The Utility of Water Quality Modeling for San Francisco Bay

Nutrient Strategy Development Project

September 16, 2011

1. INTRODUCTION

This paper was prepared for the Bay Area Clean Water Agencies (BACWA), a joint powers agency whose members collectively provide municipal sanitary services to more than seven million people in the San Francisco Bay Area. BACWA's mission is to provide an effective voice for its members' role as stewards of the San Francisco Bay environment through leadership, science, and advocacy. One of BACWA's goals is to ensure that environmental regulations and policies reflect the best available scientific, technical and economic information and that these regulations and policies balance environmental, social, and economic sustainability.

Long-term water quality monitoring indicates that many portions of San Francisco Bay have experienced marked increases in chlorophyll-a, a pigment found in all green plants including phytoplankton. The exact causes of this change are unknown, but may include changes in light regimes, changes in ecosystem function resulting from invasive species, and coastal influences. Often the availability of nitrogen is the factor that limits phytoplankton growth in estuaries. However, San Francisco Bay ambient nutrient concentrations are relatively high, and have not changed significantly in recent years. While Bay nutrient concentrations are on par with those of other nutrient-impaired estuaries, such as the Chesapeake Bay, the San Francisco Bay has typically been considered resilient to the effects of nutrient loads and has not experienced similar water quality impairment. Concern exists, however, that changes in factors other than nutrient concentrations may lead to nutrient-related impairment.

As sources of nitrogen and phosphorous – the natural end products of wastewater treatment – BACWA member agencies recognize that, should current trends continue, they will play a key role in efforts to reduce nutrient loading. Before undertaking potentially substantial investments to reduce nutrient loading, BACWA agencies have an obligation to their ratepayers to ensure that these investments are necessary to improve water quality, and will not have other unintended environmental impacts. BACWA has engaged consultant assistance to support the development of technical information to determine whether nutrients are impairing the estuary. While this question has not yet been answered, BACWA is also developing a framework for protecting the Bay in an informed way.

BACWA's objective with respect to nutrients is to support the development of scientifically based regulations that will result in water quality improvements while balancing environmental and economic objectives. To this end, BACWA is developing information on the technical aspects of nutrient management as well as potential future regulatory framework. While BACWA's priority is to understand the estuary sufficiently to know whether nutrients are causing impairment, information is also being developed to support regulatory strategy development, should it be needed.

This modeling topic paper is the first in a series of three topic papers. All three topic papers are scheduled to be reviewed and completed in October 2011. Descriptions of the other two remaining topic papers are provided below:

- Impacts of Potential Wastewater Treatment Plant Upgrades: Describe to BACWA member agencies the implications of transitioning from secondary to nutrient removal treatment. In particular, this topic paper (1) describes a range of facility requirements and the potential impact on POTW operations, (2) presents a basis for a range of unit costs, and (3) addresses the unintended consequences related to the conversion from secondary to nutrient removal treatment.

- Regulatory Framework: This topic paper will characterize the unique challenges posed by nutrient management regulatory requirements for municipal dischargers and to outline appropriate discharge permitting structures for practical, technically achievable, and affordable compliance.

1.1 Purpose

The purpose of this topic paper is to initiate the development of a BACWA model development framework, which may be presented to the State Water Resources Control Board (SWRCB) and the San Francisco Regional Water Quality Control Board (RWQCB) as a potential methodology for developing the "cause and effect" model required by the Numerical Nutrient Endpoint Project.

1.2 Background

The United States Environmental Protection Agency (USEPA) has delegated authority of implementing the Clean Water Act in California to the California State Water Resources Control Board (SWRCB). In turn the SWRCB has worked with Regional Water Quality Control Boards (RWQCB) to develop water quality objectives, which are used to assess the condition of the State's water bodies, for regional basin plans as well as SWRCB statewide plans. Towards that end, the SWRCB and RWQCBs are developing a conceptual approach to nutrient water quality objectives, using a nutrient numeric endpoint (NNE) framework. The NNE framework is based on the concept that biological response variables or indicators are better suited to evaluating beneficial use impairment rather than using pre-defined nutrient concentrations. The NNE approach permits a weight of evidence approach with multiple indicators rather than just one or two nutrient concentration values alone and thus provides greater scientific validity and defensibility.

The San Francisco Bay RWQCB is working with the SWRCB, the Southern California Coastal Water Research Program (SCCWRP) and the San Francisco Estuary Institute (SFEI) to develop NNEs for the San Francisco Bay (SFB) Estuary. One of the first steps in that process was to provide funding to SFEI to perform a literature review and data gaps analysis. This work has been completed with the issuance of a technical report by SFEI (McKee et al., 2011) detailing their findings. A key recommendation of the report was the need to develop load-response models that can simulate the ecological response of the Bay to nutrients and other important co-factors.

2. UTILITY OF MODELS

Besides the McKee et al. (2011) recommendation for the development of a load-response model (and the necessary research and data collection necessary to support its development) for use in the NNE process, a nutrient based eutrophication model of SFB could help water quality and natural resource managers better understand and respond to nutrient-related hypotheses and questions being posed by bay scientists and academics and/or environmental groups. For example, at the recent Regional Monitoring Program (RMP) workshop on nutrient science and management in San Francisco Bay held on June 29, Jim Cloern presented an analysis of long-term trends in SFB water quality, which indicated that over the past 25-30 years, there has been a 32% increase in chlorophyll-a in the Suisun Bay portion of north SFB, a 72% increase in San Pablo Bay and a 105% increase in annual chlorophyll-a in south SFB. South SFB has also seen a 213% increase in June-October chlorophyll-a levels and a 4% decline in bottom DO. Jim Cloern closed his presentation with a slide showing an analysis of the long-term trend in chlorophyll-a in South San Francisco Bay and projecting future conditions (Figure 1) and asked the question, "Will San Francisco Bay's water quality become impaired by nutrients?" A nutrient-based water quality model of SFB may help to answer this question.

It is important to remember, however, that the SFB Estuary has been and continues to be an estuary in a state of flux. This is particularly true when it comes to salinity-induced changes in benthic filter feeders in Suisun Bay, which may have contributed to the low levels of phytoplankton during the 1976-1977 drought (Nichols, 1985); the invasive clam *Potamocorbula amurensis* in San Pablo and Suisun Bay (Alpine and Cloern, 1992), which appear to contribute to reduced phytoplankton biomass in north SFB, and more recently invasive predators that have reduced suspension feeding bivalves in south SFB (Cloern et al., 2006). Therefore, it may be unreasonable to assume that a single water quality model framework would be able to directly address water quality issues in SFB that are perturbed by random changes in ecosystem function and trophic structure (i.e., invasive species). A model could be constructed, however, that would be able to answer, at least in a relative sense, "What if?" questions. For example:

- Can the water quality model, using the 35% reduction in light attenuation that has resulted from the 22% reduction in suspended sediments explain the 32% increase in chlorophyll-a reported by Cloern (2011)?
- What will future phytoplankton levels in south SFB be if turbidity decreases another ten percent over the next five years and all else remains the same?
- What will future phytoplankton levels in south SFB be if benthic filter feeders decline another ten percent over the next five years and all else remains the same?
- If turbidity decreases ten percent and benthic filter feeders decline ten percent over the next five years, how much would point source loads of nitrogen need to be reduced to maintain current levels of phytoplankton biomass?
- How might future reductions in algal growth due to the implementation of nutrient reduction management efforts in south SFB affect light availability for submerged aquatic vegetation (SAV)?

Once such a model is constructed it can also be used as the cause-effect ecosystem model recommended by the authors of the NNE literature review and data gaps analysis, as well as a tool that would permit local water quality and natural resource managers to judge the efficacy of various management scenarios.

The use of mathematical models and their utility to addressing environmental problems is not new. Beginning with the seminal work of Streeter and Phelps (1925) and Velz (1939), mathematical modeling has long been utilized to analyze potential environmental outcomes. These early models were developed for streams and provided a framework for evaluating the oxidation of organic matter, mainly organic carbon, and its impact on dissolved oxygen. The models included inputs of point sources and recognized the importance of benthic oxygen demand and photosynthetic oxygen production. These models were expanded in the late 1950's and 1960's to include a theoretical basis for reaeration (O'Connor and Dobbins, 1958) and extension to estuaries (O'Connor, 1962) and time-variable applications (O'Connor, 1967). Subsequent model developments expanded from biological oxygen demand-dissolved oxygen (BOD-DO) to include nutrients, phytoplankton and dissolved oxygen (DiToro et al. 1971, 1973, Thomann et al., 1974, 1975, Thomann and Fitzpatrick, 1982).

Water quality models are generally based on and have been used to understand cause and effect relationships, i.e., oxidation of organic matter and resulting drawdown of dissolved oxygen concentrations; inter-relationships between nutrients, primary production, phytoplankton biomass, and dissolved oxygen; and the deposition of organic matter to the sediment bed, its subsequent decomposition or diagenesis and the flux of resulting end-products back to the overlying water column.

Water quality models have continued to evolve, adding new state-variables and processes (refined algal growth models (Laws and Chalup, 1990, Los et al., 2008), sediment nutrient diagenesis and flux (DiToro and Fitzpatrick, 1993) and suspension feeder bivalves (Meyers et al., 2000)) and increasing spatial resolution (Cerco, 1995, Cerco, 2000). This evolutionary process has been driven by both advances in research science and recognition that additional processes need to be incorporated in models in order to provide a better calibration/validation of the model outputs to observed data. The needs of the latter have also been used to focus additional data collection and monitoring (e.g., forms of organic matter, attention to nutrient concentrations at the coastal or oceanic boundaries of estuaries, sediment oxygen demand, nutrient flux and denitrification rates, etc.) and research efforts (filtration and assimilation rates of benthic filter feeders (ex., Werner and Hollibaugh, 1993), pH-enhanced release of phosphorus from the sediment bed (Seitzinger, 1991)).

Water quality models have also been used by water quality and natural resource managers as a tool to define the impacts of various pollutant sources (point vs. nonpoint), to assess required levels of pollutant control (i.e., total maximum daily loads or TMDLs), to evaluate planning alternatives for water quality management, including nutrient trading programs, and to assess future water quality conditions and how long it may take for a waterbody to reach a new equilibrium after a management action is implemented.

McKee et al. (2011) identified the need for development of models that link response indicators to nutrient loads or "load-response" models for San Francisco Bay. McKee et al. stated that two general categories of models be developed: (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 would simulate the ecosystem response to nutrient loads and other management controls. The latter model would be a system-wide dynamic simulation that would predict phytoplankton biomass/community response to nutrients and management controls. Further McKee et al. recommend that the development of a complex system-wide ecosystem model should follow the testing out of key concepts, processes and assumptions in smaller, simpler models.

3. MODELING FRAMEWORK FOR SAN FRANCISCO BAY

3.1 Conceptual model

The first step in the development of a "cause and effect" water quality model for San Francisco Bay should be the development of a conceptual model (CM) of the Bay. Generally CMs have been developed for contaminated waste sites and via a series of diagrams, figures and narrative text express the inter-relationships between contaminants and the physical, chemical, and biological processes that transport contaminants from sources to receptors. However, a nutrient-based CM should also be considered for San Francisco Bay (SFB). It is recommended that a regional workshop be conducted that invites Bay-area scientists, regulatory agencies and stakeholders that would look to develop a consensus CM for the SFB system. With that in mind, this paper presents a nutrient-eutrophication "strawman" CM, as well as a modeling "strawman", for the Bay, which is based on historical and current information available in the scientific literature, that might help facilitate discussions at the regional workshop.

Although under optimum conditions (temperature, light, residence time, and zero predation) algal growth rates can be as high as 1-2 doublings per day, conditions for algal growth in the natural environment are seldom at optimal conditions. Reasons for the non-optimal conditions have to do with seasonal variations in:

- temperature, which affects algal growth and respiration rates,
- background suspended solids and dissolved organic matter, which contribute to turbidity,
- high rates of freshwater inflow and weak or non-stratified conditions, which affect either residence time in the estuary itself or residence time in the photic zone (i.e., surface waters) of the water column, and
- both pelagic (zooplankton) and benthic (bivalve suspension feeders) grazers, which consume phytoplankton biomass.

In the past, the SFB system has not appeared to support a level of algal biomass commensurate with observed levels of available nutrients (Cloern, 1982, Alpine and Cloern, 1988). The reasons for the low levels of algal biomass in various regions of the SFB estuary were hypothesized to be related to light-limitation or turbidity in north SFB (Cole and Cloern, 1984, Cloern, 1987), grazing by benthic filter feeders in south SFB (Cloern, 1982) and low residence time due to either freshwater inflows in north SFB (Cloern et al., 1985) and strong vertical mixing by tides in south SFB (Cloern, 1991). Recent data analysis, however, suggest that the SFB system may be changing. Cloern et al. (2006) documented a regime shift from 1999-2005 wherein the following were observed: larger spring algal blooms, unusual blooms during other seasons, and an increase in the annual baseline minimum chlorophyll-a. These changes have been hypothesized (Cloern et al., 2006) to be related to reductions in sediment inputs, which have resulted increases in water column transparency and light availability, reductions in benthic filter feeder biomass due to predation by bottom feeding flatfishes, and potentially due to import from the Pacific Ocean during coastal upwelling events.

In the SFB system, besides the transport of nutrients and phytoplankton biomass within the system, freshwater inputs and estuarine circulation also have a direct effect on the growth of phytoplankton. This is due to their combined effect on residence time. Jassby (2005) showed the importance of the San Joaquin River discharge on chlorophyll-a concentrations in the San Joaquin River at Vernalis and Mossdale. He noted that chlorophyll-a levels are lower at higher discharges or flows because phytopankton are flushed out of the system faster than they can grow. At low flows with longer residence times phytoplankton can accumulate within the river because their growth rates are now greater than the flushing rates and, therefore, chlorophyll-a levels are able to increase.

Freshwater inflows and estuarine circulation also affect vertical mixing, which in turn can influence phytoplankton growth. Cloern (1991) demonstrated, from both data analysis and a simple one-dimensional (1-D) vertical model of phytoplankton growth versus vertical turbulence, the importance of vertical stratification and, therefore, residence time on phytoplankton biomass in south SFB. Huzzey et al. (1990) also noted the importance of residence time to bloom development in south SFB, wherein they measured three periods of significant biomass accumulation over the northeast shoal in south SFB. They noted that the end of each bloom episode coincided with the occurrence of a large lateral (shoal-to-channel) flow event. When the residence time was long in the northeast shoal a large bloom was able to develop, until the lateral flow event flushed the region.

Besides freshwater and estuarine circulation that transport nutrients and phytoplankton biomass and that influence phytoplankton growth by affecting residence time, there are other physical, chemical, and biological processes that affect nutrients, phytoplankton and dissolved oxygen within SFB. Numerous researchers have investigated and reported on these processes over the past several decades in an effort to explain the relatively low levels of phytoplankton biomass and primary productivity within the SFB system (Cloern, 1982, Cole and Cloern, 1984, Cloern et al., 1985, Cloern, 1987, Alpine and Cloern, 1988, Cloern, 1991), as well as to explain historical and recent changes in phytoplankton biomass and primary productivity in the SFB system (Cloern et al, 2006, Cloern et al, 2007). Based on a review of this and other literature, it appears that the following processes are influencing phytoplankton biomass within the SFB system:

- Turbidity or light-availibility – related to suspended solids transported in the estuary from the Sacramento and San Joaquin Rivers as well as due to tidal and wind-induced resuspension of bottom sediments,
- Losses due to filtration – related to benthic filter feeders, including native and invasive species, such as *Potamorcorbula amurensis*, and
- Recent reductions in benthic filter feeders and zooplankton – related to high fish predation and new invasive species.

Therefore, the proposed CM model for SFB should include, at a minimum, the following environmental forcings and physical, chemical, and biological processes: freshwater inflows and associated nutrient loadings; estuarine circulation; phytoplankton growth and respiration as affected by temperature, light availability and nutrients; suspended solids, since they strongly influence light availability for phytoplankton growth; and phytoplankton grazing losses due to zooplantkon and benthic filter feeders. While in the past nutrient availability and nutrient cycling may not have been important to understanding phytoplankton dynamics within the bay, since light limitation had a greater impact on phytoplankton growth, both should be part of the CM. The reason for including nutrient cycling is that it can be used to help develop a total nitrogen balance for the bay, which may be important in future efforts to develop waste load allocations, should such efforts become necessary with the development of the NNEs.

3.2 Modeling Components

Based on the above CM, the first component of the modeling strawman or modeling framework (Figure 2) is a hydrodynamic model. It is important to have a hydrodynamic model not only to compute the movement and transport of nutrients and phytoplankton biomass, but also to reliably predict estuarine circulation, including the physical effects of winds and tidal mixing on vertical stratification and lateral transport between the intra-tidal flats and the deep channels of the SFB system. Computation of estuarine circulation, including vertical stratification and lateral transport, is important to the computation of residence time and, therefore, to the computation of phytoplankton productivity and dissolved oxygen. A number of hydrodynamic models have been constructed for the Bay and will be discussed later in this paper.

The second component of the modeling strawman is the water quality model. The water quality model is potentially comprised of three submodels (Figure 2): a nutrient-based water column eutrophication submodel; a sediment diagenesis and nutrient flux submodel; and a suspension feeder submodel. The eutrophication submodel should include state-variables for nutrients, phytoplankton, and dissolved oxygen and should include processes that affect phytoplankton growth, such as temperature, light, and available nutrients, as well as the effects of suspended solids on light, and the effects of zooplankton and benthic filter feeders on net phytoplankton growth. The latter (zooplankton/benthic filter feeders) could be accomplished either by directly including these processes as state-variables in the model or via spatially - and time-varying loss rates based on site-specific data analysis and the literature.

Modern eutrophication models (e.g., Chesapeake Bay, Long Island Sound, New York Bight, Massachusetts and Cape Cod Bays, etc.) include various nutrient state-variables (organic and inorganic and dissolved and particulate forms of nitrogen, phosphorus and silica) and processes or kinetics that describe the relationships between phytoplankton and nutrients and nutrient recycling (i.e., nutrient uptake by phytoplankton, nutrient regeneration resulting from algal death and grazing by zooplanton and benthic filter feeders, and nutrient cycling resulting from hydrolysis and mineralization). These models also include a sediment nutrient diagenesis/flux sub-model (or sediment flux model – SFM) to account for the deposition of phytoplankton biomass and detrital organic matter, its diagenesis or decomposition in the sediment bed, and the flux of resulting end-products back to the overlying water column. The SFM framework also accounts for the loss of nitrogen from a system via sediment nitrification/denitrification ($NH_4 \rightarrow NO_3 \rightarrow N_2$ (gas)). Seitzinger (1988) has estimated that sediment denitification can account for the loss of 15-70% of the nitrogen input to an estuary. HydroQual (2000), in a system-wide mass balance for the Massachusetts Bays system found that denitrification accounted for about a 50% loss of nitrogen delivered to the sediment bed and about 13% of the annual inorganic nitrogen delivered to the Bays from coastal input from the Gulf of Maine, POTWs and atmospheric deposition.

Only a few of today's eutrophication models (Chesapeake Bay and Jamaica Bay) actually include state-variables for zooplankton and suspension feeders. Most other models parameterize the effects of zooplankton or benthic grazing by specifying spatially- and time-varying loss rates based on literature and data analysis. While utlimately, it may be desireable to include a state-variable based suspension feeder bivavle model in the water quality model, as a first step in the model development process, it is recommended that only zooplankton state-variable be included and that the suspension feeders be represented by spatially - and time-varying loss rates based upon the available data and literature information. This in part due to the fact that the SFB benthic community has been strongly influenced by invasive species (both the native suspension feeder population that has been replaced by invasive suspension feeder communities, as well as predatory invasive species that have fed on native and invasive suspension feeders and reduced suspension feeder biomass). Building this somewhat stochastic process into a deterministic modeling framework would be very challenging at best and pehaps beyond the state-of-the-science.

The proposed nutrient-based eutrophication model for SFB is shown in Figure 3 and contains state-variables and kinetic processes that include phytoplankton biomass as carbon (C) and chlorophyll-a, zooplankton biomass, various forms of organic and inorganic nutrients (nitrogen (N), phosphorus (P) and silica (Si)), detrital organic carbon, dissolved oxygen (DO), and suspended sediments. Processes include nutrient uptake by phytoplankton and nutrient losses via respiration, death and grazing by zooplankton and benthic filter feeders. Other processes include nutrient recycling (hydrolysis and mineralization), oxidation of organic carbon, atmospheric reaeration, effects of suspended solids and dissolved organic matter on light-attenuation. The SFM is also included in the eutrophication model framework.

It is important to note, that the proposed eutrophication model is not meant to capture HAB (Harmful Algal Blooms) phytoplankton blooms. With the exception of a few well understood and regional HABs (ex., *Alexandrium fundyense* in the Gulf of Maine (McGillicuddy et al., 2005) and cyanobacterial blooms in the Black Sea (Roiha et al. (2010)) modeling of the development and spread of HAB blooms is generally beyond the state-of-the-science. In addition, as suggested by Cloern et al. (2005), the dinoflagellate bloom that occurred in September, 2004, while perhaps intensified in SFB by local climatic and physical conditions, may have been seeded by high dinoflagellate biomass in the coastal waters off SFB. Therefore, it would be necessary to develop large-scale coastal hydrodynamic and eutrophication/HAB models in order to set boundary conditions (or seed concentrations) for SFB. Instead, it may be more feasible to investigate the use of linked deterministic-empirical models for SFB. For example, Lane et al. (2009) developed a logistic regression model for the prediction of toxigenic *Pseudo-nitzcshia* blooms in Monterey Bay. The logistic model considered chlorophyll-a, silicate, and season-specific temperature, upwelling index, river discharge, and/or nitrate. Predictive power for bloom cases was shown to be > 75%. Similarly, Anderson et al. (2010) used a logistic Generalized Linear Model for the prediction of toxigenic *Pseudo-nitzcshia* blooms in Chesapeake Bay. Small-threshold blooms ($\geq$10cells/mL) were explained by time of year, location, and variability in surface values of phosphate, temperature, nitrate plus nitrite, and freshwater discharge. Medium- (100 cells/mL) to large threshold (1000 cells/mL) blooms were further explained by salinity, silicic acid, dissolved organic carbon, and light attenuation (Secchi) depth. If similar logistic relationships could be developed for SFB, then the proposed water quality model could be used to compute the concentrations of the environmental variables that go into the logistic model, which in turn could estimate potential HAB biomass or risk of HAB blooms.

Besides developing logicistic models for HAB prediction, logistic models or other forms of regression model might also be developed for estimating SAV. For example, Kemp et al. (2000) used logistic and classification and regression tree (CART) models to explore relationships between SAV and water quality in Chesapeake Bay. They found that salinity zone-specific models were most successful in predicting the presence of SAV in shallow waters areas as related to water quality in adjacent main channel water quality stations. These models indentified total suspended solids (TSS) as the most important variable in low salinity areas, TSS and chlorophyll-a as most important in mesohaline areas and dissolved inorganic nitrogen as most important in higher salinity areas. Again if similar regression models can be developed for SFB, then the proposed water quality model could be used to compute the concentrations of the environmental variables that go into the regression models, which in turn could estimate potential increases in SAV biomass and/or areas of potential SAV recovery.

3.3 Data Requirements

The development of a scientifically defensible model, be it a hydrodynamic model, water quality model, or ecosystem model requires a substantial investment in the collection and analysis of field and laboratory data with which to calibrate and validate the model. Ideally, the data sets to be used for model calibration and validation should span a number of years that encompass varying environmental conditions (e.g., average, wet and dry freshwater inflows). Calibrating and validating a model against varying environmental conditions provides confidence in the robustness or skill of the model and provides greater confidence in model projections of future conditions in response to management actions.

The data requirements to parameterize and calibrate/validate the proposed modeling system for SFB are substantial. For the hydrodynamic model the data requirements include:

Model Forcings

- Estimates of freshwater inputs: riverine (Sacramento and San Joaquin), POTW and industrial discharge rates, stormwater runoff, precipitation
- Meteorological: solar radiation, air temperature, relative humidity, barometric pressure, winds, etc.
- Boundary tides or water elevations, salinity and temperature

Model Calibration/Validation

- Water levels – tide gauges
- Salinity and temperature – longtitudinal and vertical casts
- Current speed and direction – Acoustic Doppler Current Profilers (ADCPs)

Most of these data can be obtained from continuous recorders. Data records should be from several weeks to several months in duration and should capture high and low flow events.

The data requirements for the water quality model are more extensive in terms of the types of data needed for both model inputs, parameterization and calibration/validation and include:

Model Inputs:

- Transport fields (advection and dispersion) – provided by the hydrodyanmic model
- Nutrient loading estimates: riverine, oceanic exchange, POTW and industrial effluent, stormwater runoff, agricultural runoff, and atmospheric inputs for various nutrient forms
- Suspended solids loading estimates: riverine, oceanic exchange, POTW and industrial effluent, stormwater runoff, and agricultural runoff
- Solar radiation, fraction of daylight, winds
- Grazing rates: depending upon whether zooplankton and benthic filter feeders are state-variables or not – spatially and time-varying rates of grazing or filtration rates

Model Parameterization and Calibration/Validation

- Water column concentrations of phytoplankton biomass (chlorophyll-a) and species composition, nutrients (N, P, Si, C) and various nutrient forms, DO and suspended solids
- Measures of primary production and community respiration
- Light attenuation or extinction coefficients – turbidity or secchi depth
- Sediment oxygen demand and nutrient flux, including rates of denitrification
- Sediment composition – solid-phase and pore water nutrients
- Zooplankton and benthic filter feeder biomass and species composition.

Generally, these data are collected as discrete samples and for the water column data are monitored on a bi-weekly to monthly basis. Sediment nutrient flux and composition can be monitored on a quarterly to semi-annual basis. Zooplankton should be monitored on a monthly to seasonal basis and benthic biomass on a seasonal or semi-annual to annual basis.

4. REVIEW OF EXISTING MODELS AND DATA FROM SAN FRANCISCO BAY

4.1 Hydrodynamic Models

A number of hydrodynamic models have been constructed for SFB. The oldest of these models was a version of the RMA2, two-dimensional (2-D) time variable finite element model applied by the USACE (Pankow, 1988) to evaluate dredged material disposal. The model extended southward from Benicia and included San Pablo Bay and central and south SFB. The model was subsequently extended upstream to include Suisun Bay by the USACE (Hauck et al., 1990). The next oldest model is the Delta Simulation Model – DSM developed by the California Department of Water Resources (DWR) in the early 1990’s (replaced in 1997 by DSM2). It is a 1-D, unsteady, open-channel flow model. Its main focus is on the San Joaquin Delta, but it does include Suisun Bay. The next of the 2-D models of the SFB system was developed by Uncles and Peterson (1995, 1996). The Uncles-Peterson (U-P) model has vertical structure (surface and bottom layers) and the upper terminus of the model is at the confluence of the Sacramento and San Joaquin River and the model extends southward to include south SFB (Figure 4a). The model was calibrated, using surface salinity data from seven stations, for a 22-year period (1967-1988). More recently Lionberger and Schoellhamer (2009) validated the hydrodynamic model for water year 1999 (October 1998-September 1999) and extended the model to include a sediment transport submodel and performed a calibration using data from water year 1999. Other 2-D models include MIKE-21, developed by the Danish Hydraulics Institute and applied by URS (2007) for the Brake Pad Partnership to evaluate the fate and transport of copper from brake pad wear debris in SFB (Figure 4b). Recently the DWR, the University of California-Berkley and the Lawrence Berkeley National Lab has been working on a 2-D adaptive mesh model of SFB and the Western Delta. Unlike the U-P model, the MIKE-21 and REALM models do not have vertical structure.

A number of three-dimensional (3-D) models of the Bay have also been constructed. These 3-D models include:

- TRIM/TRIM3D (Figure 5a) – developed by the USGS. The original TRIM model is a structured, depth-averaged (2-D) tidal hydrodynamic model developed by Ralph Cheng (USGS) and Vincenzo Casulli (Trento University, Italy). The model includes south, central and north SFB. A 3-D version of the model was developed and applied to SFB by Gross et al. (2010). Both models have been well calibrated to observed water levels, flow and salinity and temperature data collected by NOAA, DWR and the USGS. The model is proprietary.
- UnTRIM (Figure 5b) – developed by Mac Williams et al. (2007). The UnTRIM Bay-Delta model is an unstructured 3-D hydrodynamic model that extents from the Pacific Ocean through the entire Sacramento-San Joaquin Delta. The model has been well calibrated to observed water levels, flow and salinity and temperature data collected by NOAA, DWR and the USGS. The model is proprietary.
- SUNTANS (Figure 5c) – developed by Stanford University. SUNTANS is an unstructured 3-D hydrodynamic model of the SFB system. The model extends from the Pacific Ocean up to the confluence of the Sacramento-San Joaquin Rivers. The model has been partially calibrated and is open-source (i.e., publically available).
- Delft3D-FLOW (Figure 5d) – developed by Deltares (formally WL | Delft Hydraulics). The USGS and Deltares have been working together to build a 3-D time-variable hydrodynamic/sediment transport/morphological model of the SFB system. The model extends from Point Reyes to Sacramento and Vernalis. The model can run in either 2-D or 3-D mode. Deltares has recently put the FLOW module of Delft3D into the public domain.

One potential concern of the 3-D hydrodynamic models is the lengthy run times associated with some of them. For example, the TRIM3D model runs in $1/10^{th}$ real time. So to simulate one-year of hydrodynamics would require about one month to simulate. Potentially this could present a problem for running water quality models on the same computational grid, since water quality models generally have 2-4 times as many state-variables and require many more model runs to calibrate and validate.

4.2 Water Quality Models

First, it is important to note that at the present time, there is no coupled hydrodynamic/ nutrient-based eutrophication model of SFB. The most likely reason for this is that, in general, nutrients have not been a factor in limiting phytoplankton growth in the system. Cloern and Dufford (2005) analyzed data for the period 1992 to 2001 and found that nitrogen and silica were limiting only 4% and

1% of the time, respectively, with light limitation occurring 74% of the time. However, a number of researchers studying SFB have also constructed simple analytical, statistical or 1-D mathematical "process" models in an attempt to verify various hypotheses that have been proposed to explain phytoplankton dynamics in SFB. For example, Cloern (1987) used data analysis and analytical solutions to evaluate the effect of turbidity on phytoplankton biomass and productivity in SFB. Subsequently, Cloern (1999) developed a simple index of the relative strength of light vs. nutrient limitation using analytical models that describe growth rate as a function of light and nutrients. This index supported the hypothesis phytoplankton growth in north SFB is light limited; similarly phytoplankton growth in south SFB is largely light limited but there are a few occasions during the spring bloom when nutrients become limiting. Cloern (1991) used a 1-D model (phytoplankton biomass, light, zooplankton and benthic grazing, algal sinking and vertical mixing) to demonstrate the importance of the rate of vertical mixing induced by tidal stirring on phytoplankton bloom dynamics. Cloern (1982) also put together a 1-D model (phytoplankton, zooplankton and benthic grazing) that suggested that benthic filter feeders were controlling phytoplankton biomass in south SFB during the summer and fall. More recently, Lucas et al. (2009) used a pseudo-two-dimensional model (two 1-D, vertically resolved water columns – one deep and one shallow) to look at the importance of lateral transport from the productive shallow water shoals can result in the accumulation of phytoplankton biomass in the adjacent deep, unproductive channel.

Lucas et al. (1999) used the depth-averaged TRIM hydrodynamic model of SFB to evaluate transport-related mechanisms affecting phytoplankton biomass accumulation and spatial distribution on a system-wide level. Based on model simulations they concluded that tidal-timescale processes, both physical and biological, can determine whether a bloom will occur. As was the case with most of the 1-D models described in the previous paragraph, nutrients were not included in the 2-D water quality model framework.

Finally Smith and Hollibaugh (2006), using a box model approach, developed water, salt and nutrient budgets for San Francisco Bay for the time period 1990-1995. Due to organic nutrient data limitations, the nutrient budgets were only constructed for dissolved inorganic nitrogen (DIN) and dissolved inorganic phosphorus (DIP). Based on their analysis, they estimated that wastewater treatment plant effluent accounted for approximately 50% of the nutrient loading to the bay during the winter months and about 80% of the summer loading.

4.3 Water Quality Data

There appears to be a reasonable water column water quality data set with which to parameterize and calibrate a nutrient-based eutrophication model of SFB. Kimmerer (2004) provides an excellent overview of the physical, biological and chemical data available for SFB. Long-term monitoring data are generally available and are good for physical data (temperature and chemistry), biological data (phytoplankton, zooplankton, benthos, and fish) and water chemistry (inorganic nutrients and suspended solids). However, Kimmerer reports that there are some major data gaps (microzooplankton, macrobenthos, and benthic microalgae). A significant portion of the available data is available on-line and has been collected by a number of state (DWR, Interagency Ecological Program (IEP), Department of Fish and Wildlife (DFW)) and federal agencies (USGS and NOAA). A few of these programs contain data sets that go back far in time. For example, the USGS data base contains salinity, temperature, chlorophyll-a, suspended sediment, and inorganic nutrient (NH_3, NO_2, NO_3, PO_4, and SiO_2) data back to 1969 and dissolved oxygen data back to 1971, although there are some gaps in certain years. There are limited measurements of total Kjeldahl nitrogen, dissolved organic nitrogen, total phosphorus in the IEP Bay-Delta monitoring database. A number of references report measurements of primary production in the Bay (Peterson et al., 1975, Cole and Herndon, 1979, Cole and Cloern, 1984,) and model estimates (Caffrey et al., 1998, Jassby et al., 2002, Cloern et al, 2007).

There are also some measurements of sediment nutrient flux and sediment nutrient composition although they are limited in both space and time. The earliest measurements were performed by Hammond et al. (1985) at two stations (one shoal and one channel) in south SFB using in situ flux chambers. They measured sediment oxygen demand (SOD) and fluxes of ammonia and silica, using replicate measurements. Caffey (1995) measured rates of ammonia remineralization, porewater ammonia and solid phase nitrogen and carbon at five sites in south SFB and four in north SFB four times between November 1991 and January 1993. Caffrey et al. (1998) measured weekly benthic respiration at two sites (one shoal and one channel) during the 1996 spring (February-May) algal bloom. Grenz et al. (2000) also reported on weekly benthic SOD and nutrient fluxes of N, P, and Si at two stations in south SFB during the 1996 algal bloom.

A key area where information needed for a water quality modeling of the SFB system is deficient is nutrient loadings. This issue was identified by McKee et al. (2011) in their NNE literature and data gap report. They pointed out "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." (McKee et al., 2011). This is an area where BACWA can provide information concerning their effluent flows and concentrations to at least update one source of nutrients to the Bay. Additional efforts will need to be expended to develop reliable nutrient loading estimates from the Sacramento-San Joaquin Delta, stormwater runoff, exchange with the Pacific Ocean and atmospheric loadings directly impinging onto the water of the Bay.

5. COSTS AND LESSONS LEARNED FROM OTHER U.S. ESTUARINE MODELING STUDIES

Before providing site-specific conclusions and recommendations for SFB, a brief overview of monitoring and modeling costs associated with and lessons learned from other estuarine modeling studies in the United States follows:

- **Chesapeake Bay.** Chesapeake Bay is in addition to SFB one of the most monitored estuaries in the United States. It is certainly one of the most modeled estuaries in the United States if not the world. Modern water quality sampling, supported by the USEPA, began in 1985. The number of sampling stations has varied over the years, but the most recent sampling program (Figure 6) includes about 150 stations (mainstem Chesapeake and the James, York, Rappahannock, Potomac, Patuxent, Patapsco, Chester, Choptank, Nanticoke, Wicomico and Pocomke Rivers that are tributary to Chesapeake Bay – see Figure 6). There are 14 annual cruises, which sample phytoplankton, zooplankton, SAV, benthic invertebrates and water chemistry. Sediment oxygen demand and nutrient flux are also sampled at 10 stations four times a year (seasonally). The current program costs about $3.9 M annually and funding is split 50-50 between the states and federal government. Funding to support watershed and hydrodynamic/ water quality modeling has varied between $1-2.5 M/year over the years. Currently most of the funding for modeling has been allocated to the watershed model and to run management scenarios with the Bay water quality model. **The Bay water quality model has been used to develop nutrient and sediment TMDLs for the Bay and to assist in the development of a Use Attainability Analysis (UUA) for the Bay**, since there are portions of the Bay that do not meet the most protective of the USEPA dissolved oxygen criteria for marine organisms. Model computations and analysis of sediment cores have shown that this is in part due to the particularly deep thalweg that runs up the center of the Bay. **Given its depth and the vertical stratification that occurs during the summer months, it appears that hypoxia is a natural occurrence in this deeper portion of the Bay.**

- **Long Island Sound.** Summer hypoxia is an annual occurrence in the central and western portion of Long Island Sound. During especially stratified years, there are limited occurrences of anoxia in the extreme western Sound. Currently, the USEPA Long Island Sound Study office provides about $850 K/year in funding to monitor water quality in the Sound. The State of Connecticut also provides some costs sharing funds. The monitoring program includes 47 active stations (Figure 7) and sampling includes phytoplankton, zooplankton and water chemistry. The program was initiated in 1991. The development of a Long Island Sound water quality model was begun in 1988 with a laterally-averaged, 2-D segmentation. Further model improvements and refinements were conducted during the early 1990s with increased spatial resolution and coupling to a 3-D time-variable hydrodynamic model of the Sound. The latest version of the model was funded by the NYC Department of Environmental Protection (NYCDEP) and extends the model domain to include the New York Bight. This was done to move the boundary conditions well away from the area of interest in the Sound as well as to permit the NYCDEP to evaluate management alternatives for its 14 wastewater treatment facilities that discharge into the New York Harbor complex. **The Long Island Sound water quality model was used to develop a nitrogen TMDL for the Sound in 2001. The water quality model has also been used as a tool by managers of POTWs in developing a nutrient trading strategy for the Sound.** The water quality model was also used in a re-assessment of the 2001 nitrogen TMDL that is part of the adaptive management plan incorporated within the 2001 TMDL document.
- **Boston Harbor and the Massachusetts Bays System.** As a result of a court-ordered mandate issued in 1991, the Massachusetts Water Resources Authority (MWRA) initiated the construction of a 1,080 MGD secondary treatment facility and a 15.3 km wastewater effluent outfall from Boston Harbor into northern Massachusetts Bays. As part of the treatment plant and outfall permit, MWRA was required to perform water quality monitoring of Massachusetts and Cape Cod Bays to establish ambient water quality conditions prior to the outfall going on-line. The initial sampling began in 1992 and included 21 near-field (outfall) and 22 far-field monitoring stations (Figure 8). Sampling included phytoplankton, zooplankton, water chemistry (at up to five depths in the vertical) and some measurements of primary production, sediment oxygen demand and nutrient flux. Program expenditures for monitoring were between $2.5-3.5 M. In 1994, MWRA provided funding to develop a hydrodynamic/water quality model of the eutrophication processes in Boston Harbor and the Massachusetts Bays system, in particular, to investigate the potential impacts of the relocated Deer Island facility wastewater on Boston Harbor and Massachusetts and Cape Cod Bays. Initially, the model was calibrated to a 3-years pre-outfall data set. During the calibration effort, the model identified the importance of nutrient input in the Massachusetts Bays system from the Gulf of Maine and led to the modification of the existing monitoring program to include a monitoring station near the northeastern boundary of Massachusetts Bay. **A nitrogen mass balance was also conducted that indicated that approximately 92% of the total nitrogen loading to the Massachusetts Bays system was due to import from the Gulf of Maine and that MWRA only contributed about 3% of the total N loading to the Bays.** A yearly update of the Bays Eutrophication Model, as the water quality model is known, is actually written into the

MWRA permit, as is on-going monitoring. Future costs for monitoring are projected to be ~$1.5 M; this is due to a reduction in the sampling plan to 14 stations, sampled nine times/year. Ongoing modeling efforts are about $100 k/year.

6. CONCLUSIONS AND RECOMMENDATIONS FOR DEVELOPMENT OF A SFB MODELING FRAMEWORK

6.1 Conclusions

A strawman Conceptual Model of the SFB system has been developed. The key features or processes of the CM include: freshwater inflows and associated nutrient loadings; estuarine circulation; phytoplankton growth and respiration as affected by temperature, light availability and nutrients; suspended solids, since suspended solids strongly influence light availability for phytoplankton growth; and phytoplankton grazing losses due to zooplantkon and benthic filter feeders. A strawman modeling framework has also been proposed. The components of the modeling framework include: a hydrodynamic model that can predict advective transport and vertical stratification and a nutrient-based eutrophication model that includes a sediment diagenesis and nutrient flux submodel and potentially a suspension feeder submodel.

A generic monitoring program that provides data to parameterize and caibrate and validate both the hydrodynamic and water quality models has also been outlined. While much of the required data are already being collected by ongoing regional monitoring efforts (ex., the USGS monitoring program, which measures temperature, salinity, chlorophyll-a, suspended solids, extinction coefficient, and nutrients baywide, the CA Department of Fish and Wildlife's zooplankton monitoring program, which includes North SFB but does not include Central and South SFB, and the DWR's benthic monitoring program, which includes North SFB but does not include Central and South SFB), additional data collection and monitoring efforts are required. These will be discussed in the recommendations section below.

6.2 Recommendations

It is believed that sufficient information, data, and scientific analyses are available with which to begin the process of model development, calibration and validation. However, it is also believed that this work should be conducted in a phased approach, starting simple, building on existing knowledge and modeling frameworks and continuing to refine the model framework, both ecosystem-wise and spatially over time as our understanding of the Bay continues to improve and as data gaps are filled with additional monitoring and research efforts.

With respect to additional monitoring and data needs, there are some key data that should be collected before proceeding with actual modeling of the SFB system. Perhaps, the most important of these are:

- the collection of effluent water quality parameters (organic and inorganic nutrient forms and suspended solids). While there are some data available for some of the SFB POTWs, most of the effluent data are limited to measurements of inorganic nitrogen. Sampling should be extended to include organic nitrogen and/or total nitrogen as well as the inorganic forms (NH_4 and NO_2+NO_3). Additional data should include BOD_5 and BOD_u and some limited measurements of total phosphorus and PO_4. These data could be collected on a monthly basis and would likely be an inexpensive part of the total data collection/monitoring program.

- zooplankton and benthic biomass monitoring should be extended to Central and South SFB. Zooplankton data (biomass/species composition) could be collected on a monthly basis, while benthic data (biomass/species composition) could be collected on a seasonal to annual basis and would likely be a moderate portion of the annual monitoring program.
- development of a monitoring program to provide estimates of nutrient and chlorophyll-a inputs from coastal waters. This would require utilization of ADCPs at two to three locations near the mouth of SFB in order to develop estimates of net inflow and outflow between coastal waters and SFB. It would also require vertical measurements of nutrients and chlorophyll-a at the locations of the ADCPs over a tidal cycle several times per year. The combination of both the ADCP and nutrient/chlorophyll-a data would then be analyzed in order to develop temporal estimates and timing (i.e., coastal upwelling events) of nutrient and/or chlorophyll-a loadings to the bay from coastal waters. This is likely to be a moderate to high cost of the monitoring program.
- nutrient and flow data that can be used to estimate nutrient loadings from nonpoint source inputs (e.g., stormwater and watershed sources) and to support development of watershed loading models. These data should be collected over a series of storm events, both large and small. This is likely to be a moderate cost item.

Less important, but still relevant, data collection/monitoring efforts would include seasonal measurements of sediment oxygen demand and nutrient flux at several locations within SFB and some measurements of atmospheric nutrient deposition, both dryfall and wetfall.

The first two steps in the phased modeling approach should be: (1) development of a modern, comprehensive estimate of nutrient loadings entering the SFB estuary (recommended data collection efforts necessary to develop these estimates have been listed above), and (2) convening a Modeling Workshop, which would include SFB and nationally recognized scientists and water quality modelers to develop a CM of the Bay (or to validate the one presented above) and to develop a modeling framework and approach for the Bay.

In order to facilitate discussions at the Modeling Workshop, an initial model framework and approach has been developed and is presented below. It is recommended that this framework and approach be used as a “Modeling Strawman” at the Modeling Workshop.

It is a given that calibrating a eutrophication model requires a large number of iterations. This is largely due to the number of adjustable model coefficients that are incorporated in eutrophication models: phytoplankton growth rates and temperature and light optimums, phytoplankton respiration and sinking rates, Michaelis-Menton constants for nitrogen, phosphorus and silica, zooplankton and benthic filter feeder grazing rates, nutrient mineralization and hydrolysis rates, reaeration rates, suspended solids settling and wind-induced resuspension rates, etc. While, previous work and 1-D process models of SFB that have been reported in the literature may provide some guidance for the choice of algal growth and respiration rates and zooplankton and benthic filter feeder rates, it is likely that model calibration will still require a large number of trial-and-error simulation runs. If one were to utilize a very highly resolved spatial resolution grid of the SFB system, the associated runtimes for the water quality model may significantly lengthen the time required to calibrate the water quality model. Therefore, it seems prudent to start the modeling effort with a fairly simple spatial grid. It is recommended that the Uncle-Peterson (U-P) model and model grid (Figure 4a) be used as the basis for the eutrophication model. The structure of the U-P model, with its ability to represent the SFB shoal and channel areas would allow it to represent the effects of benthic filter feeder grazing (shoals) and potentially the importance of tidally driven stratification and mixing in the deeper channel portions of the eutrophication model. The fact that the U-P model is well calibrated should enable it to be extended to other years of interest or conditions of interest to the water quality modelers. The fact that it has been used for sediment transport calculations is also a benefit, as output from the sediment transport component of the U-P model could be used to help calibrate the suspended solids state-variable in the water quality model that would be used to develop spatially - and time-variable extinction coefficients key to the eutrophication model (used to determine light attenuation).

The framework proposed for the water quality or eutrophication model (Figure 3) includes state-variables for phytoplankton biomass as carbon and chlorophyll-a (potentially multiple groups based on seasonal or nutrient related processes), various forms of nutrients (dissolved organic, particulate organic, and inorganic) N, P, and Si, detrital organic carbon (as detrital carbon and phytoplankton carbon affect dissolved oxygen concentrations), dissolved oxygen, suspended solids and zooplankton and the effects of benthic filter feeders on phytoplankton biomass. While a few existing eutrophication models do include benthic filter feeders as a state-variable, it is recommended that initially they not be included directly as state-variable, but rather be treated as an exogenous variables or forcing function based on observed biomass data and literature-based filtering rates. The proposed model framework could be further simplified by eliminating the phosphorus system. A rationale for this is based on an analysis of SFB inorganic nitrogen, phosphorus and silica data for the period 1992 to 2001 performed by Cloern and Dufford (2005). Their analysis indicated that phosphorus was not a limiting nutrient in SFB, while nitrogen and silica were found to be limiting 4% and 1% of the time, respectively with light limitation occurring 74% of the time. However, as reported by Cloern et al., (2006) turbidity has been decreasing in the SFB system and, therefore, light limitation appears to be decreasing and it is possible that nutrient (nitrogen or silica) limitation may occur in the future on a more frequent basis. A potential reason for leaving the phosphorus system in the model framework concerns the potential adaptation of a uniform modeling framework for SFB and the Sacramento-San Joaquin Delta in the future. It is in the Sacramento-San Joaquin Delta where phosphorus controls may be of interest to water quality managers.

A recommended approach for calibration of the water quality model would be to select two or three recent years with differing hydrologic conditions (i.e., wet, dry and average) and perhaps one or two historical years (perhaps the early to mid-90s) that represent different levels of benthic suspension feeders for model calibration. The reason for recommending this approach is to investigate the ability of the model to differentiate phytoplanton biomass under differing residence times and affects of benthic filter feeders. After completing the loading analysis for these years, the model could be run and output evaluated to see how the model performs spatially (north SFB vs. south SFB) and temporally against observed phytoplankton biomass (chlorophyll-a) and water column concentrations of inorganic nutrients. A calibration for sediment oxygen demand and nutrient flux will be less rigorous due to the paucity of data, but should still be evaluated as part of the overall calibration process. The evalution of the model's calibration of inorganic nutrients would help provide confidence that the nutrient loading estimates are valid, while the evalution of model calibration of chlorphyll would help provide confidence that the significant processes (light and grazing pressure) that affect phytoplankton biomass (and ultimately dissolved oxygen) are well represented.

Models outputs of phytoplankton biomass, dissolved oxygen, nutrients and light attenuation could be used either directly as recommended primary indicators (phytoplankton, dissolved oxygen) or as secondary indicators (light) or post-processed to provide information to formulate emphirical relationships or logistic regression models between nutrients and indicators/co-factors (e.g., nutrients and chlorophyll-a to ephiphytes and light attenuation to SAV or nutrients to chlorophyll-a and light attenuation and together with water temperature and vertical stratification to risk of HAB bloom) as identified in the NNE literature review and data gaps identification document (McKee et al., 2011).

Once initial calibration efforts are successful, the spatial resolution of the Bay can be improved by moving to the more spatially-refined 3-D hydrodynamic models of SFB. Additional processes or state-variables (benthic filter feeders and/or SAV) can be incorporated into the eutrophication model framework as additional monitoring and field and laboratory studies are completed.

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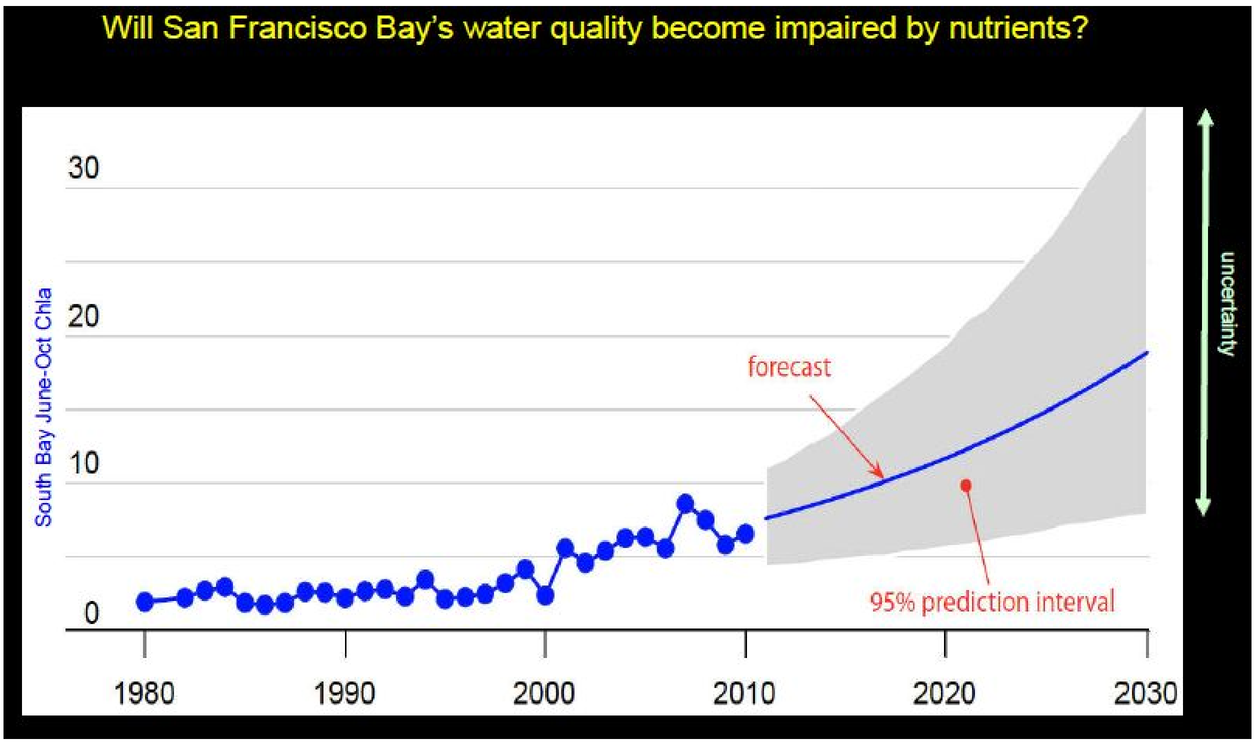

Figure 1. Long-Term Trend in Chlorophyll-a in South San Francisco Bay and Future Trend (from Cloern (2011)

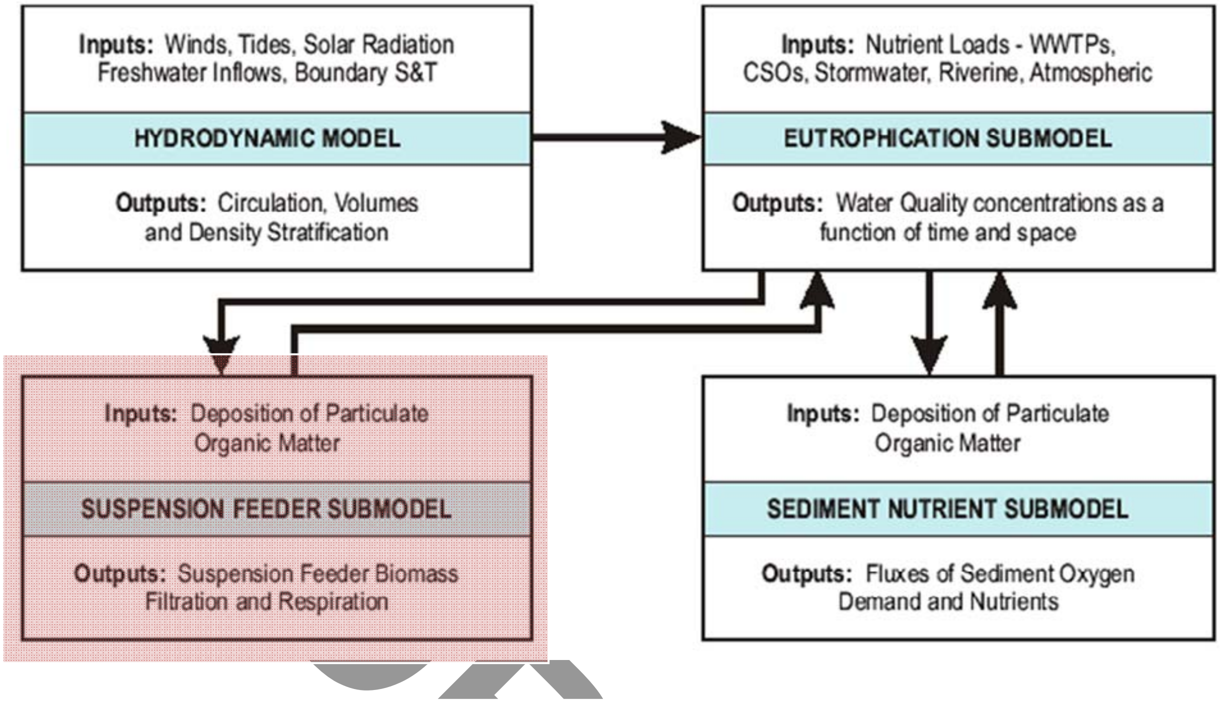

Figure 2. Modeling Framework

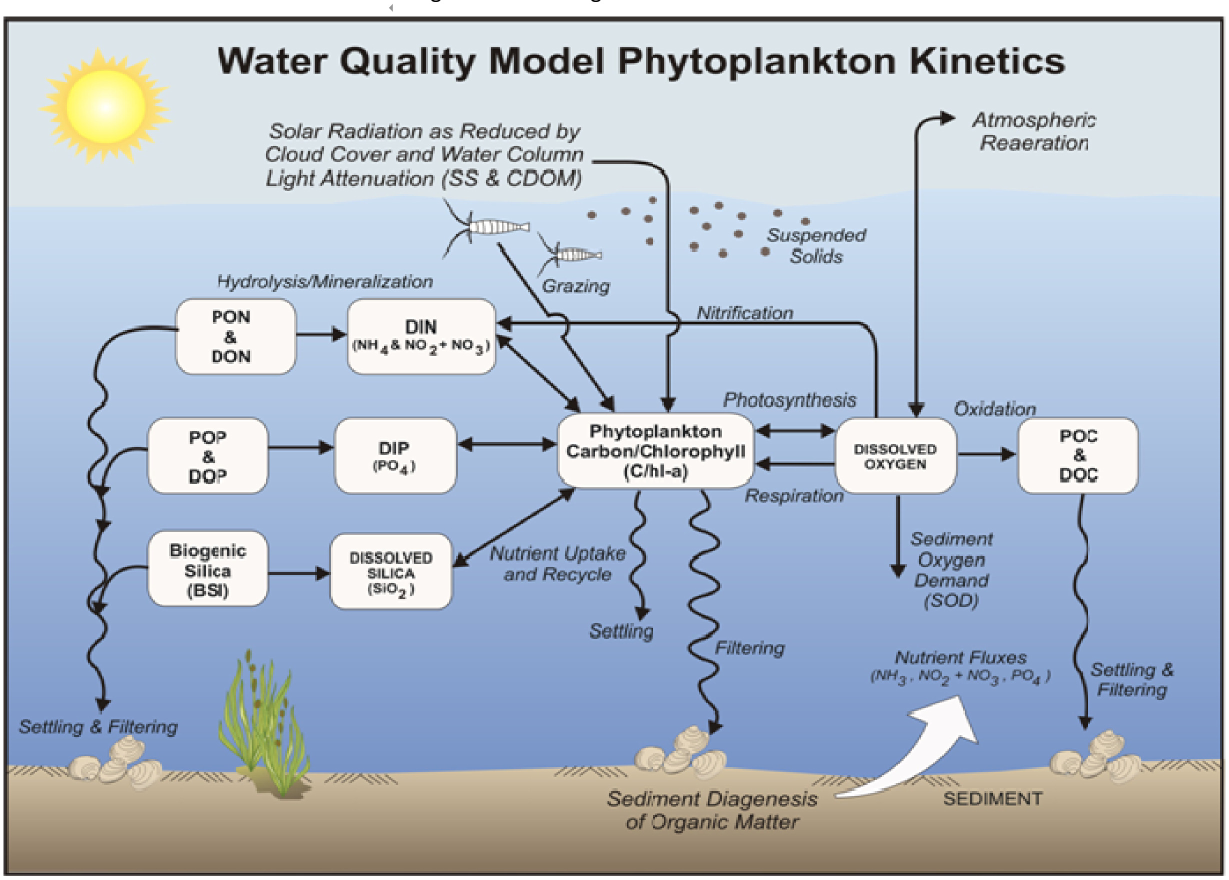

Figure 3. Eutrophication Modeling Framework

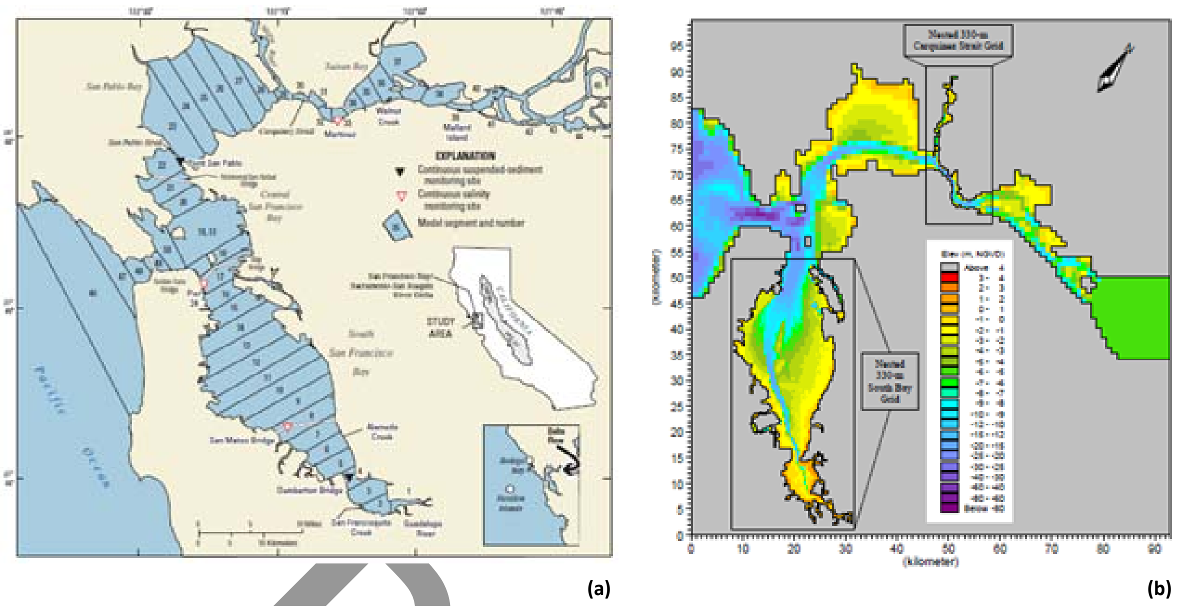

Figure 4. Two-dimensional hydrodynamic models of San Francisco Bay: (a) Uncles-Peterson and (b) Mike-21

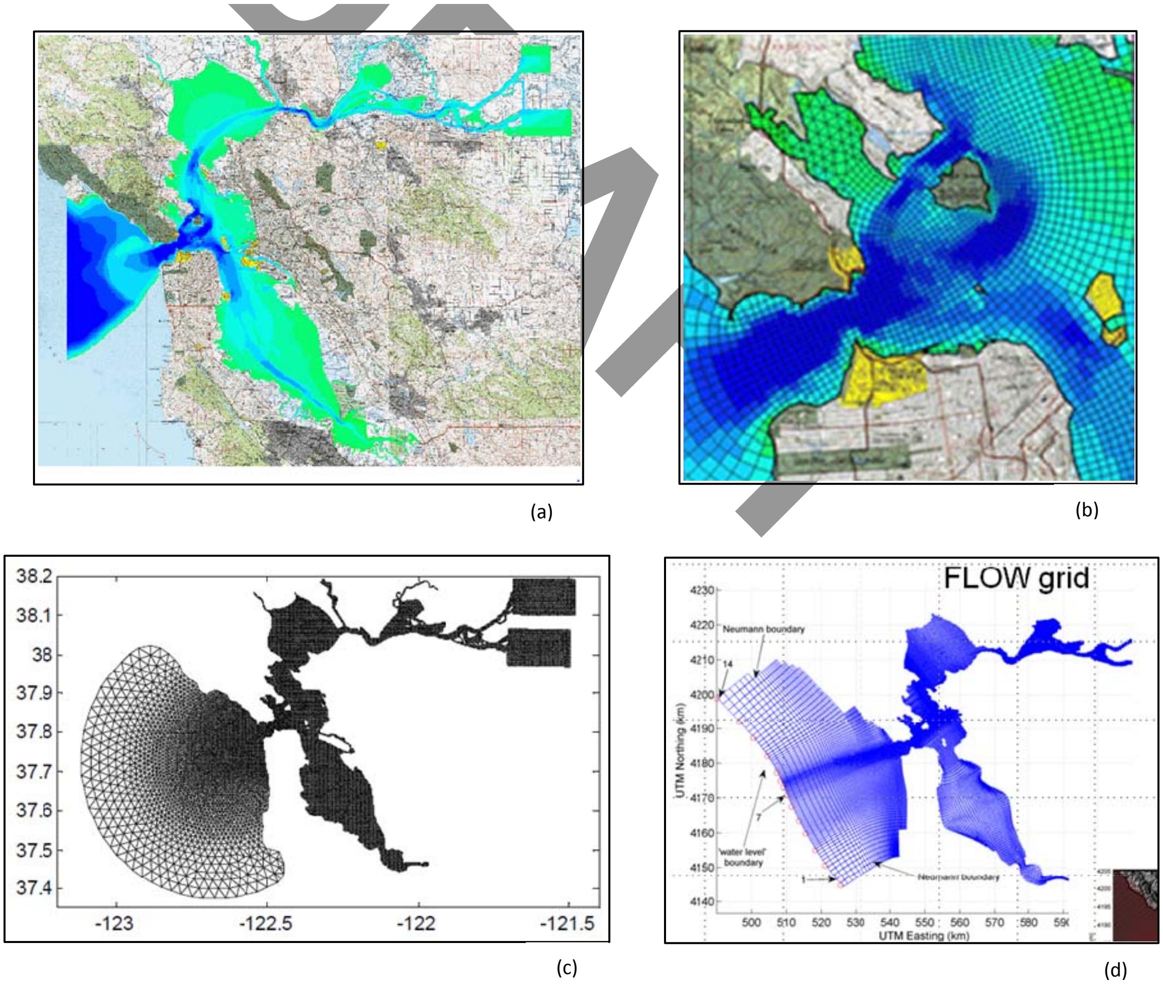

Figure 5. Three-dimensional hydrodynamic models of San Francisco Bay (a) TRIM/TRIM3D, (b) UNTRIM (vicinity of Golden Gate Bridge, (c) SUNTANS, (d) Deltf3D-FLOW.

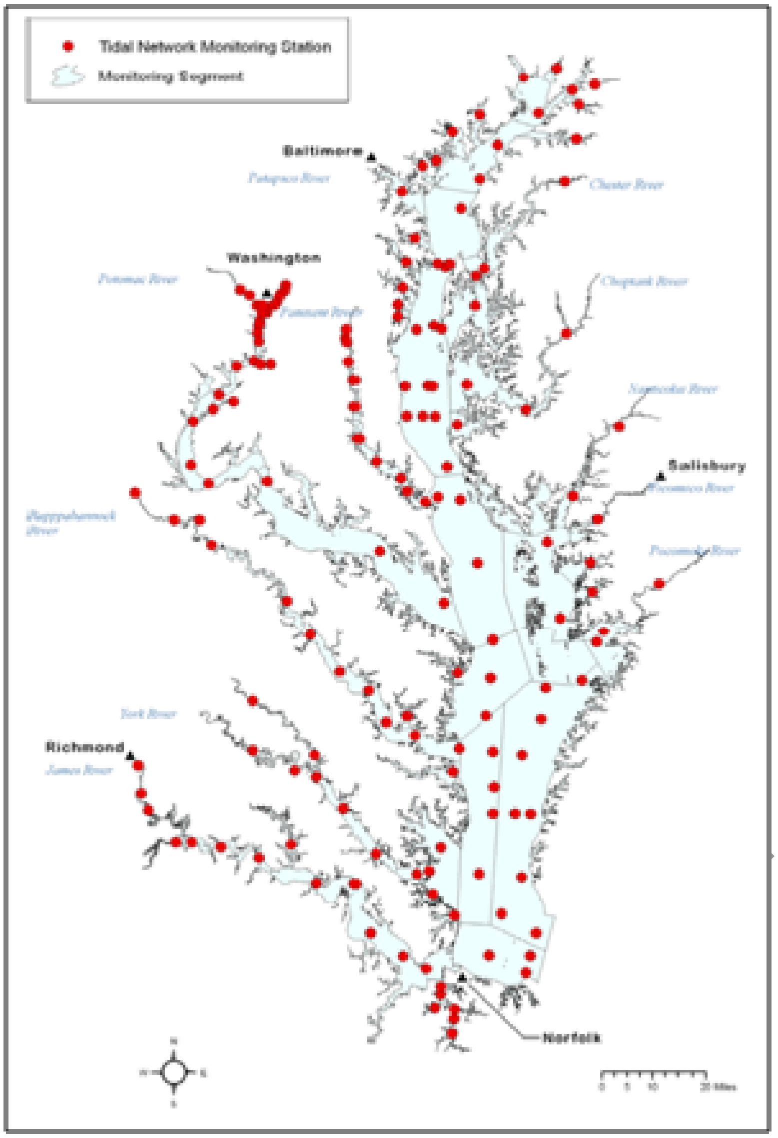

Figure 6. Chesapeake Bay

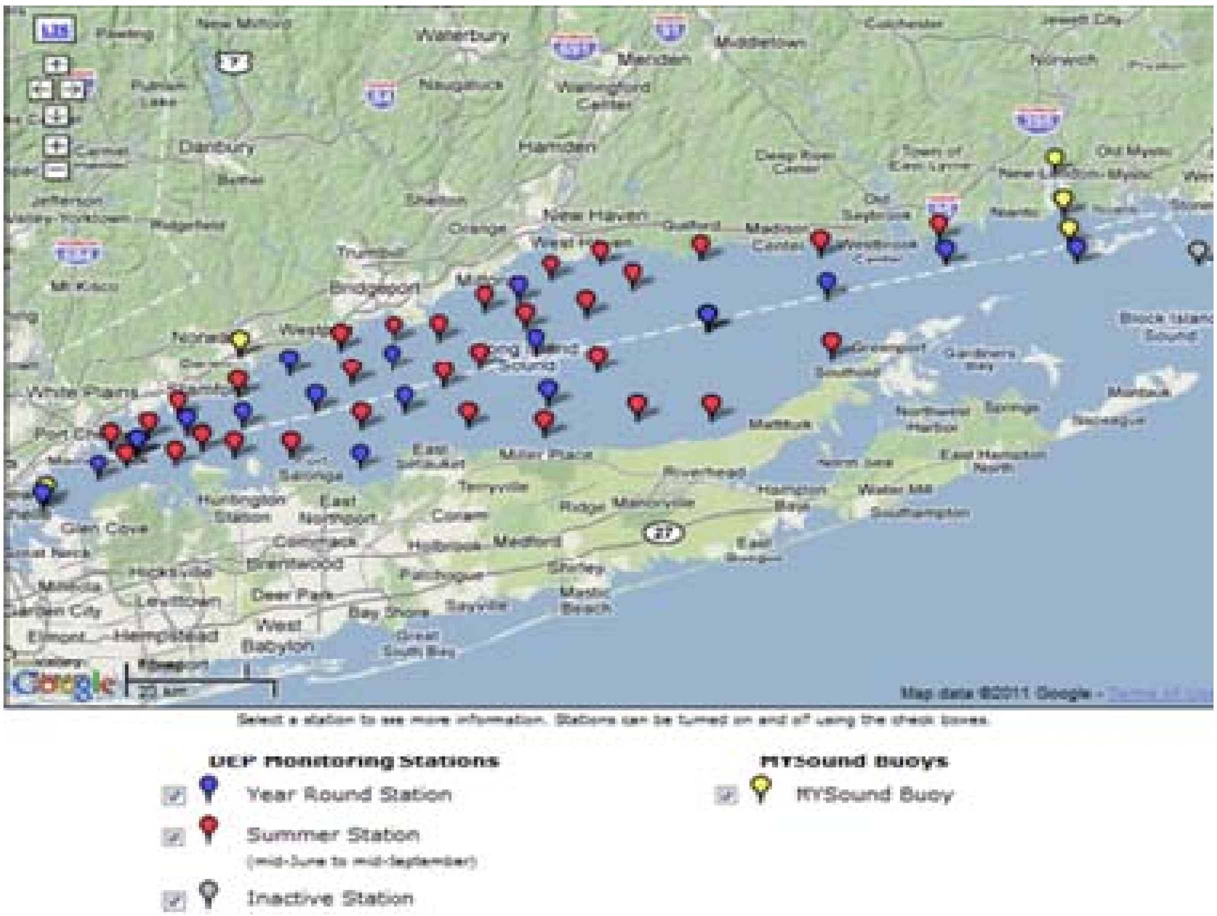

Figure 7. Long Island Sound

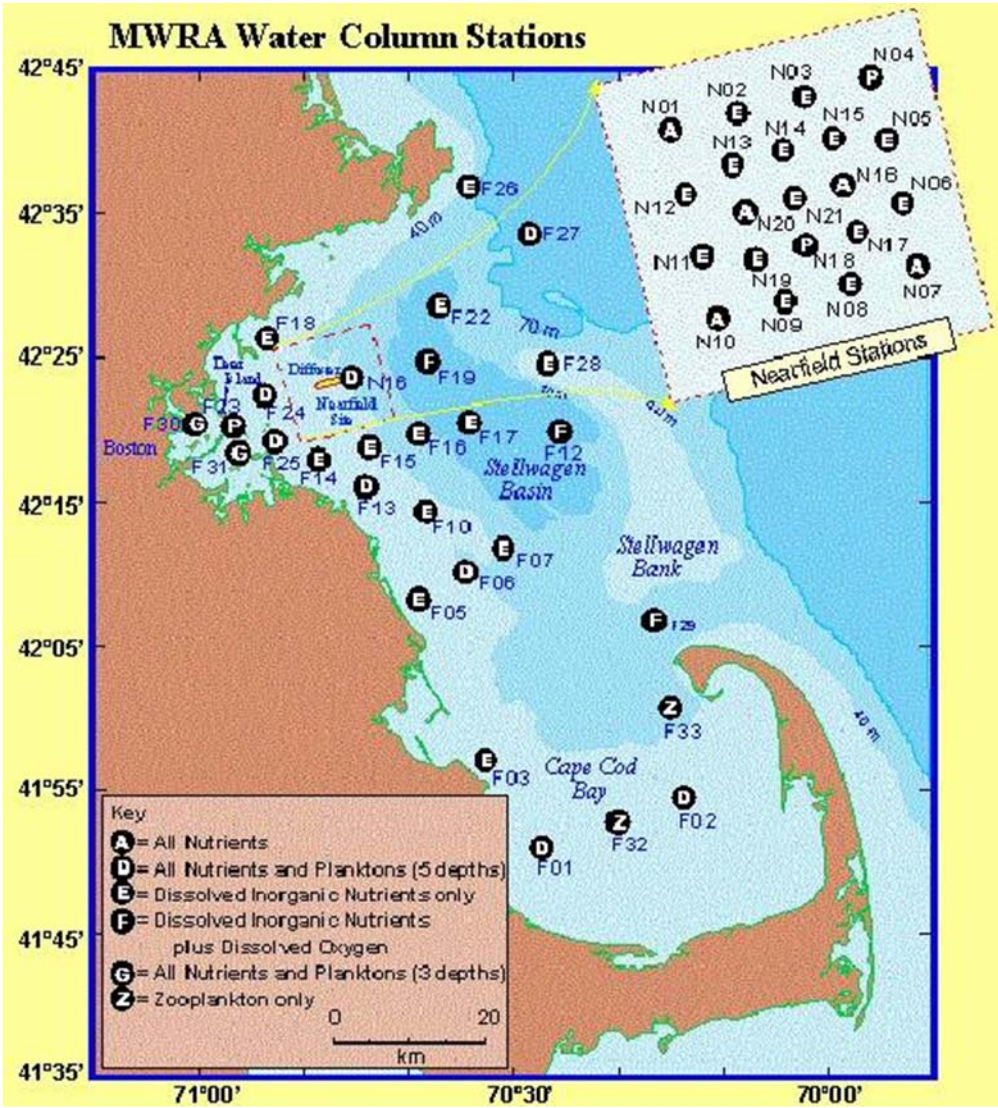

Figure 8. Massachusetts and Cape Cod Bays