith et al., 2001; Bay et al., 2009; Ranasinghe et al. 2009).Standardized protocols for sampling of taxonomic composition and abundance are currently part of the SWRCB 's sediment quality objective protocol (www.swrcb.ca.gov/water_issues/programs/bptcp/sediment.shtml).

It should be noted that these are applicable in shoal areas, and channel edges, however, the difficulty lies in the ability to sample the benthos in open water. In the SF Bay, that there is no recent comprehensive study of species composition and distribution is a major limitation and data gap. Long term studies in specific areas by Thompson and others are helpful, but frequent repetitions of Nichols 1986 survey is imperative in order to develop and maintain an accurate and useful dataset.

Acceptable Signal to Noise Ratio

There is a large amount of information available on the effects of eutrophication on the macrobenthic community. The major impediment to the development of community-based indicators specifically in estuaries is partially due to the variable nature of the estuarine environment and the physiological stress this places upon endemic estuarine fauna (Dauvin, 2007; Dauvin and Ruellet, 2009). The estuary represents an ecotone between the marine and freshwater systems and the fauna that inhabit this area are a mix of organisms invading (at geologic time scales) landward from the coastal ocean and seaward from riverine systems (Attrill and Rundle, 2002). The osmotic stresses of fluctuating salinity, the physical stress of tidal erosion/deposition of surface sediments, and other natural stressors act in concert to select for fauna that are relatively predisposed to be tolerant of environmental stressors, which may make them better adapted to deal with eutrophic stressors than fauna from more stable marine or freshwater systems. This problem has been referred to as “the estuarine quality paradox” (Elliot and Quinto, 2007); where the paradox is how to define or detect anthropogenic reductions in habitat quality on a community that is adapted to deal with changing physical conditions and high rates of primary production naturally occurring in estuaries (Dauvin, 2007; Dauvin and Ruellet, 2009). This problem is even further complicated when looking at eutrophic impacts, particularly at the beginning of the eutrophication process where impacts maybe more subtle. Though this paradox makes it a challenge to use macrobenthos as an indicator of eutrophication in estuaries, it is not impossible given the use of the macrobenthos as a monitoring tool in estuaries around the world (Diaz et al., 2004). If the community characteristics that are chosen to be used as indicators are sensitive/or unique only to eutrophication and if the choice of reference condition(s) incorporates the environmental variation of the estuarine ecosystem by stratifying sampling and assessment tools by environmental gradients (e.g., Weisberg et al., 1997; Llansó et al., 2002) then the problems associated with the estuarine quality paradox can be reduced.

There are a number of reasonable conceptual models and experimental data to describe how eutrophication alters the composition and functioning of macrobenthic communities. Using the macrobenthic community for detection and quantification of eutrophic conditions in estuarine systems is complicated however, because most water bodies that experience eutrophication are also subject to a variety of other stressors (e.g., chemical contamination or physical disturbances) that have been shown to effect macrobenthic community structure (USEPA, 2008). There is a wide array of different chemicals that that accumulate in estuarine sediments, including organic compounds, heavy metals, pesticide, pharmaceuticals (Sanger et al., 1999a,b; Kennish, 2002). Many of the these chemicals can have toxic effects on the macrobenthos at the community level, reducing the number of sensitive taxa and overall community species richness, but without the potentially positive effects that the extra organic matter from eutrophication can create (Peterson et al., 1996; Gaston et al., 1998; Dauvin, 2008). Additionally, the effects of many contaminants are taxonomically specific (organotins and gastropods, pesticides and crustaceans, or metals and annelids) (Rand et al., 2000; Valiela, 1995). Because of the diversity of chemical contaminants in estuarine sediments, there are not any generalized models of contaminant-driven changes in community structure like the Pearson-Rosenberg (1978) model. An overall loss in community diversity and disproportionate mortality among sensitive taxa with increasing chemical contamination should be expected (Peterson et al., 1996; Rakocinski et al., 1997; Gaston et al., 1998), but the impact on community abundance and biomass is unknown. That said, abundance and biomass should not increase, as they can with non-hypoxic eutrophication.

As alluded to earlier, physical disturbance of the benthic habitats, either natural (large storms or ice scour) or anthropogenic (dredging or benthic trawling), can have important influences on the structure of the macrobenthic community. These types of disturbance can defaunate a habitat and the recovery of the community will occur in a relatively predictable fashion through time (Rhoads and Boyer, 1982; Rhoads and Germano, 1986). At the beginning of the successional process, many of the same species that are pioneering, opportunistic organisms are also resistant to the stressors of eutrophication and the accumulation of organic matter in sediments (Pearson and Rosenberg, 1978; Rhoads and Boyer, 1982; Gray et al., 2002). The model of macrobenthic community succession of Rhoads and Boyer (1982) is conceptually almost the mirror image of Pearson and Rosenberg's (1978) organic enrichment model. Physical disturbances severe enough to "restart" the successional process in estuarine systems are, however, much more stochastic than eutrophic stressors, which are persistent, systemic problems that even when corrected in the water column, have a legacy of organic matter in the sediments that will continue to negatively affect benthic fauna for a 5-10 years (e.g., Rosenberg, 1976; Borja et al., 2006; Tett et al., 2007; Diaz et al., 2008). In contrast, recovery time from large-scale physical habitat disturbance like dredge-material disposal occurs over 2-3 years, with detectable changes in community structure in the short-term (e.g., Zajac and Whitlach, 1982b; Wilbur et al., 2008; Schaffner 2010). Given these temporal differences, year-to-year comparisons of community data should allow for the separation of physical stress (significant year-to-year change in structure) and eutrophic stress (less year-to-year change) on the macrobenthic community. Additionally, like the chemical stressors, physical stressors should not have the positive biomass/production benefits to the macrobenthic community that accompany eutrophication of an ecosystem.

5.5.8 Summary: Use of Macrobenthos for San Francisco Bay NNE

Overall, macrobenthos appear to satisfy three of four review criteria. Macrobenthic taxonomic composition, abundance and biomass have the potential to be used as a supporting indicator in a NNE assessment framework in enclosed bays and estuaries with salinities of > 10 psu. Using only singular aspects of macrobenthic community structure (i.e., taxonomy, abundance, or biomass) will likely not be a robust method to assess eutrophication or, more generally, the trophic-state of an estuary. Though simpler metrics should be tested as well, it is most likely that a combination of all three aspects of macrobenthic community structure will prove to yield the best assessment tool. Measures of mean per capita biomass (community biomass ÷ community abundance), relative biomass distribution among different taxonomic or ecological groups, or the species-specific abundance in different size classes of organisms are slightly more complex measures than total abundance or a species list, but they also have the potential to capture more subtle changes in community structure brought upon by eutrophic stressors.

The review criterion not well satisfied is the ability to establish a predictive model between macrobenthic taxonomic composition, abundance, biomass and nutrient loads. Science is evolving in this area, and thus predictive capacity may be possible in the future. However, available data and our understanding of factors driving macrobenthos in SF Bay are lacking. Thus while macrobenthos may be a useful supporting indicator or co-factor in shallow muddy, subtidal habitats of SF Bay, it is not likely to be a primary NNE indicator in the near term. Because macrobenthic taxonomic composition and abundance are already standardized components of the State's ambient monitoring program, it merits considering what key data gaps and next steps would be necessary to use macrobenthos as an additional supporting line of evidence to diagnose eutrophication.

Several key data gaps exist and a number of steps will be required in order to determine the ultimate utility of macrobenthos in this capacity. First, we recommend assembling a workgroup to identify potential species or metrics based on taxonomic composition and abundance, then use existing data collected through EPA Environmental Monitoring and Assessment Program (EMAP) and regional monitoring programs to test out the utility of these metrics as a tool for eutrophication. Second, though it is not currently collected in California's existing, state-wide monitoring programs, biomass data (or a reasonable approximation thereof) are probably going to have to be collected to successfully distinguish eutrophication from other stressors. We recommend a small pilot project, which would include the collection of macrobenthic biomass in a new ambient monitoring framework, in order to test out the applicability of these kinds of data to detect eutrophication. Finally, looking at spatial variation will also likely be necessary to separate different stressors: where measures of poor/impacted community structure with less spatial variation would be indicative of eutrophic stress on the community, compared to those with larger spatial variation, which would be indicative of other types of physical disturbance or successional changes in the community. Different combinations of metrics or differential thresholds will likely have to be implemented to tailor any assessment tool to the different sediment, salinity, or flow regimes within and among California's varied estuarine systems.

5.6 Jellyfish

The term 'jellyfish' refers to free-floating gelatinous animals (Figure 5.6.1) belonging to the phylum *Cnidaria* (hydromedusae, siphonophores and scyphomedusae) and to planktonic members of the phylum *Ctenophora* (Mills, 2001; Richardson, 2010). Although many *Cnidaria* are able to actively swim by contracting the muscles of their bells and *Ctenophora* are able to propel themselves by the sequential beating of cilia, neither can swim against currents and are therefore defined as zooplankton (Bushek, 2005). These organisms share many characteristics including their watery or 'gelatinous' nature, and a role as higher-order carnivores in plankton communities. They are often referred to as gelatinous zooplankton (Mills, 2001; Lucas, 2001; Pitt, 2007).

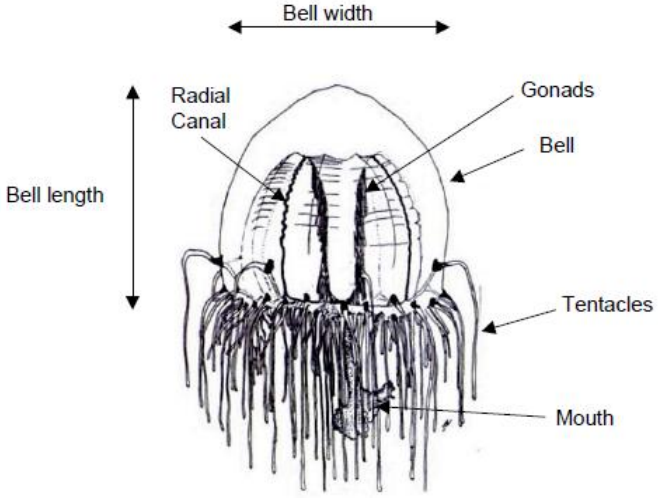

Figure 5.6.1. Main body parts of a jellyfish (Herdon et al., 2003).

The life cycle of the jellyfish is biphasic; an alternation between a small, often unseen benthic polyp and a free-floating/swimming, open-water medusa. The polyp produces medusae by asexual budding, whereas the medusa reproduces sexually (Bushek, 2005; Lucas, 2001; Pitt, 2009; Purcell, 2005). The asexual reproduction process is seasonal in temperate climates and varies between species, with the period of medusa budding varying from days to months long, but nearly always resulting in seasonal appearance and disappearance of medusa populations (Mills, 2001). This alternation of life stages means jellyfish can be present even when they are not obvious to the casual observer and more information is known about the medusa stage than the polyp stage (Wintzer, 2010; Bushek, 2005). The ability of these gelatinous species of cnidarians (*Scyphozoa, Cubozoa, Hydrozoa*) to occur in large numbers, or, to bloom, is due to having both asexual and sexual reproduction (Purcell, 2007). The life history of ctenophores does not include an asexual benthic stage. Larval ctenophores grow and bloom directly into adults without passing through the polyp stage. Perhaps because of their fragility, little is known about factors that affect their sexual reproduction and population abundances (Purcell, 2005).

Most populations are found in small shallow coastal embayments, fjords, and estuaries where there are suitable substrates for the benthic polyp to attach, and limited tidal exchange, with variable productivity, and food availability (Lucas, 2001). Most gelatinous zooplankton are suspension-feeders and respond quickly to changes in food availability by exploiting their high feeding and growth rate potentials; during starvation periods of up to 25 days, they survive by shrinking, instead of dying (Moller, 2007).

The life cycle of a jellyfish is normally less than one year, yet gelatinous zooplankton come and go seasonally (seasons of reproduction vary greatly by species), appearing and disappearing with annual regularity (Mills, 2001). However, medusa and ctenophore populations also undergo interannual variation, some years blooming with much greater intensity and much larger populations of each species than others (Mills, 2001).

5.6.1 Applicable Habitat Types

As an indicator, jellyfish would be applicable to the subtidal habitats of the North, Central and South Bays.

5.6.2 Indicator Relationship to Nutrients and Water Quality

The input of excessive nutrients from fertilizer runoff, sewage and other anthropogenic sources into estuaries has been shown to greatly alter pelagic communities (Richardson, 2010; Pitt, 2007; Mills, 2001). Nutrient enrichment stimulates primary production, increasing the biomass of phytoplankton. Feeding rates of grazers then increase, which stimulates secondary and, potentially, higher-order production (Pitt, 2007). As more food becomes available, polyps and jellyfish increase asexual production and sexual reproduction, allowing populations to “bloom” (Pitt, 2007; Purcell, 2001; Bushek, 2005).

Jellyfish populations are characterized by large and rapid fluctuations in abundances and “boom and bust population dynamics” and thus often represent a substantial proportion of the pelagic consumer biomass (Condon, 2010; Pitt, 2009). During bloom formation, when both individuals and populations are increasing in size, jellyfish and ctenophores act as a net sink for C, N and P, rapidly assimilating carbon and nutrients from their planktonic prey (Pitt, 2009). Because of their high biomass during blooms, gelatinous zooplankton can influence nutrient cycling as they both excrete and take up dissolved organic matter, inorganic nitrogen and phosphorus (Condon, 2010; Pitt, 2007; Welsh, 2009). When in high abundances, gelatinous species may contribute significantly to nitrogen and phosphate budgets (Pitt 2007; West, 2009).

Eutrophication and Hypoxia

Large phytoplankton blooms, as discussed above, and dead jellyfish resulting from nutrient enrichment can sometimes sink to the seafloor, where their bacterial degradation can cause localized hypoxia (Richardson, 2010; West, 2009). Polyps and medusae are more tolerant to lower oxygen conditions than fish, which often ensure jellyfish survival over fish during hypoxic events (Richardson, 2010). Fish avoid, or die in, waters of ≤2–3 mg O_2 L^{-1} but many jellyfish species are tolerant of ≤1 mg O_2 L^{-1} (Pitt, 2007). However, though tolerant of low dissolved oxygen at adult stages, several species can, in fact, be intolerant at polyp stage (Purcell, 2001; Wintzer, 2010).

Eutrophication has been suggested to be an important environmental factor for increasing mass occurrence of jellyfish (Mills, 2001). As jellyfish are tolerant to low dissolved oxygen concentrations (Pitt, 2007; Purcell, 2001), they can take over oxygen depleted waters previously inhabited by zooplanktivorous fish. For example, "Skive Fjord (Denmark) suffers every summer from oxygen depletion in the near-bottom water causing large amounts of nutrients (phosphate and ammonium) to be released from the anoxic sediment. This subsequently stimulates a phytoplankton bloom, followed later on by an increase in the zooplankton. The surface chlorophyll a concentrations may become very high during periods with exceptionally severe oxygen depletion. In certain years when biomass occurrence of *Aurelia aurita* in high, peak chlorophyll a concentrations as high as 60 to 80 µg L^{-1} have been measured in Skive Fjord because the jellyfish effectively eliminate the zooplankton-grazing impact on the phytoplankton bloom (Moller, 2007). It seems that jellyfish may benefit from eutrophication, which can increase small-zooplankton abundance, turbidity and hypoxia, all conditions that favor jellyfish over fish. Overfishing can also remove predators of jellyfish and zooplanktivorous fish competitors as well as cause large-scale ecosystem changes that improve conditions for jellyfish (Purcell, 2007).

On a micro-scale, the presence of individual jellyfish was shown to have an influence on benthic oxygen and nutrient dynamics by researchers at the Australian Rivers Institute (Welsh, 2009). Sediment patches occupied by individual jellyfish showed dramatically different dynamics than adjacent unoccupied or "bare" sediments, and also varied temporally. For example, during the night the presence of a *Cassiopea spp*. individual enhanced benthic respiration by 3.6-fold and benthic ammonium regeneration rates by 4.5-fold. "However, during the high light period, photosynthetic oxygen production by the jellyfish increased benthic oxygen production by almost 100-fold and although the sediment alone was net source of ammonium to the water column, ammonium assimilation by the jellyfish reversed this flux creating a benthic sink for water column ammonium (Welsh, 2009)."

Another relationship change which favors jellyfish is the decline, in the Chesapeake Bay for example, of oyster populations. Phytoplankton that would have been consumed by oysters now is available to zooplanktivores such as medusae, increasing their populations, leading to the hypothesis that if oysters were restored to their former abundance in Chesapeake Bay there would be a reduction in gelatinous zooplankton (Purcell, 2001).

Impact on Pelagic and Benthic Communities

In addition to eutrophic conditions, increased nutrients from sewage effluents or fertilizers in estuarine environments may change plankton food webs towards small phytoplankton (or microplankton) and zooplankton species causing a trophic cascade. This size reduction to lower trophic levels is considered to favor gelatinous zooplankton which are non-visual and consume small and large size of prey, over fish (Purcell, 2001, 2007; Pitt, 2007; Mills, 2001). Predators such as *A. aurita* may also affect fish standing stocks, either directly by predation on fish larvae, or indirectly by competing with fish larvae for available food resources (Lucas, 2001). Studies in the Baltic Sea of populations of A. aurita show considerable variability, but in peak densities, can consume more than 60% of the daily production of copepods and other zooplankton (Bushek, 2005). Similarly, a tendency of algal blooms related to high abundances of jellyfish was noticed by researchers in a Canadian fjord due to reduced herbivore grazing caused by a high predation impacts by medusa on zooplankton (Moller, 2007).

5.6.3 Species Composition in San Francisco Bay

San Francisco Bay has 20 native jellyfish species and four introduced, based on year round studies at 52 sites in the estuary, conducted by the California Department of Fish and Game's San Francisco Bay Study in 2000 (Herndon et al., 2003). Native species tended to be found in high salinity and cooler waters. Introduced species have been primarily found in Suisun Bay and the Sacramento San Joaquin delta where there is lower salinity, and higher water temperature in the summer (Herndon et al., 2003). Of the most common species found in the Bay, the most abundant species, both native and non-native are discussed in this section (Figure 5.6.2 and 5.6.3). Jellyfishes of all sizes capture food from suspension as they drift and swim in the water column (Bushek, 2005). However, Rees and Kitting (2002) noted that *M. marginata* were able to kill juvenile fishes in laboratory experiments, and Schroeter (unpublished data) found goby larvae in 6 out of 39 medusae collected in July 2004 from Suisun Marsh (May, 2006).

Invasive jellyfish have been documented to have severe effects on the ecosystems they invade because many species are voracious predators, consuming large amounts of prey and disrupting planktivorous food webs more than native hydrozoan species (Mills and Sommer, 1995). Preliminary information on diet of invasive hydromedusae in the SF Bay shows that they feed on a wide variety of planktonic species, including larval fishes (Mills and Sommer, 1995, R. E. Schroeter, unpublished data).

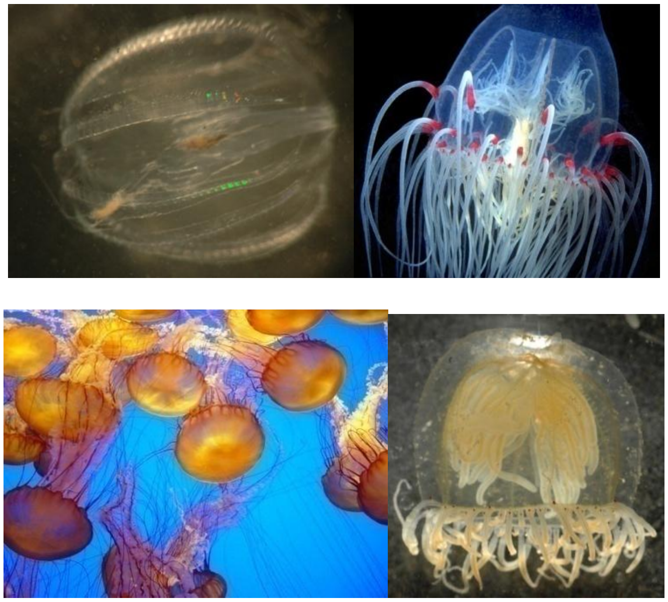

Figure 5.6.2. Native species: Starting from top left clockwise: *Pleurobrachia bachei*, "Sea Gooseberry" or "Comb Jelly" (photo: Dave Cowles 2007), *Scrippsia pacifica* (photo: Garry McCarthy, 2000), *Polyorchis penicillatus* (photo: Dave Cowles 2006), *Chrysaora fuscescens* or "Sea Nettle" (photo: Monterey Bay Aquarium).

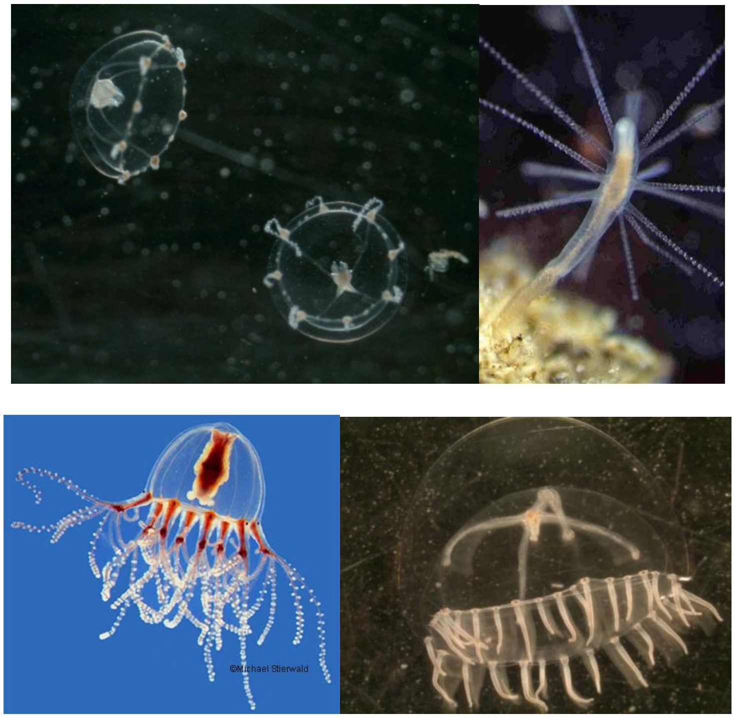

Figure 5.6.3. Introduced species: Starting top left clockwise: *Blackfordia virginica* (photo: Meek, UC Davis, 2010), *Cordylophora caspia* (photo: USGS invasive aquatic species), *Maeotias margina* (photo: USGS invasive aquatic species), *Moerisia sp.* (photo: Meek, UC Davis, 2010).

5.6.4 Indicator Trends and Factors Effecting Temporal and Spatial Variation

Climate and micro-climates, seasonal variation, water quality changes from land uses, the Pacific decadal oscillations, and other oceanic shifts such as upwelling are factors which effect the temporal and spatial variation of jellyfish populations. Furthermore, substrate composition, temperature and salinity levels impact communities on a shorter time scale. A study on spatial and temporal variation and abundance looking specifically at the polyp stage of the four non-natives species demonstrated that water quality factors are strongly correlated with the majority of variation of seasonal observations (Wintzer, 2010).

Spatial Variability

Polyorchis penicillatus has been found as far upstream as Suisun Bay and as far south as the Dumbarton Bridge, with the highest concentration in Central Bay (Figure 5.6.4) (Herndon et al., 2003). Pleurobrachia bachei has been found as far upstream as Suisun Bay and as far south as the Dumbarton Bridge, with the highest concentration in South Bay (Figure 5.6.4) and in a study of macroplankton species composition in the SF Bay between 1997 and 2000 was found to be the most abundance macroplanktonic species living in the estuary (Herndon et al., 2003; Gewant, 2005). *Maeotias marginata* is an introduced species, most commonly found in Suisun Bay and West Delta areas (Figure 5.6.4). *Scrippsia pacifica* is a native jellyfish with a distribution ranging from the southern half of San Pablo Bay to most of South Bay (Figure 5.6.4). *Chrysaora fuscescens* or "Sea Nettle" is a native species of jellyfish found in the southern part of San Pablo Bay and Central Bay (Figure 5.6.4) (Herndon et al., 2003).

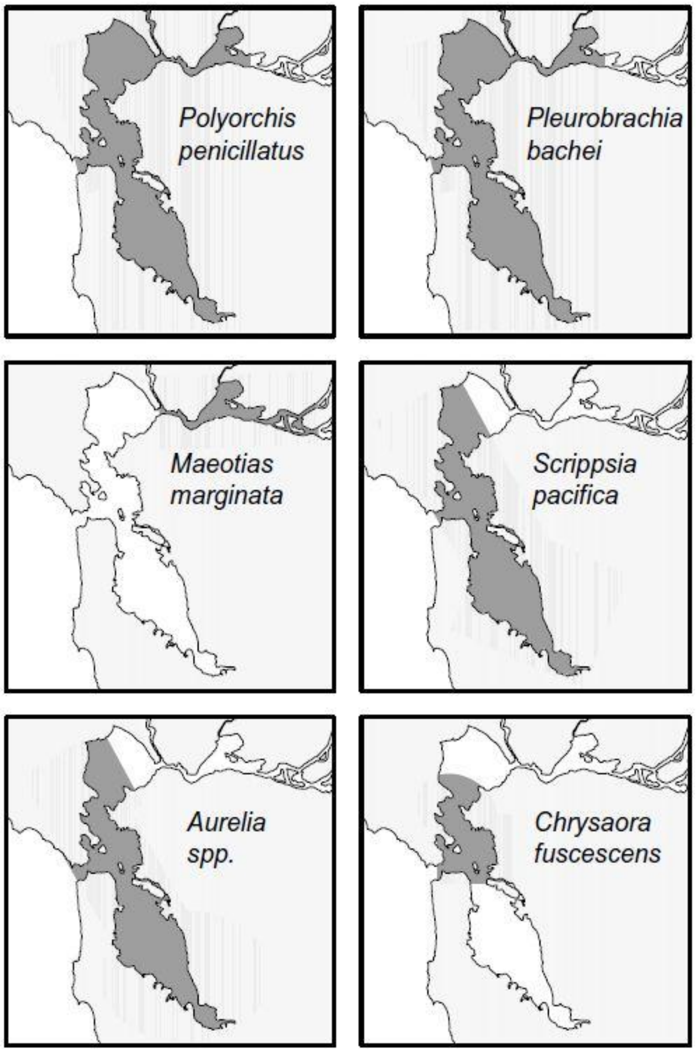

Figure 5.6.4. Maps of common jellyfish species found in the San Francisco Bay (Herndon et al., 2003).

Researchers at the Romberg Tiburon Center trolled the Bay monthly between September 1997 and December 2000 at six stations spanning the North, South and Central Bays. They found that macrozooplankton and micronekton communities were dominated by four fishes and seven invertebrates which comprised 98% of the total catch (Gewant, 2005). Polyorchis penicillatus, was the most commonly occurring, specifically in Central Bay (station 17), but reached maximum densities (70 individuals/ 1,000 m^3) in the North Bay in the winter of 1998-1999 (station 13 and 15) (Figure 5.6.5) (Gewant, 2005).

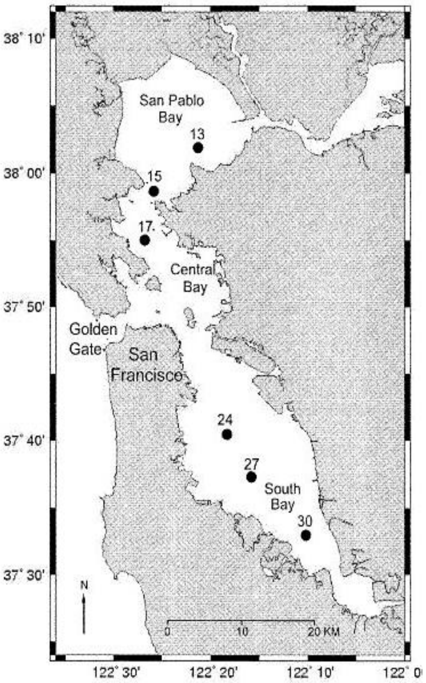

Fig. 1. San Francisco Estuary, showing the location of th six stations (13, 15, 17, 24, 27, and 30) sampled in this study.

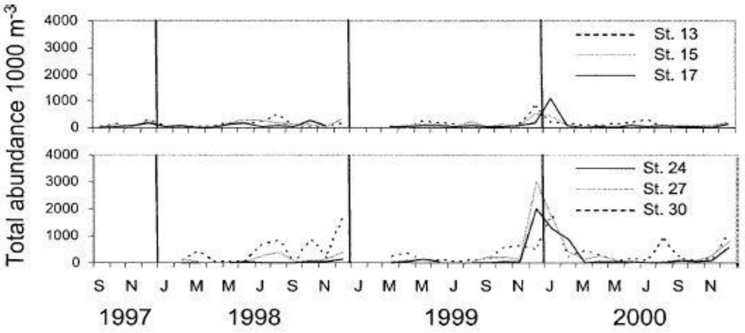

Figure 5.6.5. San Francisco Bay showing locations of the six sample stations, and abundance of macrozooplankton and micronekton in the San Francisco Bay (1997-2000) (Gewant, 2005).

At a more localized scale, individual jellyfish may represent local sources/sinks for oxygen and nutrients their distributions may enhance spatial heterogeneity of the benthos at the small (adjacent patches of sediment with and without a jellyfish) and medium (nearby areas of sediment colonized or not colonized by jellyfish) scales (Welsh, 2009).

Temporal Variability

Polyorchis penicillatus is a native species of jellyfish, found year-round, but with increased abundance during the winter months, perhaps due to their preference for lower salinity levels exhibited by native species, specifically November and December although largely absent during the winter of 1998– 1999 according to Gewant et al. (Herndon et al, 2003, Gewant, 2005). Pleurobrachia bachei, "Sea Gooseberry" or "Comb Jelly" is a native ctenophore that only occurs only in winter and spring. Maeotias marginata, an introduced species, is highly seasonal, found only during summer and fall (Herndon et al., 2003).

In Gewant's study, *P. bachei* and *Polyorchis spp.,* showed very strong seasonal occurrence patterns. They were captured exclusively in the late fall and winter months, during periods of high salinity, and then disappearing by early spring (Gewant, 2005). However, questions remain as to whether these large numbers were advected into the Bay by wind or currents or if they bloomed within the Bay from a resident seed population. The appearance of peak abundances in North and South Bays, but not Central Bay, which is most proximate to the coastal ocean, is particularly noteworthy (Gewant, 2005).

In another study, researchers at UC Davis found that the polyp phases of *Moerisia sp*. and *B. virginica* followed the common recruitment pattern in benthic hydroids in temperate waters, showing a peak in productivity during spring and summer, which slows down in fall and winter (Wintzer, 2010).

Invasive Species

The four invasive hydrozoans, *Maeotias marginata, Blackfordia virginica, Moerisia* sp., and *Cordylophora caspia*, have become established in the brackish waters of the SF Bay (Figure 5.6.6), where they reach seasonally high abundances during medusae blooms (June-November) (Mills and Sommer 1995; Mills and Rees, 2000). In Suisun Marsh, R. E. Schroeter (UC Davis) recorded Moerisia densities of more than 500 individuals per m^3 (R. E. Schroeter, unpublished data) and “tens of thousands” of *M. marginata* have been collected in the Napa River during July 2003 surveys alone (May, 2006).

Indicator Trends

Many jellyfish populations appear to be increasing around the world, most likely in response to human-induced alterations of the oceanic environment, such as global warming, eutrophication, and over-harvesting of fish stocks (Mills, 2001; Wintzer, 2010). Analyses of several long-term (8- to 100-year) trends in jellyfish populations demonstrate that their abundances vary with climate, often at decadal scales (reviewed in Purcell 2005). Some evidence suggests continued upward trends; however, recent time series are still too short to exclude decadal climate cycles (Purcell, 2007). Bushek (2005) hypothesizes that climate fluctuations serve as the major source for interannual variation in jellyfish populations, due to the changes in primary production, zooplankton, fishes and seabird abundance in the 20th century (Bushek, 2005). During the summer of 2007 in the Gulf of Mexico, nutrient-rich outflows from the Mississippi River resulted in large phytoplankton blooms and 25 000 km^2 of oxygen-depleted waters, favoring jellyfish because of their tolerance for low dissolved oxygen as compared to commercially valuable fish and shellfish (Richardson, 2010). Finally, warming of the oceans may increase many populations of gelatinous species and also shift the population distributions poleward, as seems to be occurring for the ctenophore *Mnemiopsis leidyi* (Purcell, 2007). Warming of the sea surface can enhance water column stratification, leading to nutrient-poor surface waters where flagellates, because of their ability to migrate vertically into nutrient-rich deeper waters, can out-compete diatoms. Such flagellate-dominated food webs might be more favorable for jellyfish than for fish (Richardson, 2010). Warmer temperatures have also been shown to accelerate medusae growth and reproduction (Purcell, 2007; Richardson, 2010).

Locally, limited data are available on jellyfish populations, though Peter Moyle has tracked the invasion of *Maeotius marginata* in the Suisun Marsh since 1981, showing a growing abundance of the invasive gelatinous zooplankton (Moyle, ppt, 2009). The Pacific Decadal Oscillation (PDO) reversed signals in 1999 and has remained negative (Cloern et al., 2010; Peterson and Schwing, 2003) which has been supported by cooler temperatures, increased salinity, weakened winds, a doubling of zooplankton biomass and a favoring of cool water species over warm (Peterson and Schwing, 2003). These changes also imply a shift to stronger coastal upwelling which would continue to support jellyfish populations, and though Anderson and Piatt (1999) note that jellyfish populations increased with the last positive PDO cycle, there is perhaps, conflicting evidence regarding what impact the negative PDO cycle will have on gelatinous zooplankton in the SF Bay (Cloern et al., 2010; Anderson and Piatt, 1999).

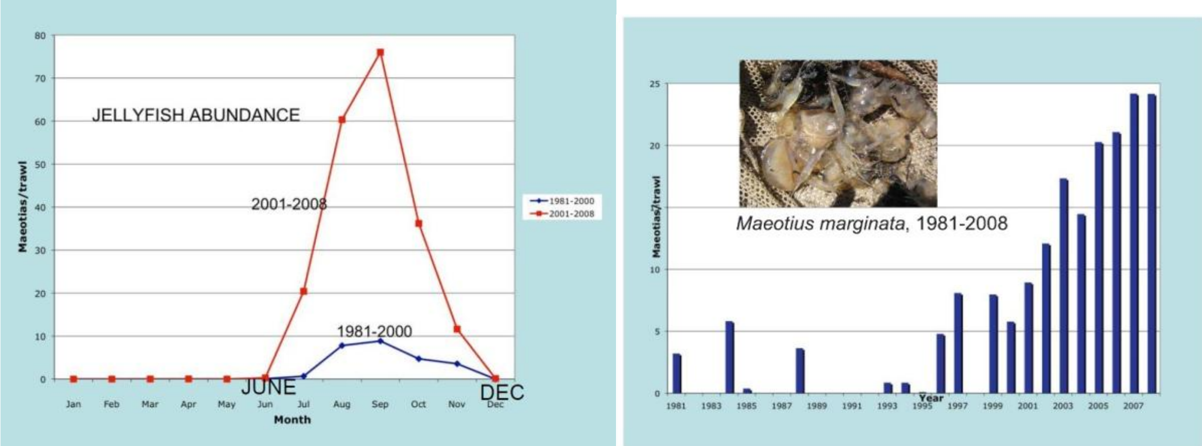

Figure 5.6.6. Invasive jellyfish abundance over time in Suisun Marsh (Moyle ppt. 2009).

5.6.5 Utility of Indicator as an Eutrophication Indicator for San Francisco Bay

Clear Linkage to Beneficial Uses

Jellyfish often have direct negative impacts on beneficial uses of estuaries and open waters. They impact primary and secondary contact recreation by stinging and deterring swimmers sail boarders, and boaters, they can increases costs to marine businesses by clogging fishing nets and can kill fish in aquaculture net-pens. They can also interfere with power plants by clogging cooling-water intake screens (Purcell, 2007; Richardson, 2010). They have indirect effects on fisheries by feeding on zooplankton and ichthyoplankton, and, therefore, becoming predators and direct competitors of planktivorous fish for available food resources (Purcell, 2007; Purcell, 2001).

Predictive Relationships to Causal Factors

While jellyfish populations appear to increase as a consequence of nutrient enrichment and increases in hypoxic conditions, they also increase in relation to mortality rates of competitors, harvesting of zooplanktivorous fish and overfishing, and other predators. There are also changes in jellyfish populations due to increases in water temperatures and other global processes, such as the reverse in the Pacific decadal oscillation (Cloern et al., 2010; Purcell, 2001). These many reasons which explain changes in jellyfish populations create a challenge in how to isolate nutrient enrichment from other causes of change. Therefore, presently there is not a proven predictive relationship with causal factors which indicate eutrophication and the many cofactors which influence this relationship would need to be explored. Anderson and Piatt (1999) suggest the need to observe changes over a complete climate cycle, if not several, before confidently drawing conclusions between climate change and the biological impacts of jellyfish (Anderson and Piatt, 1999).

Sound and Practical Measurement

Although Peter Moyle's long term study on Suisun Marsh has shown increasing invasive jellyfish populations, there are very few other long term studies tracking invasive species colonization of the SF Bay, as well as long term monitoring of blooms of native jellyfish populations. The use of jellyfish as an indicator for nutrient numeric endpoints in the SF Bay is challenged by this lack of long term distribution and abundance datasets from species specific sampling for both invasive and native species, as well as a lack of biological understanding of life histories of jellyfish (May, 2006; Purcell, 2007; Mills, 2001, Mills and Sommer, 1995). Environmental changes affect both the benthic and pelagic stages of the jellyfish though, even less is known about the benthic polyp stages than the pelagic jellyfish (Purcell, 2007). Knowledge of both life stages is essential to understanding the causes of blooms well as being able to predict and manage any resulting impacts (Mills, 2001; Wintzer, 2010).

Acceptable Signal to Noise Ratio

The existing data on the relationship between jellyfish and eutrophication, although anecdotally compelling, do not support an acceptable signal to noise ratio as understood thus far. The changes in nutrient enrichments over time do not correlate to the studies of population dynamics and spatial and temporal variation of jellyfish and thus more models or experiments would need to be developed to confidently associate jellyfish dynamics with a nutrient numeric endpoint.

5.6.6 Summary: Use of Jellyfish as an NNE Indicator

Based on this review, jellyfish do not meet the evaluation criteria as an acceptable indicator for the SF Bay NNE. We do not recommend pursuing jellyfish as an indicator further for this purpose.

5.7 Dissolved Oxygen

Dissolved oxygen (DO) refers to the concentration of oxygen in the water column (reported in units of concentration (mg L^{-1}) or percentage of estimated saturation). Adequate DO is required for the health of aquatic systems and organisms. Hypoxia is the term used by scientists studying aquatic systems to describe stress of organisms (usually fish but also invertebrates) due to low oxygen. Hypoxia as a stressor differs from chemical toxicants in that it can occur naturally; hypoxia is a consequence of the balance of atmospheric oxygen diffusion to surface waters, the *in situ* production of oxygen by primary producers during daylight hours, their night time respiration, in combination with the respiration of decaying organic matter and other biogeochemical processes that consume oxygen within surface waters and sediments. In cases where hypoxia has anthropogenic origins, the assumption is that hypoxia may be reduced by controlling nutrient availability and reducing the supply oxygen-demanding material to a waterbody.

Hypoxia exhibits temporal variability, on diurnal, tidal, lunar, and seasonal timescales. Seasonal hypoxia often develops in association with stratification. Hypoxic water can occur as stratified water prevents the oxygenated surface water from mixing downward or when upwelled hypoxic water is advected into an estuary from offshore. Hypoxia can appear in water near the sediment interface when respiration in the water and sediment depletes oxygen faster than it can be replenished. Breakdown of the stratification allows the surface and bottom waters to mix. Stratification can occur in both deepwater habitat of perennially tidal enclosed bays, such as SF Bay, or in lagoonal or river mouth estuaries that are intermittently closed to tidal exchange and that are known to “trap salt” (Largier et al., 1991). Diel cycles of hypoxia often appear in stratified or unstratified shallow habitats where nighttime respiration, in combination with water column and sediment dissolved oxygen demand, can deplete DO. Tidal and lunar frequencies can become apparent, particularly in poorly flushed areas where greater exchange occurs on flood or ebb tides or during a spring tide.

Oxygen demand and resulting reductions in dissolved oxygen vary spatially and temporally and may be more or less persistent. 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). Organisms have developed several physiological and behavioral adaptations to deal with temporary periods of low oxygen availability. Organisms can: 1) temporarily utilize anaerobic pathways to produce energy (ATP); 2) scavenge oxygen from hypoxic waters and increase the efficiency of oxygen transport to cells; 3) emigrate from hypoxic zones; 3) utilize the abundant oxygen from the surface or breathing aerial sources; or 4) reduce demand for oxygen by reducing activity. However, these are all short-term strategies and will not enable the animal to survive during long hypoxic periods. Adaptations are well developed in animals such as intertidal and burrowing animals that commonly experience hypoxia but poorly developed in animals that inhabit well-oxygenated environments such as the upper water column. If oxygen deficiency persists, death will ensue. Sublethal effects also occur. For example, reduced motor activity from mild hypoxia may make the animal more vulnerable to predators or decrease its growth or reproduction. Several components of SF Bay can have an associated oxygen demand, e.g.:

- Organic rich waste loads from agricultural, municipal, or industrial sources
- Allochthonous or autochthonous organic matter produced as live or dead plants, algae, and animal tissue, found in sediments or surface waters
- Chemical oxygen demand resulting from redox reactions in sediments or surface waters

5.7.1 Applicable Habitat Types

Dissolved oxygen is applicable principally to the subtidal habitats of the North, Central and South Bays. Utility and applicability of DO to diked Baylands requires additional discussion, particularly because muted habitats are to some degree subject naturally to hypoxia. In addition, these habitat types are known to influence subtidal DO. Low DO water can exist in salt ponds that, if breached, can supply high organic discharge or low DO discharge to the Bay (Shellenbarger et al., 2008; Thebault et al., 2008). Similarly, managed duck ponds can also have local influences on DO if waters from duck clubs are allowed to mix with Bay waters too rapidly.

5.7.2 Available Data on Dissolved Oxygen

The USGS has been collecting water quality data in San Francisco Bay on water quality and nutrients continuously for 39 years beginning in 1968 (see Section 5.2). It is important to note that this was a research program drive by hypothesis related questions. There are gaps in the data set. For example, the USGS didn't collect DO data from the mid 1970s to the mid 1980s. That accepted, the database generated presently includes over 6900 discrete laboratory measurements of dissolved oxygen in water samples and 119,685 estimates of dissolved oxygen made from a linear relationship between the oxygen electrode voltage output and discrete lab measurements. Without support data sets like this may not always continue into the future since the USGS research program is not mandated. There is the critical need for a commitment to support regular sampling to measure and understand future changes in DO related to nutrient enrichment.

5.7.3 Indicator Relationship to Nutrients and Water Quality

San Francisco Bay has generally been considered a nutrient enriched but low primary production environment (Cloern, 1987; Cloern et al., 2005b; Wankel et al., 2007; Cloern et al., 2007). Factors that limit primary production in the Bay include light limitation due to high turbidity in the water column, strong physical tidal/wind forcing that prevents thermal/saline stratification, and high predation from filter feeding bivalves (Cloern, 2001; Cloern et al. 2007). Due to these controls, SF Bay has not experienced the water quality issues e.g., hypoxia and anoxia associated with high nutrients and high primary production that have impacted other coastal estuaries (Cloern et al., 2001; Diaz, 2001; Wankel et al., 2006; Breitburg et al., 2009).

Therefore, SF Bay is not considered impaired by low dissolved oxygen conditions. Based on data collected by the USGS, minimum concentrations have only rarely dipped below 5 mg L^{-1} (the water quality objective outlined in the San Francisco Bay Basin Plan (SFBRWQCB, 2007)) on a few occasions in the Central, South and Lower South Bay segments (Table 5.7.1). Central Bay had the highest incidence of dissolved oxygen levels less than 5 mg L^{-1}. However, in the past decade there has been an increase in primary production, including a new South Bay autumn phytoplankton bloom. This period of increased primary production has coincided with decreases in nutrient inputs (due to improved wastewater treatment) over the same time period (Cloern et al., 2007). There has also been a co-occurring shift in Eastern Pacific oceanic conditions and the California Current that has resulted in favorable Bay conditions for bivalve predators, a subsequent reduction in bivalve biomass, and increased primary production (Cloern et al., 2007).

A major threat, in high nutrient estuarine systems, is the potential for increased primary production, including HABs, and the subsequent increase in heterotrophic activity that can lead to hypoxic and anoxic conditions (Diaz, 2001). This scenario has played out in many estuaries around the world (Diaz, 2001; Breitburg et al., 2009). San Francisco Bay has had incidents of HABs in the past 15 years (See phytoplankton section Table 5.2.2). However, there were no reports of reduced dissolved oxygen during or following a large dinoflagellate bloom in South SF Bay (maximum chlorophyll a concentrations were 195 mg/m^3 which was 65 times higher than the 27 year average for August – October time period)

(Cloern et al., 2005b). Adequate dissolved oxygen levels were most likely maintained due to a quick dissipation of the bloom from physical forcing of the tides and winds mixing the water column. There is little information for SF Bay on dissolved oxygen concentrations during other HAB episodes. That said, it seems feasible that if the increased production trend continues or if the incidence of harmful algae blooms increases, there might be a concomitant increase in the frequency of lower DO events especially if they happen to coincide with neap tides or lower wind conditions which can lead to stratification.

Table 5.7.1. Minimum, maximum, and mean dissolved oxygen concentrations in each segment of San Francisco Bay based on data from 1999-present. The number of samples in each measurement and the percent of dissolved oxygen measurements less than 5.0 mg L^{-1} are also shown. (Source: J. Cloern, USGS): http://sfbay.wr.usgs.gov/access/wqdata).

	Calculated Oxygen* (mg/L)				
Segment	**Minimum**	**Maximum**	**Mean**	**n**	**% < 5 mg/l**
Rivers	7.2	10.9	8.9	5,071	0.00%
Suisun Bay	6.9	11.0	8.8	5,435	0.00%
Carquinez Straight	6.6	11.0	8.3	7,563	0.00%
San Pablo Bay	5.6	10.7	7.9	5,802	0.00%
Central Bay	4.2	12.3	7.7	30,616	0.20%
South Bay	4.7	14.4	7.8	18,517	0.04%
Lower South Bay	4.8	14.6	7.7	5,657	0.05%

*Note: Calculated Oxygen is the estimated concentration of dissolved oxygen, calculated from the oxygen electrode voltage calibrated with the discrete measures of the dissolved oxygen using linear regression. (USGS website: http://sfbay.wr.usgs.gov/access/wqdata).

5.7.4 Factors Affecting Temporal and Spatial Variation of Indicator

Dissolved oxygen concentration and percent saturation varies both spatially and temporally in SF Bay. Dissolved oxygen levels can range from anoxic (oxygen depletion) to super saturation (>100% saturation). Factors that control dissolved oxygen variability include tidal forcing (advection of oxygen from and to coastal waters), wind stress (oxygen exchange between atmosphere and water surface), biological activity (photosynthesis and respiration), and freshwater input (Conomos et al., 1979). Gaseous atmospheric exchange with the Bay is the greatest oxygen source.

Spatial Variability

Freshwater outflow from the Sacramento/San Joaquin Delta is the largest freshwater supply to the Bay (Conomos et al., 1979). Freshwater input from South Bay tributaries is much lower. Dissolved oxygen concentrations tend to be higher in the northern reaches of the Bay (decreasing the oxygen concentration gradient from north to south) (Table 5.7.1 and Figure 5.7.1) and in the upper portions of the water column (Conomos et al., 1979; Peterson, 1979). Water column oxygen variability is controlled by density/thermal stratification and results in higher oxygen concentrations in the upper portions of the water column. Isohaline conditions prevail during summer due to strong wind and tidal forcing which results in a mixed water column with less vertical variability in oxygen levels (Conomos et al., 1979). Decreased dissolved oxygen generally occurs in the lower portions of the water column during periods of stratification when there is high availability of organic matter (Diaz, 2001).

Temporal Variability

There are two major scales of dissolved oxygen temporal variation: seasonal and hourly (tidal and variation in photosynthesis/respiration). The Bay has a Mediterranean climate with mild wet winters and dry temperate summers. The majority of precipitation occurs from October through April with rain runoff occurring during the winter months and snowmelt runoff during the early summer (Conomos, 1979). Freshwater runoff is a source of oxygen to the Bay with maximum runoff occurring during winter storm events (Peterson, 1979; Cloern, 1996; McKee et al., 2006). Dissolved oxygen concentrations are highest during winter due to increased solubility of oxygen in colder water, decreased oxygen dependent biological activity, and increased oxygen supply from freshwater inflow (Figure 5.7.1 and 5.7.2) (Conomos et al., 1979; Peterson, 1979). Dissolved oxygen levels can also increase during the spring phytoplankton bloom (Peterson, 1979; Cloern, 1996) and can become supersaturated during these blooms (Cloern, 1996; Cloern et al., 2005). Dissolved oxygen levels are lowest during the summertime in all Bay segments (Figure 5.7.1 and 5.7.2).

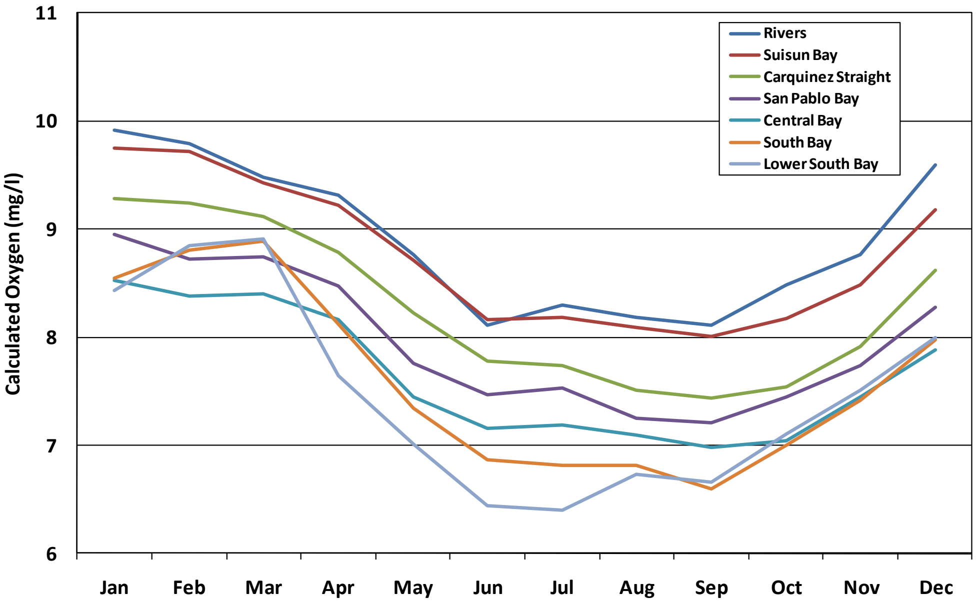

Figure 5.7.1. Monthly averaged dissolved oxygen concentrations in the defined segments of the San Francisco Bay for 1999 – 2010. Data obtained from the USGS: http://sfbay.wr.usgs.gov/access/wqdata).

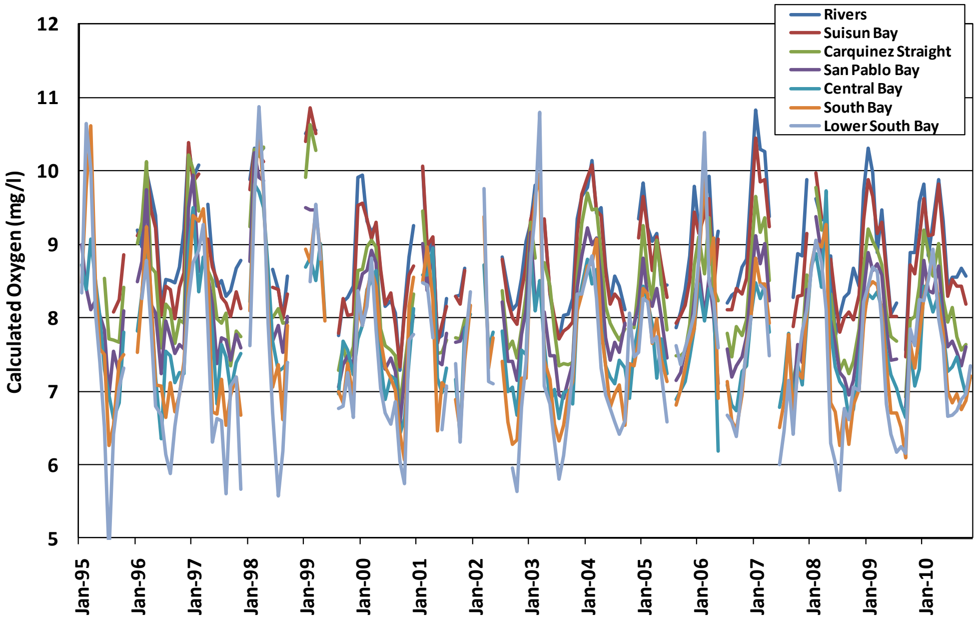

Figure 5.7.2. Average dissolved oxygen concentrations in the defined segments of the San Francisco Bay for measurements between 1995 – 2010. Data obtained from the USGS: http://sfbay.wr.usgs.gov/access/wqdata).

Hourly variability in dissolved oxygen is controlled by tidal forces (semi-diurnal) and the balance of photosynthesis and respiration. There are daily (2 high and low tidal cycles per day) tidal cycles that produce varying tidal currents with spring tides producing maximum tidal currents. Tidal currents generally supply lower oxygenated waters to the bottom waters of the Bay (Peterson, 1979). Hourly variation of dissolved oxygen is due to increased primary productivity during daylight hours and subsequent decreases in primary productivity during nighttime hours (Thebeault et al., 2008).

Indicator Trends

Generally speaking, dissolved oxygen concentrations and percent saturation in the Bay have been fairly consistent, over the long term, since implementation of secondary wastewater treatment (Cloern, 2003). Figure 5.7.3 below illustrates no marked shift over the 15 year period (1995 – 2010) in either the lowest or highest concentrations. This was not always the case, however. Low oxygen events were a common occurrence in the 1960s, but since the introduction of secondary wastewater treatment in the 70s, events of low oxygen (<5 mg L^{-1}) have been rare (Table 5.7.1 and Figure 5.7.3) (Cloern et al., 2003). The most notable historic example studied was a sewage spill that occurred near the mouth of Coyote Creek in the South Bay in 1979. For two weeks fishermen noted a lack of fish and pelagic invertebrates in Coyote Creek. Within several weeks of resuming treatment, at the San Jose/Santa Clara plant, both fish and invertebrates returned to the Creek (Cloern and Oremland, 1983). Presently, a reanalysis of DO data is being completed to further evaluate summer trends. Preliminary data analysis conducted by Alan Jassby and James Cloern suggests decreasing DO in bottom waters (>5 m) in South Bay, San Pablo Bay, and Suisun Bay since the early 1990s at an average rate of 1.5-2.5% per decade (note a much lower rate of change compared to chlorophyll a) (James Cloern Personal Communication, March 2011). These new analyses provide further evidence that the Bay is changing, perhaps motivating further interest to understand the effects of nutrient loads and other co-factors. While for DO the rate of change is low, the ramifications could be large and the rate may not be linear.

5.7.5 Utility of Dissolved Oxygen as an NNE Indicator for San Francisco Bay

Clear Linkage to Beneficial Uses

Dissolved oxygen has a clear and well established linkage to beneficial uses in the SF Bay. While 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), USEPA has an extensive database documenting adverse effects of low DO on a variety of fish and invertebrates with respect to juvenile and adult survival, reproduction and recruitment (EPA, 2003). Impacts of hypoxia on SF Bay pelagic and benthic organisms would have a direct impact on important SF Bay beneficial uses, including food web support for marine and estuarine aquatic organisms (EST, MAR) including the commercial and sport fisheries (COMM), shellfish such as clams, oysters and mussels (SHELL and AQUA), migratory (MIGR) birds and fish, support for fish nursery habitat (SPAWN). Poor water quality and increase heterotrophic bacterial production would adversely affect the health of recreational swimmers, sailboarders, and boaters (REF-1) and decrease aesthetic enjoyment of the Bay (REC-2) through nuisance buildup and smell during decay.

San Francisco Bay currently has a dissolved oxygen water quality objective (See section 3, Table 3.5) for maintaining organism/ecosystem health. However, the science supporting the selection of these regulatory thresholds have not been reviewed recently and it is likely that additional science is available that may shed light on whether revised DO objectives are needed for SF Bay. Currently, as part of the NNE framework for estuaries, a review of the science supporting dissolved oxygen objectives is currently being undertaken for California estuaries, exclusive of SF Bay. A review of this nature could be beneficial for SF Bay, including applicability of DO to diked Bayland and salt pond habitat.

Predictive Relationships to Causal Factors

Reduced dissolved oxygen is a measurable indirect impact of high nutrients and high primary productivity under certain conditions. Reduced dissolved oxygen can occur under conditions of water column stratification, high nutrients, ample sunlight, and high primary production and subsequent decomposition by heterotrophs. Modeling these conditions using empirical SF Bay data could help predict when and if reduced/low dissolved oxygen responses could occur. But to-date, that we are aware of, there has been no models developed to predict the spatial and short or long term temporal characteristics of DO in relation to external and internal causative factors. Open-source dynamic simulation models exist to predict dissolved oxygen concentrations from nutrient and organic matter loading and other co-factors.

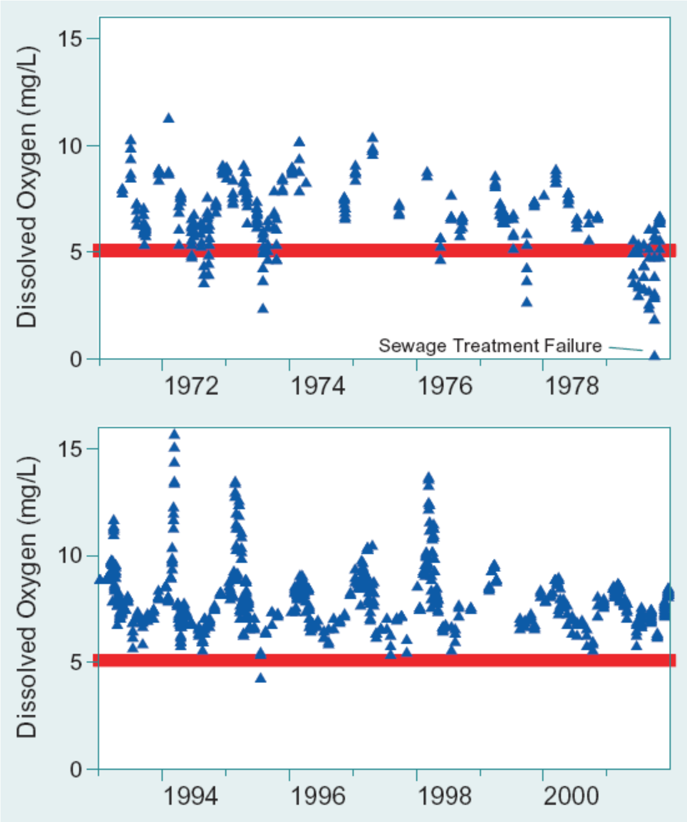

Figure 5.7.3. Dissolved oxygen concentrations in bottom waters of the South Bay. Red line shows the San Francisco Bay water quality objective for dissolved oxygen. Figure directly excerpted from Cloern et al., 2003.

Sound and Practical Measurement

Dissolved oxygen has a well-established and practical means of measurement. In addition, there is a long term dissolved oxygen data set available for SF Bay. USGS collects monthly measurement of dissolved oxygen along the spine of the Bay. RMP collects measurements of dissolved oxygen once a year in the summer according to a randomized study design. Both data sets provide a measure of the spatial variability in dissolved oxygen. USGS data can be used to indicate short-term temporal dissolved oxygen variability within a year as well as long term trends while RMP data can be used to indicate long-term trends and spatial variation of the Bay axis. There is ongoing monthly monitoring of the Bay for multiple water quality indicators including dissolved oxygen and chlorophyll a which, in combination, could be useful indicators.

Acceptable Signal to Noise Ratio

As with phytoplankton, dissolved oxygen concentrations in SF Bay have a high degree of spatial and temporal variability. However, DO is a well-vetted indicator of eutrophication and there is considerable experience with its use in a regulatory context to manage eutrophication. Due to its ease of measurement and the existence of a long term data set spanning 39 years, and an advance understanding of and measurement of phytoplankton blooms in the Spring and autumn, there is very high potential for development of DO ad an indicator of Bay health. The existence of a water quality guideline and clear relationships to beneficial use impairment provides a strong opportunity to measure trends either towards increasing or/and decreasing eutrophication with a very high signal: noise ratio.

5.7.6 Summary: Use of Dissolved Oxygen as an NNE Indicator for San Francisco Bay

Dissolved oxygen satisfies all four evaluation criteria and is recommended as an NNE indicator for the subtidal habitats of SF Bay. Existing basin plan objectives for dissolved oxygen exist for SF Bay. This review focused exclusively on summarizing status and trends of dissolved oxygen along the longitudinal axis of the Bay. Existing DO data available for tidally muted habitats (managed ponds) or in shallow areas of the Bay were not summarized in the review.

The SF Bay TAT did not have the expertise or budget to review the adequacy of dissolved oxygen objectives for SF Bay. A helpful framework to consider the scientific basis for dissolved oxygen objectives is the the USEPA Virginia Province Salt Water Dissolved Oxygen Criteria approach (USEPA, 2003). The fundamental goal of this approach is to maintain and support aquatic life communities and their designated uses. The approach was developed for the region of the east coast of the US from Cape Cod, MA, to Cape Hatteras, NC. This approach has been adapted for use in setting DO criteria for Chesapeake Bay (Batiuk et al., 2009), and has been applied, with appropriate modification, the other coastal regions of the US including Maine and Alabama. A review of the science supporting dissolved oxygen objectives in”other” California estuaries is onging, utilizing the EPA Virgina Province approach. If desired a similar review could be undertaken for SF Bay in its entirety or for selected habitat types such as tidally muted areas.

5.8 Ammonium and Urea Concentrations

San Francisco Bay is a nutrient-enriched estuary (Peterson, 1985a; Alpine and Cloern, 1988; Cloern et al., 2007). While primary productivity has been trending upwards for the past decade and the incidence of short lived harmful algae blooms appear to be on the increase, the Bay has not seen impacts from high nutrients like other eutrophic systems and for the last 30 years has been classified as a high nutrient, low primary productivity environment (e.g., Chesapeake Bay, Gulf of Mexico) (Cloern et al., 2007; Dugdale et al., 2007).

The primary pathways of nitrate and ammonium loads to the Bay are freshwater inflow (Sacramento/San Joaquin River and smaller tributaries), atmospheric deposition (Anderson et al., 2002), oceanic tidal exchange, and wastewater effluent (Conomos et al., 1979; Hanger and Shemel, 1992; Wankel et al., 2006). Nitrogen sources to the North Bay are primarily anthropogenic and include agricultural runoff as a main source in addition to nutrient loads from the Sacramento Regional Wastewater Treatment Facility (Sac Regional) (Hanger and Shemel, 1992; Wankel et al., 2006; Dugdale et al., 2007). Urea dominantly enters coastal ecosystems through the anthropogenic sources of stormwater and wastewater discharge (Glibert et al., 2006). The main sources of urea in these two types of discharge is uric acid from animal and human sources, direct application of urea as a fertilizer, production during industrial process (see review by Glibert et al., 2006, p447). Urea can also be found in atmospheric deposition and concentrations that can be similar to nitrate. While external loads typically exceed *in situ* production, urea is also known to be produced through excretion from zooplankton, bacterial regeneration, and release from bottom sediments (see review in Glibert et al., 2006, p446). Based on measurements in few coastal systems, rates of in situ urea production appear to be lower than ammonium production and too low to explain concentrations found in the water column suggesting that external sources are dominant (see review in Glibert et al., 2006, p446).

Inclusion of ammonium and urea concentration in the list of candidate NNE indicators represents a slight departure from the NNE framework, in which diagnosis of adverse effects is made on the ecological response to nutrients. The reason for this exception is based on recent evidence that ammonium concentrations may be responsible for limiting the spring diatom blooms in the North Bay and the lower Sacramento River (Wilkerson et al., 2007, Dugdale et al., 2006). In addition, urea has been proposed as one important factor that favors the dominance of certain species of HABs. This work is reviewed here in order to determine whether it is appropriate to include either ammonium and/or urea as NNE indicators in SF Bay.

5.8.1 Available Data on Nitrate+Nitrite, Ammonium, and Urea in the San Francisco Bay

The USGS has been collecting water quality data in SF Bay on nitrite, nitrate, ammonium and other nutrients for 39 years beginning 1968. Their research program has included many measurements of water quality from a monthly ship cruse at 39 fixed locations along the spine of the Bay. Since the USGS sample-collection was driven by research questions, it has not always been as regular or systematic as would occur in a monitoring program. For example, the USGS stopped sampling completely in 1981 after the spring bloom and didn't sample in the North Bay from about 1980-1987. That said, the database of these measurements presently includes over 9,000 discrete laboratory measurements of nitrate + nitrite (NOx) and over 8500 discrete laboratory measurements of ammonium (NH_4^-). There are isolated measurements of urea in Bay waters collected during specific studies (Cochlan and Herndon, unpublished data (cited in Kudela et al., 2008, p108); Herndon et al., 2003; Kudela et al., 2008). In addition, monthly concentrations of urea, ammonium and nitrate data are being collected at a series of sites in SF Bay as part of the National Estuarine Research Reserve (see comment by Kudela et al., 2008, p108). We have not reviewed this data.

5.8.2 Indicator Trends and Factors Affecting Temporal and Spatial Variation

Nutrients vary over multiple temporal and spatial scales. Temporal scales of variability include daily (flux of nutrients with daily photosynthetic activity and tidal exchange), seasonal (influx in winter freshwater flows and flux from phytoplankton blooms/decay), and interannual (long-term variability due to changes in watershed management/use). Spatial variability includes vertical (vertical flux in the water column) and horizontal (freshwater/wastewater effluent inflow, tidal exchange) scales.

Nitrogen and phosphorus are required nutrients for growth of photosynthetic organisms such as unicellular phytoplankton. Nitrate (NO_3^-) and ammonium (NH_4^-) are the two primary dissolved inorganic nitrogen forms in aquatic systems that are assimilated by primary producers (Peterson, 1985). Ammonium is the preferred nitrogen source for assimilation by phytoplankton and can become limiting in the environment (Peterson, 1985b; Hogue et al., 2001; Hogue et al., 2005; Wankel et al., 2006). However, in SF Bay nitrate is generally in high supply and phytoplankton can switch to nitrate assimilation when ammonium is depleted (Hogue et al., 2001). There is also evidence in North Bay (Suisun Bay and lower Delta) studies that ammonium has an inhibitory effect on phytoplankton nitrate uptake when ammonium concentrations exceed 4 µmol L^{-1} (Wilkerson et al., 2006; Dugdale et al., 2007). During springtime blooms in the North Bay there is an initial ammonium uptake by phytoplankton, subsequent ammonium depletion below 4 µM, followed by high rates of nitrate uptake resulting in a bloom period (Wilkerson et al., 2006). This inhibitory effect may prevent phytoplankton blooms from occurring since phytoplankton uptake of ammonium occurs at slower rates than nitrate uptake thus limiting rates of primary productivity (Dugdale et al., 2007). Dugdale et al. suggest that ammonium inhibition could be one of the limiting factors that control primary productivity in the Bay and contributes to the Bay's low primary productivity. Since, the greatest rates of production can occur when ammonium is low and nitrate is high, the concept of an ammonium:nitrate ratio as a key indicator have been advanced (Wilkerson et al., 2006; Dugdale et al., 2007).

In addition, urea has also been identified as a nutrient source taken up by phytoplankton (Gilbert et al., 2001; Anderson et al., 2002; Kudela et al., 2008). Loadings of urea have increased in certain areas, primarily due to the increased use of urea-based fertilizers (as reviewed in Anderson et al., 2002; Gilbert et al., 2006). There is some evidence that certain phytoplankton species, mostly flagellates, prefer uptake of urea over other nitrogen forms and that urea can increase the toxicity of a bloom (as reviewed in Anderson et al., 2002 and Kudela et al., 2008). Some of these flagellates have been identified as harmful or nuisance species (Gilbert et al., 2001; Anderson et al., 2002; Gilbert et al., 2006; Kudela et al., 2008). Experiments using local harmful algal species (coastal California and Bay species) showed some preferential uptake of urea when ambient nutrient concentrations were low (Kudela et al., 2008). It has also been suggested that urea may maintain harmful blooms (Kudela et al., 2008; Gilbert et al., 2001). In Chesapeake Bay, high urea concentrations were measured prior to a spring HAB bloom (Gilbert et al., 2001). The unusually high urea levels were correlated with high springtime precipitation that may have increased urea loading prior to the bloom. Urea has been measured in Bay waters but it is unclear how urea may be stimulating and/or maintaining phytoplankton blooms, particularly HABs, in the Bay (Herndon et al., 2003; as reviewed in Wilkerson et al., 2006). However, there is some evidence that urea may have an inhibitory effect on nitrate uptake by phytoplankton (as reviewed in Kudela et al., 2008). Given urea use as fertilizer continues to rise, urea will likely to continue to form a greater portion of the dissolved organic nitrogen (DON) pool available for primary production in coastal ecosystems (Glibert et al., 2006). Quantification of the standing stock of DON and the processes of assimilation remain an under studied area of research.

Spatial Variability

Nutrient spatial variability is explained by the balance of nutrient inputs (source), transport, assimilation by primary producers (sink), and re-mineralization/vertical flux (regeneration) (Grenz at al., 2000). In the North Bay, there is a decreasing gradient in nitrate and ammonium concentrations from the fresher Suisun Bay to the more saline Central Bay (Hogue et al., 2001; Wilkerson et al., 2006). The Lower South Bay has the highest concentrations of NO_x due to the influence of wastewater effluent (Table 5.Y) (USGS data; Wankel et al., 2006). High ammonium levels are also associated with wastewater effluent and are more equally distributed around the Bay (Table 5.8.1). Ammonium and NO_x can be drawn down during spring phytoplankton blooms in all Bay segments (Hogue et al., 2001; Wilkerson et al., 2006). There are limited measurements of urea in the Bay. Data have been primarily collected from North Bay locations so spatial variation is not currently known.

Table 5.8.1. Average nitrate + nitrite and ammonium concentrations in each Bay segment 1999 – present (Source: J. Cloern, USGS: http://sfbay.wr.usgs.gov/access/wqdata). Data for urea-N are from short term studies by various authors as indicated.

Segment	(NO_x) Nitrate + Nitrite-N (mg L^{-1}) Average	Ammonium-N (mg L^{-1}) Average	Urea-N (mg L^{-1})	
			Location, date	**Average (Range)**
Suisun	0.38	0.11	[1]"North Bay", unknown date	>0.024
Carquinez Straight	0.36	0.11		
San Pablo Bay	0.32	0.09		
Central Bay	0.26	0.09	[2]Richardson Bay, Jun, Jul, Sep 2002	0.0090 (0.0063-0.0137)
			[3]Western Richardson Bay and Paradise Cay, May-Oct, 2005	(0.00056 – 0.0051)
South Bay	0.35	0.08		
Lower South Bay	0.70	0.09		

[1] Cochlan and Herndon, unpublished data (cited in Kudela et al., 2008).

[2] Herndon et al., 2003.

[3] Kudela et al., 2008.

Temporal Variability

Temporal variability of dissolved nitrate and ammonium are controlled by river inflow, inflow of wastewater treatment plant effluent, photosynthetic drawdown, and tidal exchange with nutrient rich coastal waters (Conomos et al., 1979; Cloern and Nichols, 1985; Peterson, 1985a; Cloern et al., 2001). In the North Bay, there is a temporal dynamic of nutrient concentrations (interplay between source and sink) that is dependent on winter freshwater flow. Winter periods are generally characterized by low primary productivity and high nutrient concentrations while during spring blooms, primary productivity dominates nutrient dynamics (Figure 5.8.1 and 5.8.2) (Peterson, 1985a; Wilkerson et al., 2006; Dugdale at al., 2007). There is much interannual variation in this dynamic depending on the magnitude of winter river flow, the duration of high river flow into the spring/summer, and the magnitude of the spring phytoplankton bloom. Nitrate and ammonium concentrations can be drawn down during phytoplankton blooms (mostly occurring during spring) due to removal by primary producers (Peterson, 1985a; Cloern, 1996; Hogue et al., 2001; Wilkerson et al., 2006; Cloern et al., 2007). In the South Bay there is interannual variation of nitrate and ammonium during springtime dependent on timing of the spring phytoplankton bloom (Cloern, 1996) however, there is presently no peer-reviewed articles that describe any influence ammonium inhibiting nitrate uptake in the South Bay.

Ocean dynamics also contributes to the temporal variation in nutrients, especially for nitrate (Peterson, 1985a). The coastal upwelling maximum occurs in June which can result in high nitrate concentrations in coastal waters outside the Golden Gate which then enter the Bay via incoming tides. Upwelling variation occurs on a longer-term timescale known as the Pacific Decadal Oscillation (PDO). The California coast is currently in a cold phase of the PDO which results in stronger upwelling bringing higher nitrate concentrations to surface coastal waters (Cloern et al., 2007).

There is also temporal variation in the ratio of NH_4^- and NO_x (NO_3^- + NO_2^-). This ratio may be used as an indicator of an ammonium inhibited environment. The ratio generally remains below 1.0 most months except for two peak periods in the South Bay (April and September) (Figure 5.8.3). This peak in ammonium concentrations, relative to NO_x, coincides with the spring and fall phytoplankton blooms. This may indicate that ammonium is not limiting primary productivity in the southern portions of the Bay. There are limited measurements of urea in the Bay. Data that have been collected show variability in urea concentrations with urea representing between 3 and 42% of total nitrogen (Kudela et al., 2008). However, measurements were only taken between May and October so overall seasonal variation is not known.

Indicator Trends

There are long-term data sets for nitrate and ammonium in SF Bay (USGS and RMP data). Long-term trend analysis of nutrients is sparse in the literature (Figure 5.8.4, 5.8.5, and 5.8.6). In South SF Bay, daily nitrogen loads from the San Jose – Santa Clara wastewater treatment plant have declined since the late 1970s (Cloern et al., 2007). Longer-term trends in the South Bay (1990 – 2005) suggest that dissolved inorganic nitrogen levels are decreasing up to a maximum of 10% per year (Cloern et al., 2007). There are no long-term data on urea concentrations in the Bay. However, there have been documented increases of urea loadings to other estuaries due to increased usage of urea-based fertilizers.

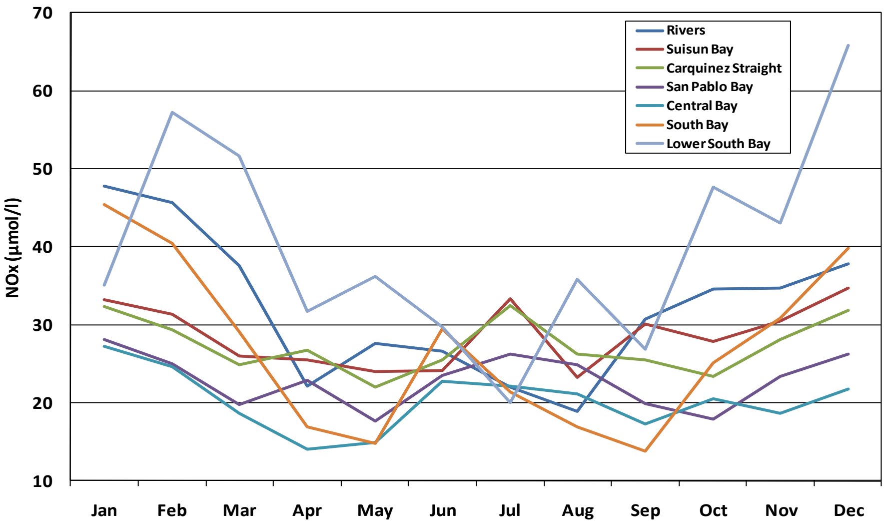

Figure 5.8.1. Monthly averaged nitrate + nitrite (NOx) concentrations in the defined segments of the San Francisco Bay for 1999 – 2010. Data obtained from the USGS: http://sfbay.wr.usgs.gov/access/wqdata).

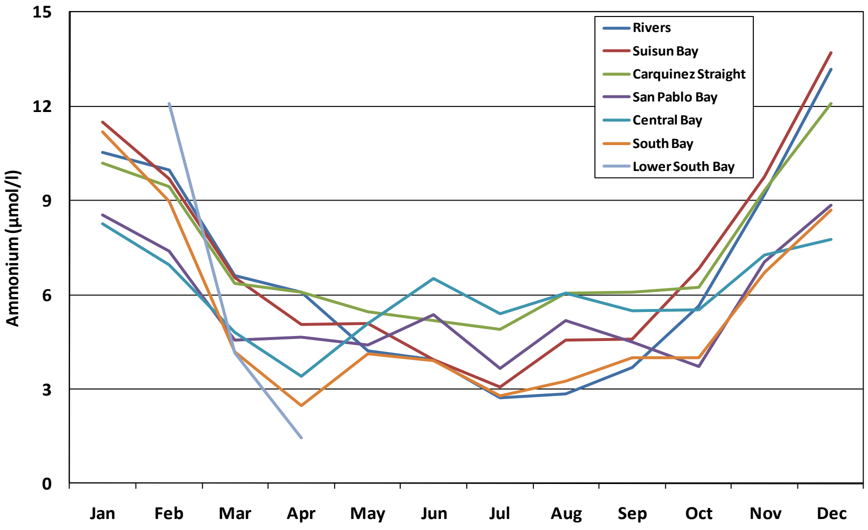

Figure 5.8.2. Monthly averaged ammonium concentrations in the defined segments of the San Francisco Bay for 1999 – 2010. Data obtained from the USGS: http://sfbay.wr.usgs.gov/access/wqdata).

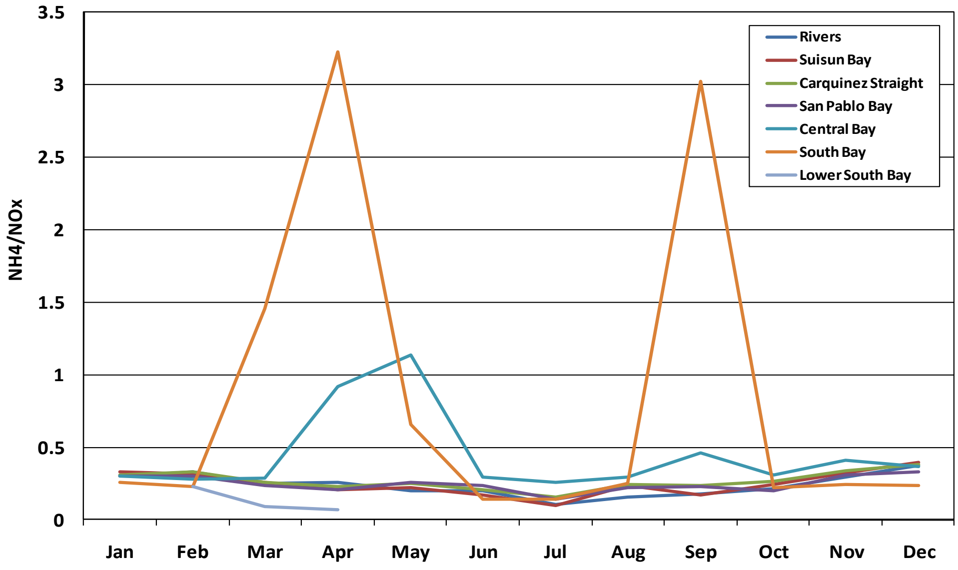

Figure 5.8.3. Monthly averaged ratios of ammonium to nitrate + nitrite in the defined segments of San Francisco Bay 1999 - 2010. Data from the USGS: http://sfbay.wr.usgs.gov/access/wqdata).

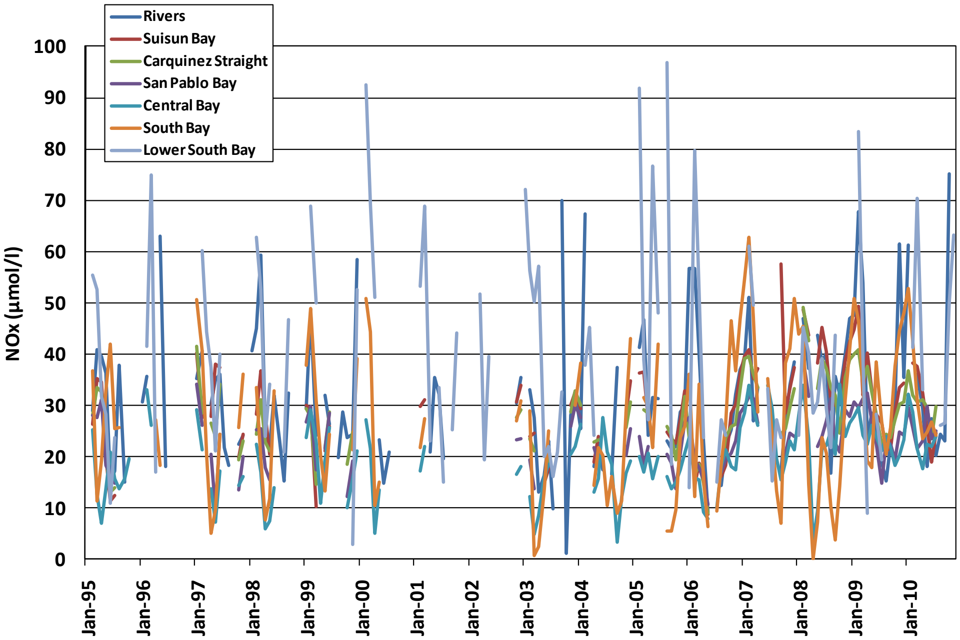

Figure 5.8.4. Monthly averaged nitrate + nitrite (NOx) in the defined segments of San Francisco Bay 1995 – 2010. Data from the USGS: http://sfbay.wr.usgs.gov/access/wqdata).

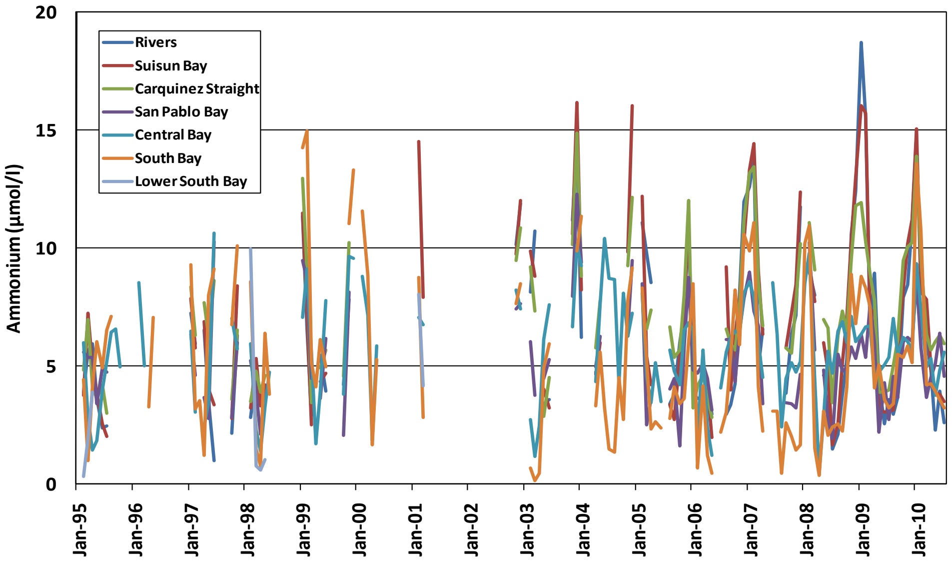

Figure 5.8.5. Monthly averaged ammonium in the defined segments of San Francisco Bay 1995 – 2010. Data from the USGS: http://sfbay.wr.usgs.gov/access/wqdata).

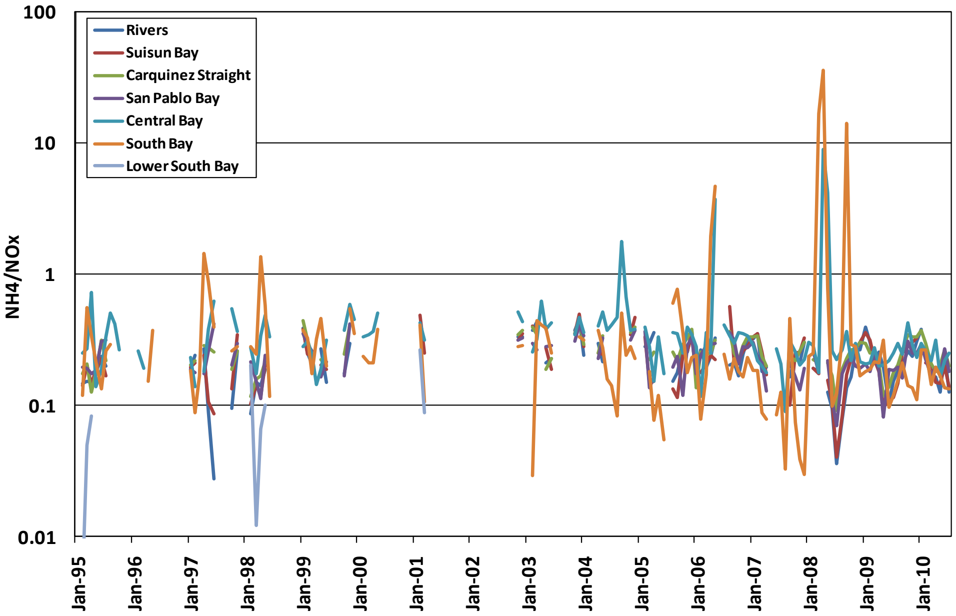

Figure 5.8.6. Monthly averaged ratios of ammonium to nitrate + nitrite in the defined segments of San Francisco Bay 1995 – 2010. Data from the USGS: http://sfbay.wr.usgs.gov/access/wqdata).

5.8.3 Utility of Indicator as an Eutrophication Indicator for San Francisco Bay

Clear Linkage to Beneficial Uses

Several studies (cited in Section 5.2) provide a broad base of evidence that phytoplankton have a direct linkage to important SF Bay beneficial uses, including food web support for marine and estuarine aquatic organisms (EST, MAR) including the commercial and sport fisheries (COMM), shellfish such as clams, oysters and mussels (SHELL and AQUA), migratory (MIGR) birds and fish, support for fish nursery habitat (SPAWN). Increased incidence of HABs can adversely affect the health of humans (REC-1) by irritation and injury to recreational swimmers, sailboarders, and boaters (Lehman et al., 2005). Thus adverse effects to phytoplankton primary production and the occurrence of dominant assemblages known to support Bay ecosystem services (e.g., fisheries) would be considered to adverse effecting Bay beneficial uses.

Ammonium inhibition of nitrate uptake by diatoms has been documented in several regions of the Bay and ammonium concentrations above 4 μM have been suggested as a major mechanism by which spring diatom blooms appear to be suppressed in the North Bay and lower Sacramento River (Wilkerson et al., 2006; Dugdale et al., 2007). Despite this evidence, the ecological importance of ammonium inhibition of spring diatoms blooms is not well understood relative to factors known to control primary productivity, particularly in other regions of the Bay where water column chlorophyll a appears to be increasing.

It has been suggested that HABs in the North Bay may be supported by regenerated ammonium in the summer and autumn (North Bay: Dugdale et al., 2007; South Bay: Thompson et al., 2008) and that HAB species that utilize ammonium as a nitrogen source can produce greater toxicity (see review by Glibert et al., 2006 and in Kudela et al., 2008). Likewise, some HAB organisms that can preferentially uptake urea may have a competitive advantage (Glibert et al., 2006). However, as with ammonium inhibition of diatom nitrate assimilation, the influence of ammonium on HABs in SF Bay has not been sufficiently investigated. Thus the linkage between ammonium concentrations and Bay beneficial uses is not at this time universally accepted. SF Bay TAT members agree that additional data synthesis is required to better understand the role of ammonium in SF Bay.

Kudela et al. (2008) noted that some HAB species can show a preference for urea versus other inorganic nitrogen constituents (e.g., *A. sanguinea*) or similar affinities for urea versus other nitrogen substrates (*L. polyedrum*), which under conditions of high urea concentrations could favor these HAB species. However, the importance of urea in promoting HABs in SF Bay is not understood, in part because measurement of urea is not part of a routine research programs conducted in the Bay and the focus of HAB research has been largely restricted to cyanobacteria. Thus the linkage between urea and adverse effects on the phytoplankton community and Bay beneficial uses are not well documented.

Predictive Relationships to Causal Factors

Conceptual models have shown a complex relationship between nutrients and primary productivity in SF Bay. There are multiple factors that have limited primary productivity in the Bay including light limitation, strong tidal/wind forcing that prevents long periods of stratification, and recent evidence that ammonium concentrations may be limiting the magnitude of spring blooms in the North Bay and that nitrate limitation might be an important factor on some occasions to causing bloom termination in the South Bay (Thompson et al., 2008). An added challenge in developing linkages between nutrient loads, standing concentrations, and productivity response is that these are often out of phase in our dry summer Mediterranean climate (Kudela et al., 2008). Empirical modeling could be beneficial to better understand relationships and predictive power between nutrients and impacts to beneficial uses. Existing models for the North Bay have not yet utilized recent empirical observations of the role of ammonium in bloom formation, and ammonium thresholds in relation to nitrate uptake rates (Dugdale et al., 2007). The impact of ammonium on nitrate uptake rates may also extend into San Pablo and Central Bays as advected ammonium that is not taken up in Suisun Bay due to unfavorable conditions and clam grazing appears to help maintain ammonium concentrations that do not allow nitrate uptake (Dugdale et al., 2007). However, as stated previously, the effect of ammonium inhibition on phytoplankton productivity throughout the Bay has not been modeled vis-à-vis other controlling factors.

There is growing evidence that DON, and in particular urea, is able to support greater population of HAB species. However at this time a causative linkage has not been established (Glibert et al., 2006); “while there is clear evidence that HABs can utilize multiple sources of nitrogen, this is not a unique characteristic and does not imply that urea selects for HAB species” (Kudela et al., 2008).

Given these data gaps, the logical next step is to the develop models that synthesize understanding of the relative importance of ammonium and urea versus other factors in controlling phytoplankton assemblages.

Sound and Practical Measurement

The laboratory methods for analysis of nitrate, nitrite, ammonium, and urea in natural waters are well established. However care should be taken to ensure reporting limits are sufficient to provide enumeration throughout all areas of the Bay during all periods of the annual cycle. There are long-term data sets on ammonium and nitrate levels in the Bay that are ongoing through the USGS and the RMP. During the peak of algal blooms, concentrations can be low and bloom termination can sometimes result from very low nitrate concentrations. Therefore, in order to study these phenomenon, using laboratories and methods that are able to report concentrations when they occur at $<10\mu g\ L^{-1}$ should be maintained. Data on urea concentrations are very limited but Kudela et al. (2008) proposed that this is an important data gap that if filled will provide an additional basis for improving our understanding of anthropogenic nutrient loadings and relationship to HAB species. In order to get useful information, monitoring programs should aim for reporting limits of $<1\mu g\ L^{-1}$ for urea.

Acceptable Signal to Noise Ratio

Since much of the nitrate and ammonium present in SF Bay has been cycled many times through the processes of assimilation and regeneration, there will be challenges with using either of these two nutrients or a ratio between the two as an indicator for changing Bay conditions. It seems that a more complex indicator model would give us a better understanding of the dynamics that lead to WQ impacts (low DO, HABs) and give us better predictive power on when/where these events might occur. The ratio of NH_4:NO_3 may be a good indicator if it can be developed to predict ammonium limitation, bloom initiation, or bloom termination. Urea, other hand perhaps holds great promise as the presence of urea in the Bay likely directly correlated with external anthropogenic loads since proportionally much less urea is likely derived from within the Bay through *in situ* generation (Kudela et al., 2008). However at this time the science of urea and HABs is in its infancy and needs further development to increase the signal to noise ratio especially for SF Bay.

5.8.4 Summary: Use of Ammonium and/or Urea as an NNE Indicator for San Francisco Bay

Elevated ammonium concentrations have been suggested as a major mechanism by which spring diatom blooms appear to be suppressed in the North Bay and lower Sacramento River (Wilkerson et al., 2006; Dugdale et al., 2007). Despite this evidence, the ecological importance of ammonium inhibition of spring diatoms blooms is not well understood relative to factors known to control primary productivity, particularly in other regions of the Bay where water column chlorophyll a appears to be increasing. Thus the linkage between ammonium concentrations and Bay beneficial uses is not at this time universally accepted. San Francisco Bay TAT members agree that additional data synthesis is required to better understand the role of ammonium in SF Bay. The SF Bay TAT recommends that a broader review of the subject, with the intent of synthesizing expert opinion on factors that influence the importance of this phenomenon in different regions of SF Bay and identifying critical data gaps.

While there is growing evidence that urea has a role in support an increased frequency of certain HAB species, a causative linkage has not been established (Glibert et al., 2006) and it is not possible to state that urea selects for HAB species (Kudela et al., 2008). Moreover little data are available on urea concentrations in SF. Therefore, recommended next steps are to begin collecting urea data in USGS SF Bay research cruises events and to undertake a broader review of the effects of urea on phytoplankton community composition, with the intent of identifying critical data gaps for its use in the NNE.

6.0 Synthesis, Data Gaps, and Recommendations

The ultimate goal of NNE program in SF Bay is to develop a diagnostic assessment framework to determine when SF Bay is experiencing adverse effects of nutrient and models to link those effects with nutrient loading and other controls on the estuary.

As a step towards this goal, this review had several objectives:

- Evaluate appropriate indicators to assess eutrophication and other adverse effects of anthropogenic nutrient loading in SF Bay
- Summarize existing literature and identify data gaps on the status of eutrophication in SF Bay with respect to these indicators
- Investigate what data and tools exist to evaluate the trends in nutrient loading to the Bay and, summarize, to the extent possible, what do they reveal about trends in nutrient loads over time?

This section summarizes recommends specific indicators, based on explicit indicator evaluation criteria, summarizes current data available in the Bay for those indicators, identifies data gaps, and recommended next steps.

6.1 Which Indicators Met Review Criteria?

The review found a suite of indicators met all evaluation criteria, while others met three of four criteria (Table 6.1). Selected indicators for the SF Bay NNE framework vary by habitat type. For the purposes of developing an NNE, SF Bay can be separated into four main habitat types, the first of which is tidally muted, while the remaining three habitat types are found in areas of the Bay which are not subject to anthropogenic tidal muting.

- Muted tidal areas (i.e., possessing tidal hydrology that is restricted by impoundments, dikes, or weirs such as estuarine diked Baylands, managed ponds (e.g., former salt ponds)

- Unvegetated intertidal habitat (intertidal flats)

- Unvegetated subtidal habitat

- Seagrass habitat

Unvegetated intertidal habitat represents 9.8% of SF Bay (~1200 km^2). Unvegetated intertidal habitat is currently believed to be dominated by benthic microalgae (microphytobenthos) and, to a lesser extent, macroalgae. For such habitat types, Sutula (2011) propose the use of macroalgae to assess eutrophication for the NNE. However, there is a general lack of data on the spatial and temporal variation in biomass, cover, and taxonomic composition of microphytobenthos and macroalgae in SF

Bay. The SF Bay TAT recommends additional data collection establish a baseline of information that can be used to further consider the establishment of a NNE for intertidal flats.

Table 6.1. Summary of review of candidate NNE indicators for San Francisco Bay.

Indicator	Applicable Habitat Type	Met Review Criteria	Comments
Dissolved oxygen	All subtidal	Yes	Wealth of data, existing objectives in SF Water Board basin plan
Phytoplankton biomass, and productivity, and assemblage	All subtidal	Yes	Wealth of data and reliable if used together in multiple lines of evidence. lack of experience predicting phytoplankton assemblages
Harmful algal bloom cell counts and toxin concentration	All subtidal	Three of four	Limited data and understanding of controls on HAB occurrence and toxin production, ***with exception of cyanobacteria***
Ammonium	All subtidal	Three of Four	Evidence of ammonium inhibition of diatom nitrate assimilation, but ecological importance of process not well understood for entire Bay.
Urea	All subtidal	Three of Four	Causative link between urea and HABs has not been established and little data are available on urea concentrations in SF Bay.
Light attenuation	Seagrass, All subtidal	Two of Four	Phytoplankton biomass a component of light attenuation, but other factors unassociated with eutrophication can contribute (e.g., suspended sediment), so poor linkage to BUs and signal:noise
Macroalgae biomass and cover	Seagrass, intertidal flats, muted subtidal	Yes	Insufficient data on macroalgal blooms on intertidal flats and seagrass habitat.
Epiphyte load	Seagrass habitat	Three of four	Epiphyte load difficult to quantify. Use in conjunction with macroalgal biomass/cover and phytoplankton biomass.
Macrobenthos taxonomy, abundance and biomass	Subtidal	Three of four	Macrobenthos not uniquely affected by eutrophication; lack of experience predicting community measures as a function of nutrient loads. Additional research required. May be used as a supporting indicator or co-factor in so much as it could part of a routine sediment quality ambient monitoring

6.2 Recommended Primary, Supporting Indicators to Develop a NNE Diagnostic Assessment Framework for San Francisco Bay

Within the regulatory context, waterbody assessments are made in order to make a determination of whether the waterbody is meeting beneficial uses or impaired, as an example, for nutrients or to set TMDL numeric targets. In this context, a diagnostic assessment framework is the structured set of decision rules and guidance for interpretation that helps to classify the waterbody in categories of minimally disturbed, to moderately disturbed, to very disturbed. Although scientists can provide guidance and data synthesis to illustrate how the assessment framework could be formed, ultimately the decision of what levels to set thresholds (i.e., NNEs) that separate the categories (e.g., minimally versus moderately and very disturbed) is a policy decision.

Development of the diagnostic assessment framework begins by choosing indicators that would be measured and used to determine waterbody status. It is important to distinguish between three types of indicators for an NNE assessment framework: 1) Primary indicators, 2) Supporting indicators, and 3) Co-factor indicators required for data interpretation.

Primary indicators are those for which regulatory endpoints could be developed. Designation of these indicators as "primary" implies a higher level of confidence in these indicators to be used to make an assessment of adverse effects, based on a wealth of experience and knowledge about how this indicator captures and represents ecological response. Primary indicators are those which met all explicit criteria (see Section 2.5) established to evaluate candidate NNE indicators. Supporting indicators are those which could be collected to provide supporting lines of evidence, but would not be used alone to make determination of whether the waterbody was meeting beneficial uses. These indicators may have met many, though not all evaluation criteria, but are considered important because they are commonly used to assess eutrophication. Use of the indicator as supporting evidence over time may increase confidence and cause it to be promoted to "primary." Finally, co-factors are indicators that could be part of a routine monitoring program and important for data interpretation and trends analysis.

The SF Bay TAT has recommended a suite of "primary" (those for which regulatory endpoints would be developed), "supporting" (those for which no regulatory endpoints would be developed in the near term, but could be used as supporting lines of evidence) and "co-factors" (data that would be used for interpretation of primary and supporting indicators) for four major habitat types in SF Bay. The real distinction between "primary" and "supporting" and how these classes of indicators would be used as multiple lines of evidence in an NNE assessment is entirely dependent on indicator group and particular applications to specific habitat types.

San Francisco Bay is dominated by unvegetated subtidal habitat. In this habitat type, phytoplankton dominate primary production. Therefore, the primary NNE indicators to assess eutrophication in SF Bay are measures of phytoplankton (biomass, productivity, taxonomic composition and abundance), algal toxin concentration and dissolved oxygen. In the designation of phytoplankton indicators (biomass, productivity, and assemblage) as primary, the SF Bay TAT stresses the importance of using these indicators together in as multiple lines of evidence, as use of any one alone is likely to be insufficiently robust. We use the term algal here to remain broad and acknowledge that cyanobacteria do not constitute a significant percentage of the phytoplankton community within the geographic scope of the review. However, there is an expressed intent to capture potential adverse effects of cyanobacteria that may be transported downstream from the Delta as whole cells or as toxin.

Seagrass habitat found in subtidal and intertidal areas of the Bay are an important habitat, though currently a minor percentage of Bay habitats (approximately 1.3% of 1200 km^2). The primary NNE indicators for seagrass habitat represent a combination of factors that result in reduced light attenuation to the bed, resulting in reduced photosynthetic activity by the plants. Thus phytoplankton biomass, macroalgal biomass and percent cover and epiphyte load are the primary NNE indicators for this habitat type.

Muted intertidal and subtidal habitat also represent an important but minor habitat type, representing an additional 269 km^2 in addition to the 1200 km^2 of habitat subjected to full tidal action these areas include salt ponds, duck clubs, and diked Baylands, many of which are actively managed and undergoing active change through restoration. The primary NNE indicators include macroalgal biomass and cover, which are found in both intertidal and subtidal habitat, and measures of dissolved oxygen and phytoplankton (biomass, productivity, HAB species abundance and toxin concentration), which applies to subtidal habitat only.

6.3 Is San Francisco Bay Currently Experiencing Eutrophication Based on These Indicators?

Ultimately, the conclusion of whether SF Bay is adversely affected by the consequences of nutrient over enrichment is made by the SFRQCB and the SWRCB. This review does not attempt to assume that responsibility, but rather summarizes relevant information on the topic based on available data and peer-reviewed articles. The review utilized available data to summarize spatial and temporal trends in the candidate NNE indicators for the Bay (Table 6.3). Brief synopsis of the review is provided below for each of the three habitat types for NNE development is recommended (subtidal, seagrass and muted subtidal).

Table 6.3. Summary of available data sets with which to assess eutrophication. Unvegetated and vegetated (seagrass) subtidal habitats and intertidal habitats are not tidally muted.

Habitat Type	Phytoplankton	Dissolved Oxygen	Nutrients (including ammonium & urea)	Macroalgae & epiphytes
Un-vegetated subtidal habitat	40 years of water quality data from USGS research cruises with some gaps depending on research questions in any one year. Data include phytoplankton biomass, productivity, and taxonomic composition as well as DO and nutrients[6]. The IEP Environmental Monitoring Program (EMP) for the Sacramento-San Joaquin Delta, Suisun Bay, and San Pablo Bay has 40 years of data phytoplankton, zooplankton, and benthos, and water quality. Little data available on water column or tissue phytotoxin concentrations, with exception of cyanobacteria from IEP-EMP in North Bay. Little data available on urea concentrations.			N/A
Seagrass habitat	Limited data available through CALTRANS mitigation monitoring (ended in 2010)	No data available	No data available	Limited data from via recent SFSU monitoring
Intertidal habitat	Not applicable	Not applicable	No data available	No known data
Muted Tidal Habitat (managed ponds)	Limited data available through recent monitoring by USGS			No known data

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[6] Note that USGS started measuring chlorophyll in 1977 and DO has not been measured consistently (e.g., not in early 1980's). Nutrients are not measured on every cruise and there are big gaps.

Table 6.2. Table of proposed primary and supporting NNE indicators by habitat type. Primary indicators are those for which regulatory endpoints could be developed. Supporting indicators are those which could be collected to provide supporting lines of evidence. Co-factors are indicators that could be part of a routine monitoring program and important for data interpretation and trends analysis. The list of co-factor indicators is provided as an example and not exhaustive. Note that primary and supporting indicators recommended for unvegetated subtidal habitat are also applicable for seagrass habitat.

Habitat	**Primary Indicators**	**Supporting Indicators**	**Co-Factors**
All Subtidal Habitat	Phytoplankton biomass, productivity, and assemblage Cyanobacteria cell counts and toxin concentration Dissolved oxygen	Water column nutrient concentrations and forms[7] (C, N, P, and Si) HAB species cell count and toxin concentrations	Water column turbidity, pH, conductivity, temperature, light attenuation Macrobenthos taxonomic composition, abundance and biomass Sediment oxygen demand Zooplankton
Seagrass Habitat	Phytoplankton biomass Macroalgal biomass & cover Dissolved oxygen	Light attenuation, suspended sediment conc. Seagrass areal distribution and cover Epiphyte load	Water column pH, temperature, conductivity Water column nutrients
Intertidal Flats	Macroalgal biomass and cover	Sediment % OC, N, P and particle size Microphytobenthos biomass (benthic chl a)	Microphytobenthos taxonomic composition
Muted Intertidal and Subtidal	Macroalgal biomass & cover Phytoplankton biomass Cyanobacteria toxin concentration	Sediment % OC, N, P and particle size Phytoplankton assemblage Harmful algal bloom toxin concentration	Water column pH, turbidity, temperature, conductivity Water column nutrients

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[7] Forms referred to relative distribution of dissolved inorganic, dissolved organic, and particulate forms of nutrients, including urea

6.3.1 Phytoplankton, Dissolved Oxygen, and Nutrients in Subtidal Habitats

Cloern and Dugdale (2010) provide an excellent synthesis of the status of eutrophication in the subtidal habitats of SF Bay, based on available phytoplankton, dissolved oxygen and nutrient data in SF Bay. Our review largely restates their observations and conclusions.

San Francisco Bay is a nutrient-enriched estuary, receiving external loads of N and P comparable to Chesapeake Bay (Cloern and Dugdale, 2010). However, dissolved oxygen is much higher and phytoplankton biomass and productivity is lower than would be expected, implying that phytoplankton dynamics and ultimately eutrophication are driven by processes other than nutrient-limitation of primary production. However, all regions of the SF Bay system, from Suisun to South Bay, have experienced significant increases in phytoplankton biomass since the late 1990's (e.g., Cloern et al., 2007, 2010). Recent analysis of water quality data collected by USGS from 1978 to 2009 show of a significant increase in water column chlorophyll a per decade (30-50% per decade from Suisun to South Bay respectively) and a significant decline in DO concentrations (1.6 to 2.5% in South Bay and Suisun Bay respectively ; J. Cloern, personal communication March 2011). Thus evidence is building that the historic resilience of SF Bay to the harmful effects of nutrient enrichment is weakening.

A synthesis of research (Cloern and Dugdale, 2010) has pointed to a number of factors which can be controlling phytoplankton production in the Bay: 1) high turbidity that constrains phytoplankton productivity, 2) top-down control by benthic suspension feeders, 3) salinity stratification that promotes blooms by stationing cells in a high-light and high-nutrient surface layer and isolates them from benthic consumers, 4) connectivity to the Ocean which provides a source of phytoplankton cells that seed blooms, 5) the necessity of approx. 5 days of favorable irradiance for bloom initiation and interactions between diel light and semi-diurnal tidal cycles which control available light to sustain blooms, 6) currents that transport phytoplankton between habitats that function as a net source or sink of algal biomass, and 7) importance of high ammonium inputs in reducing the frequency and intensity of spring blooms through inhibition of nitrate uptake in the lower Sacramento River and Suisun Bay. Temporally, data show that phytoplankton productivity is higher and is increasing in Central Bay, San Pablo Bay and Suisun Bay since the mid 1990s. The causes for the Bay wide trends include changes in water clarity due to less suspended sediment (Schoellhamer, 2009), lower metal inhibition due to improvements in wastewater treatment, increased seeding from ocean populations (Cloern et al., 2005), declines in consumption by bivalves due to increases in predation by juvenile English sole and speckled sanddabs, and declines in phytoplankton consumption by bivalves and zooplankton due to recent new invasive species introductions (Cloern et al., 2006). Data suggest that primary productivity in Suisun Bay is limited by strong grazing pressure by the invasive clam Corbula amurensis (Alpine and Cloern 1992), light limitation by high turbidity (Cloern, 1999), and undetermined chemical inhibition, (Cloern and Dugdale 2010).

San Francisco Bay contains over 500 phytoplankton taxa. Diatoms (Bacillariophyta) dominate the biomass making up 81% of the total cumulative biomass; dinoflagellates and cryptophytes (Pyrrophyta and Cryptophyta) made up 11 and 5%, respectively (Cloern and Dufford, 2005). Despite the persistent nutrient enriched status of SF Bay, few HABs have been reported recently in SF Bay, apparently because nutrient enriched turbid conditions in the estuary favor diatoms associated with new inputs of nutrients as opposed to nutrient regeneration (Cloern, 1996; Ning et al., 2000). However, there have been occasional historical occurrences (see Cloern et al., 1994 referenced in Cloern, 1996), and recently cyanobacteria and dinoflagellate blooms have been documented.

Low oxygen events were a common occurrence in the 1960s, but since the introduction of secondary wastewater treatment in the 70s, events of low oxygen (<5 mg L^{-1}) have been rare. Currently, SF Bay is not experiencing periods of hypoxia and is currently meeting Basin Plan objectives for dissolved oxygen. Water column oxygen variability is controlled by density/thermal stratification and results in higher oxygen concentrations in the upper portions of the water column. Isohaline conditions prevail during summer due to strong wind and tidal forcing which results in a mixed water column with less vertical variability in oxygen levels (Conomos et al., 1979). Decreased dissolved oxygen generally occurs in the lower portions of the water column during periods of stratification when there is high availability of organic matter (Diaz, 2001). Adequate dissolved oxygen levels are most likely maintained due to a quick dissipation of blooms from physical forcing of the tides and winds mixing the water column. There is little information for SF Bay on dissolved oxygen concentrations during other HAB episodes. That said, it seems feasible that if the increased production trend continues or if the incidence of harmful algae blooms increases, there might be a concomitant increase in the frequency of lower DO events especially if they happen to coincide with neap tides or lower wind conditions which can lead to stratification. Recent analysis of DO data collected by USGS from 1978 to 2009 indicates that DO concentrations have significantly declined on the order of 1.6 to 2.5% per decade in South Bay and Suisun Bay respectively (J. Cloern, personal communication March 2011).

6.3.2 Phytoplankton, Macroalgae and Epiphytes in Seagrass Habitat

Very limited data exist on symptoms of eutrophication in seagrass habitats in SF Bay. Some beds have been documented to have persistent macroalgal biomass and cover, but data are inadequate to make an assessment of effects and it is unclear as the sources of nutrients responsible for maintaining macroalgal blooms in this area (K. Boyer, personal communication). There was consensus by SF Bay TAT members that seagrass is currently limited by turbidity associated with suspended sediments rather than a combination of phytoplankton biomass, macroalgae and/or epiphytes.

6.3.3 Macroalgae and Microphytobenthos in Intertidal Flats

No assessment of current status of eutrophication on intertidal flats can be made because of limited data.

6.3.4 Dissolved Oxygen, Macroalgae and Phytoplankton in Muted Intertidal and Subtidal Habitats

Data that exist on symptoms of eutrophication in muted subtidal habitats (such as estuarine diked Baylands and restored salt ponds) are a result of monitoring associated with restoration or special studies conducted by USGS. Utility and applicability of DO to diked Baylands requires additional discussion, particularly because muted habitats are to some degree subject naturally to hypoxia. In addition, these habitat types are known to influence subtidal DO. Low DO water can exist in salt ponds that, if breached, can supply high organic discharge or low DO discharge to the Bay (Shellenbarger et al., 2008; Thebault et al., 2008). Limited monitoring data of phytoplankton biomass and community composition of these habitats are insufficient to make an assessment of current status of eutrophication in these systems, although it is established that some managed ponds in lower South Bay harbor species of toxin-producing phytoplankton (Thebault et al., 2008). Data on macroalgal biomass and cover in these habitat types are not known to exist.

6.4 What Are the Nutrient Loads to San Francisco Bay From Various Sources and How Are These Loads Changing over Time?

San Francisco Bay is regarded as a nutrient enriched estuary, based on the ambient concentrations and estimated loads of nutrients to the Bay (Cloern and Dugdale, 2010). Nutrients loads from external sources and pathways are poorly understood, though data exist with which to improve published load estimates from some sources. For the most part, published load estimates are outdated by one or even two decades and were either based on data collection methods that were not designed for loads estimation, were based on assumptions that provided guesses about loads at best or were based on data sets that have now been substantially improved with ongoing collection through time. Given changes to wastewater treatment technologies, increases in population, changes to land use, etc., nutrient loads have likely changed over the past four decades.

6.5 Data Gaps and Recommended Next Steps

The SF Bay NNE framework consists of two principle components: 1) primary and supporting indicators used in an assessment framework to assess eutrophication of SF Bay habitats and 2) models that link these indicators back to nutrient loads and other controlling factors that mitigate the ecological response to eutrophication. A set of data gaps and recommended next steps are recognized for both of these components of the SF Bay NNE framework.

The development and use of an NNE framework for SF Bay is completely contingent on the continued availability of monitoring data to formulate, test and periodically assess the status of the Bay with respect to eutrophication. Over the past forty years, the USGS has conducted a research program in the subtidal habitat of SF Bay, with partial support by the SF Bay Regional Monitoring Program (RMP). This USGS research program cannot be considered replacement for a regularly funded monitoring program. **The SF Bay TAT strongly recommends that a nutrients/eutrophication monitoring strategy be developed and funded for successful development and implementation of the NNE in SF Bay. This program should be coordinated and complementary to the IEP Environmental Monitoring Program that terminates in San Pablo Bay. The Regional Monitoring Program for Water Quality in San Francisco Bay is currently in the early stages of developing a "nutrient strategy." Efforts to develop the NNE assessment framework should be coordinated with the SF Bay RMP.**

6.5.1 Data Gaps and Recommended Next Steps for Development of a San Francisco Bay NNE Assessment Framework

Development of an NNE assessment framework for SF Bay involves specifying how primary and supporting indicators would be used as multiple lines of evidence to diagnose adverse effects of eutrophication in SF Bay. While development of the assessment framework begins as a scientific or technical work element, ultimately the selection of thresholds that would be used to determine whether the Bay is meeting beneficial uses is a policy decision. That policy decision is made by the SWRCB and the SF Water Board, with advice from its advisory groups (the SF Bay SAG and the STRTAG).

Assessment frameworks would need to be created for the habitat types identified in this review: 1) subtidal habitat, 2) seagrass habitat, 3) intertidal flats, and 3) muted intertidal and subtidal habitat. The Table 6.4 summarizes data gaps and recommended next steps for development of a SF Bay NNE assessment framework by habitat type. **Note that no attempt is made to prioritize or reduce/eliminate "next steps" any habitat types, despite acknowledged limitation in available resources. The SF Bay TAT assumes this prioritization and focusing of resources would be done by the SWRCB, the SF Water Board, with advice from its advisory groups.**

Unvegetated Subtidal Habitats. For unvegetated subtidal habitats in SF Bay, a long term data set on primary indicators such as dissolved oxygen, phytoplankton biomass, productivity, and taxonomic composition exist, albeit with some data gaps and poor information on the prevalence of HAB toxins in SF Bay. Adequate understanding exists of factors controlling long-term temporal and spatial trends in these indicators. The SF Bay TAT recommends that development of an NNE assessment framework for this habitat type proceed by:

- Sponsoring a series of expert workshops to develop a draft assessment framework based on phytoplankton biomass, productivity, cyanobacteria cell counts and toxin concentrations, and dissolved oxygen as the primary indicators

- Augmenting baseline monitoring data on HAB toxin concentrations

Use of supporting indicators such as ammonium, urea, and phytoplankton taxonomic composition or assemblages, and HAB species cell abundance and toxins in the NNE assessment framework would be greatly benefited by additional work and, in some cases, baseline data collection. This work includes:

- Formation of a workgroup of SF Bay scientists and outside expertise to develop indices of Bay health based on measures of phytoplankton taxonomic composition or assemblages.
- Formulating of a working group of SF Bay scientists to synthesize available data on factors known to control primary productivity in different regions in the Bay, developing consensus on relative importance of ammonium inhibition of phytoplankton blooms to Baywide primary productivity, and determining next steps with respect to incorporating ammonium into the NNE assessment framework for SF Bay.
- Augment USGS water quality data collection with sampling urea, HAB cell counts and toxin concentrations (water and faunal tissues, for a minimum of two years.

Table 6.4. Indicator status and recommended next steps for development of an NNE Assessment framework for San Francisco Bay.

Habitat	Indicator	Indicator Designation	Data Gaps	Recommended Next Steps
Subtidal Habitat	Dissolved oxygen	Primary	Wealth of data exist. TAT does not have expertise to review adequacy of DO objectives. Review did not address dissolved oxygen data in the tidally muted habitats of SF Bay. Additional analysis of existing data and consideration of scientific basis for DO objectives in these habitats is warranted.	Consider update of science supporting Basin Plan dissolved oxygen objectives, if warranted by additional review by fisheries experts. Review could be for entire Bay or limited to the tidally muted areas of the Bay.
	Phytoplankton biomass, productivity, and taxonomy	Primary.	Wealth of data exist. Need a review of science supporting selection of endpoints. Additional work required to improve ability to predict phytoplankton assemblage.	Recommend development of a white paper and a series of expert workshops to develop NNE assessment framework for phytoplankton biomass, productivity, taxonomic composition/assemblages, abundance and/or harmful algal bloom toxin concentrations. Recommend augmentation of current monitoring to include measurement of HAB toxin concentrations in water and faunal tissues.
	Harmful algal bloom abundance and toxin conc.	Cyanobacteria cell counts and toxin = primary; HAB cell counts and toxin = supporting	Little data on HAB toxin concentrations in surface waters and faunal tissues.	
	Ammonium and urea	Supporting	Lack of understanding of relative importance of ammonia limitation of nitrate uptake in diatoms on Bay productivity vis-à-vis other factors Lack of data on urea concentrations in SF Bay	Recommend formulation of a working group of SF Bay scientists to synthesize available data on factors known to control primary productivity in different regions in the Bay and evaluate potential ammonium endpoints. Recommend collecting data on urea concentrations in SF Bay over a two year period.
	Macrobenthos taxonomy, abundance and biomass	Co-factor	IEP-EMP has data on macrobenthos, but data lack in regions south of San Pablo Bay; lack of information on how to use combination of taxonomy, abundance, and biomass to assess eutrophication.	Recommend utilization of IEP-EMP data to explore use of macrobenthos to assess eutrophication in oligohaline habitats. Consider including biomass in the protocol to improve diagnosis of eutrophication. Determine whether combination of indicators can be used reliably to diagnose eutrophication distinctly from other stressors.

Seagrass Habitat	Phytoplankton biomass, epiphyte load and light attenuation	Phytoplankton biomass = primary, epiphyte load and light attenuation = secondary	Poor data availability of data on stressors to SF Bay seagrass beds. Studies needed to establish light requirements for species of seagrass found in SF Bay and studies to assess duration of reduced light/photosynthesis that results in adverse effects to the seagrass bed.	Recommend 1) Continued monitoring of aerial extent every 3-5 years (currently no further system scale monitoring is planned beyond 2010), 2) studies to establish light requirements for SF Bay seagrass species, 3) development of a statewide workgroup to develop an assessment framework for seagrass based on phytoplankton biomass, macroalgae, and epiphyte load and 4) collection of baseline data to characterize prevalence of macroalgal blooms on seagrass beds. Studies characterizing thresholds of adverse effects of macroalgae on seagrass currently underway in other California estuaries should be evaluated for their applicability to SF Bay.
	Macroalgae biomass and cover	Primary	Data gaps include studies to establish thresholds of macroalgal biomass, cover and duration that adversely affect seagrass habitat	
Intertidal Flat Habitat	Macroalgal biomass and cover	Primary	Lack of baseline data on frequency, magnitude (biomass and cover) and duration of macroalgal blooms in these intertidal flats	Recommend collection of baseline data on macroalgae , microphytobenthos and sediment bulk characteristics. Recommend inclusion of SF Bay scientists and stakeholders on statewide workgroup to develop an assessment framework for macroalgae on intertidal flats
	Sediment % OC, N, P and particle size	Supporting		
	MPB taxonomic composition and benthic chl a biomass	Supporting		

Habitat	Indicator	Indicator Designation	Data Gaps	Recommended Next Steps
Muted Habitat	Macroalgae	Primary indicator	Lack of baseline data on frequency, magnitude (biomass and cover) and duration of macroalgal blooms in muted habitat types	Recommend collection of baseline data on macroalgae, dissolved oxygen, phytoplankton biomass, taxonomic composition and HAB species/toxin concentration in these habitat types Recommendation to develop an assessment framework based on macroalgae, phytoplankton and dissolved oxygen in these habitat types. One component of this discussion should be a decision on beneficial uses that would be targeted for protection and to what extent the level of protection or expectation for this habitat type differ from adjacent subtidal habitat.
	Phytoplankton biomass and community composition	Primary indicator	Lack of baseline data on phytoplankton biomass and community composition in these habitat types	
	Dissolved oxygen	Primary indicators	Some data on dissolved oxygen exist. Unclear about what levels of DO are required to protect beneficial uses of muted habitats.	
	Phytoplankton taxonomy, abundance, and/or harmful algal bloom toxin conc.	Cyanobacteria cell counts and toxin = primary; 'taxonomic composition/assemblage and HAB cell counts and toxin = supporting	Little data on taxonomic composition, HAB toxin concentrations in surface waters and faunal tissues.	

Over time, work on macrobenthos may show that this component of the Bay ecosystem could provide useful information on eutrophication. However, science in this area is evolving and sufficient evidence does not exist to use it as a primary or supporting indicator. The SF Bay recommends including it as a co-factor.

Seagrass Habitat. For seagrass habitat, specific recommendations to develop a NNE assessment for seagrass include: 1) studies to establish light requirements for SF Bay seagrass species; 2) collection of baseline data to characterize prevalence of macroalgal blooms and other stressors on seagrass beds; and 3) inclusion of SF Bay scientists and stakeholders in statewide group to develop an assessment framework for eutrophication in seagrass, based on phytoplankton biomass, macroalgae, and epiphyte load.

It should be noted that studies characterizing thresholds of adverse effects of macroalgae on seagrass currently underway in other California estuaries. The findings of these studies should be evaluated for their applicability to SF Bay.

Intertidal Flat Habitat*.* For intertidal flat habitat, specific recommendations to develop a NNE assessment include: 1) development of a NNE assessment framework based on macroalgae and 2) collection of baseline data to characterize frequency, magnitude and duration of macroalgal blooms on SF Bay intertidal flats.

It should be noted that studies characterizing thresholds of adverse effects of macroalgae on intertidal flats currently underway in other California estuaries. The findings of these studies should be evaluated for their applicability to SF Bay. San Francisco Bay scientists and stakeholders should be included in statewide group to develop an assessment framework for macroalgae on intertidal flats.

Muted Intertidal and Subtidal Habitat*.* While some data on dissolved oxygen and HAB species cell counts exist for muted intertidal and subtidal habitat, there is a general lack of extended baseline data on these and other indicators (macroalgal biomass and cover and phytoplankton biomass, taxonomic composition, and HAB toxin concentrations) in these habitats needed to make a full assessment of eutrophication. Finally, it is recommended that SF Bay scientists and stakeholders be included in the statewide effort to develop an assessment framework for muted habitats, based on macroalgae, dissolved oxygen, phytoplankton measures, and HAB toxin concentrations.

6.5.2 Data Gaps and Recommended Next Steps to Quantify External Nutrient Loads

Table 6.5 provides a summary of data gaps and recommended next steps. Recommendations generally fall into two categories: 1) Revising and updating estimates of nutrients from the different sources, based on existing data; and 2) Identification of data needed to develop a dynamic loading model.

The exercise of revising and updating estimates of nutrients from the various sources, based on existing data would help to better inform our understanding of the dominant nutrient sources for each distinct region of the Bay. This would, in turn, assist in decision-making to prioritize new data collection to develop the loading model(s) (discussed further below).

Table 6.5. Summary of data gaps and recommended next steps for quantification of external nutrient loads to San Francisco Bay.

Source	Data Gaps Identified	Recommended Next Steps
Atmospheric Deposition	No recently published data on wet & dry atmospheric deposition	Loads likely relatively small. Literature review to determine range of N and P deposition rates for West Coast coastal urban areas. Recommend baseline atmospheric deposition monitoring of wet and dry N and P deposition over 1-2 yr period to better constrain estimates.
Terrestrial Loads from Delta	Data available through RMP on dry season concentrations. No data available on wet weather concentrations during storm flow	Loads likely large. Recommend analysis of existing RMP data to estimate dry season nutrient loads. It is recommended that wet weather data collection of nutrients be initiated at the DWR sampling location at Mallard Island at the head of Suisun Bay to support improved daily loads estimates for 1995-present.
Municipal Effluent	Data available through 15 of approx. 40 POTWs	Loads likely large. Synthesize nutrient discharge and concentration data to estimate loads over period of last 10-20 years Encourage all treatment plants that discharge to the Bay to begin analyzing effluent for total and dissolved inorganic nutrients and to submit these data to the SFRWQCB on a regular basis. Recommend that the POTWs conduct a laboratory inter-comparison on nutrient methods to assure comparability of estimates.
Industrial Effluent	Some data available from the 1990s. Recent data availability unknown	Loads likely small relative to municipal wastewater. Synthesize available data to provide information for prioritization of any future steps.
Stormwater	Some data available but general lack of wet weather data sufficient to calibrate and verify a dynamic loading model	Loads likely large. Synthesize data to provide an updated estimate of stormwater contributions to assist prioritization of next steps. Scope the data needs associated with the development of a dynamic loading model.
Groundwater	Some data available from 79 USGS monitoring stations surrounding the Bay. Flow data currently less well understood.	Loads likely small. Refine current loads estimates after review of local USGS groundwater experts in order to support prioritization of next steps if any.
Exchange with Coastal Ocean	Some data available for fluxes of water and sediments during selected tides and seasons in the past decade collected by USGS and UC Berkeley using comparable methods.	Initiate a workgroup of local experts to design a sampling program for nutrient flux at the Golden Gate boundary, with the intent of developing a hydrodynamic and material flux dynamic model to describe exchange with coastal ocean

6.5.3 Data Gaps and Recommended Next Steps for Development of Load-Response Models

An important component of implementing the NNE framework in SF Bay is the development of load-response models that can simulate the ecological response of the Estuary to nutrients and other important co-factors.

Models developed to manage nutrient loads and eutrophication in the Chesapeake Bay Estuary (http://archive.chesapeakebay.net/pubs/backgrounder_CBP_Models.pdf) are one example of linked watershed loading and receiving water models. These models were developed and refined through a 30 years of collaboration by federal, state, academic and private partners. Several types of models would need to be developed that fit into two general categories:

- Air, Oceanic and Watershed Loading Model, which could incorporate information about land use, fertilizer applications, wastewater plant discharges, septic systems, air deposition, farm animal populations, weather and other variables to estimate the amount of nutrients and sediment reaching the SF Bay estuary and where these pollutants originate. The loading model would include three components: 1) a hydrologic sub-model, 2) a non-point source sub-model, 3) a river sub-model which routes flow and associated nutrient loads to the Estuary, and 4) an ocean exchange model.
- Estuary water quality model, which simulates the ecosystem response to pollutant loads, and consists of two sub-models: 1) a hydrodynamic sub-model that will simulate the mixing of waters in the Estuary and its tidal tributaries and 2) a water quality sub-model that simulates the Estuary's biological, chemical and physical dynamics in response to nutrient loads from the watershed, air and ocean and other factors (light, temperature, grazing, etc.).

The models would be used to establish load allocations of nutrients that the SF Bay estuary can sustainably assimilate. It would also be used to generate simulations of the past, present or future state of the Estuary, watershed, airshed, and ocean (e.g., population growth, climate change, etc.) to explore potential effects of management actions and evaluate alternatives. Thus these models would be a key component of a strategy to adaptively manage SF Bay.

Ideally sufficient data and knowledge of SF Bay should exist to support the development of system wide dynamic simulation models to predict phytoplankton biomass/community response and relationships to models of secondary productivity. This goal is not likely in the short term, so it is important to consider that the development of a more complex model should follow the testing out of key concepts and assumptions in smaller, simpler models.

Scoping the development of these NNE load response models should begin through the iterative development of a modeling strategy and workplan. During this strategy workshop, participants would describe the types of models that would be needed. Sufficient detail would need to be given to accomplish three elements:

- *Nutrient Budget for San Francisco Bay*. This step would utilize existing data to synthesize a nitrogen and phosphorus nutrient budget for SF Bay. Existing data that describe the timing and magnitude of external sources, internal sources, sinks, and pathways of transformation such as benthic nutrient flux, nitrification, denitrification, etc. would be compiled in order to synthesize current understanding of sources and fate of nutrients as well as identify critical data gaps in advance of the modeling strategy development.
- *Conceptual Model Development*. There is a need to develop conceptual models that explicitly show linkage between watershed, airshed, ocean and estuarine hydrology, nutrient loads, ecological response indicators, and “co-factors” that control ecological response to eutrophication or oligotrophication. The conceptual model would identify key sources, sinks and processes of transformation that would need to be incorporated into the numeric simulation models. Areas of disagreement on causal mechanisms should be synthesized as alternative hypotheses than can be tested through experiments, field studies and model sensitivity analyses.
- *Review of Existing Models and Model Selection*. The next step in the scoping of model development is to select the appropriate models. This should be done by reviewing available loading and receiving waterbody models and present an analysis of the advantages and disadvantages of their use for modeling eutrophication and other adverse responses to nutrients, based on the explicit conceptual models. A review of existing hydrodynamic, sediment transport, and water quality models developed for SF Bay and their applications should be undertaken, with the intent of understanding what existing tools may be used to leverage development of load-response models.
- *Data Needs Assessment*. Based on explicit conceptual models and the modeling platform selected, the next step would be to identify data required to support model development, calibration and validation.

The product of this effort would be the a coarse nutrient budget for SF Bay, identification of the appropriate models, data gaps and recommended studies, a phased workplan, timeline and budget to develop these models, and identification of and coordination among key institutions, programs and respectively roles. This information could be synthesized into a workplan to develop the loading and estuary water quality models and a preliminary timeline and budget for Phase I of the effort.

6.5.4 Coordination of Development of the SF Bay NNE Framework with Nutrient Management in the San Joaquin and Sacramento River Delta

Development and implementation of a NNE framework for SF Bay will require improve coordination with nutrient management activities in the San Joaquin and Sacramento River Delta. At the time of writing this report, preliminary discussions on this topic are beginning with the Central Valley Water Board staff. Coordination should be improved, at minimum, with respect to any future monitoring and/or modeling of nutrient loading, transport and source identification, as SF Bay and the Delta exchange nutrients across their aquatic and terrestrial boundaries. Coordination would be further enhanced by a similar review of NNE candidate indicators, summary of existing science, and identification of data gaps and recommended next steps specifically for the Delta.

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