Science Supporting Decisions on Management of Eutrophication in Elkhorn Slough Estuary

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Southern California Coastal Water Research Project
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EXECUTIVE SUMMARY

The Elkhorn Slough estuary, a National Estuarine Research Reserve (NERR), is listed as impaired on the State of California's 2018 303(d) list for low dissolved oxygen, as well as other constituents (pH, nitrate) related to eutrophication. The Central Coast Regional Water Quality Control Board (Water Board) is developing a total maximum daily load evaluation (TMDL) to address these issues. To support this TMDL, a conceptual model of eutrophication and a synthesis of the scientific basis for biostimulatory targets was conducted. A watershed loading model and receiving water hydrodynamic and water quality model were developed and calibrated. Receiving water models predicted key indicators of eutrophication, including benthic algal abundance (as macroalgal biomass) and phytoplankton biomass (as chlorophyll-a) and its impact on the diel variability and mean dissolved oxygen (DO), given "biostimulatory conditions" including total nitrogen (TN) and phosphorus (TP) concentrations, sediment oxygen demand, and other site-specific factors. These models were used as the toolkit to conduct linkage analyses and determine what implementation actions can be taken to achieve those targets, including establishing loading limits and restoration to address biostimulatory conditions.

This report synthesizes information from those investigations to support stakeholder engagement with the Central Coast Water Board to identify actions to improve beneficial use support in the estuary.

Major Findings

Synthesis of Information to Inform Biostimulatory Targets. Decades of research of the NERRS staff and collaborators provided the foundation for a conceptual model of eutrophication in the Slough, from which we chose indicators of eutrophication relevant for salt marsh, mudflats, and subtidal habitat. These consisted of primary indicators (macroalgal biomass, phytoplankton chlorophyll-a, sediment nitrogen, surface water DO and pH) and supporting indicators (dissolved inorganic nutrients, HAB toxins and macroalgal % cover). We reviewed the science supporting selection of thresholds, based largely on the work of Sutula (in prep) and literature cited therein.

Central Coast Water Board staff used this review as the basis for selecting a set biostimulatory numeric targets that they are considering using for the TMDL. We applied these targets to 12 years of NERR water quality monitoring data. The demonstration found that every subregion of the Slough systematically failed to meet most targets for most years, with conditions in the tidally restricted areas worse than in unrestricted areas. Data were lacking to evaluate sediment TN and macroalgal biomass.

Indicator	Metric	Type	Threshold	Temporal statistic for a segment
DO	Concentration	Primary	7 mg/L	10th percentile of 7 day mean daily DO minima
pH	Unitless	Primary	7.8	10th percentile of 7-day mean daily pH minima
Macroalgae	Biomass	Primary	30 g dw m^{-2}	Mean of highest 2 consecutive monthly periods or annual maximum if based on bimonthly data
	% Cover	Screening	20 %	
Microalgae	Chlorophyll-a	Primary	15 µg/L	90th percentile of annual monthly samples
Sediment Quality	%TN	Primary	0.1%	Multiple sites within a segment, once per year
	aRPD	Screening	> 2 cm	Geomean of benthic camera images
Dissolved Inorganic Nutrients	Nitrate+ Ammonia	Screening	0.5 mg/L	90th percentile of monthly samples over a year
	Ammonia	Primary	0.1 mg/L	
	Phosphate	Screening	0.09 mg/L	

Modeling Analyses Link Nutrient Loading and Biostimulatory Conditions to Harmful Algal Blooms and Hypoxia. Modeling analyses identified that: 1) inorganic nutrient loads are driving dense macroalgal blooms in the estuary, and 2) macroalgal blooms are the primary driver of DO diel swing and also contribute to sediment organic matter accumulation, which is another important sink for DO in the system. Published studies have documented that DO conditions are contributing to stress and lower survival in fish and oysters in the Elkhorn Slough. Macroalgal blooms are smothering salt marsh, oyster reed and seagrass habitat and causing weakening and erosion of marsh banks, decreasing recruitment and survival of benthic invertebrates in mudflats and subtidal habitat, reducing carrying capacities for fishes and shorebirds, and creating poor aesthetics that can negatively affect ecotourism in this NERR. Modeling analyses do not extend specifically to toxic HABs, but ample published literature links nutrient loading and eutrophication in the Salinas River valley to cyanotoxin related disease and mortality in sea otters and California sea lions.

While Modeling Analyses Suggest N Loading is Dominated By Natural Ocean Sources, Chronic Macroalgal Blooms Are Likely Fueled By Slough Sediments Enriched In Nitrogen. The Slough serves as a reactor that converts inorganic N to organic material (primarily as macroalgal biomass). As these extensive macroalgal blooms senesces, this N-rich algal tissues replenishes the supply available from the sediment, thus creating a chronic, self-sustaining eutrophication problem. In tidally mixed areas of Elkhorn Slough, the dominant source of TN and TP is Monterey Bay (~60-77%). Watershed and releases from the sediment (sediment diagenesis) contribute roughly an equivalent amount of inorganic N while sediment diagenesis is the dominant controllable source of TP (36%). Within tidally restricted areas, sediment diagenesis becomes the major source of TN and TP (~65%). Macroalgal blooms divert N away from denitrification (the only permanent means to eliminate nitrogen from the system), setting up a cycle of macroalgal sequestration and regeneration that can cause rapid accumulation of sediment N over time. Furthermore, attached Ulva mats have been shown to drive concentrations gradients across the sediment water interface that promote the efflux of nutrients, accelerating macroalgal growth. Thus, while Bay source of N appear to dominate the whole estuary budget, macroalgal blooms can begin and thrive solely on the basis of enriched sediment nitrogen sources. Tidal restrictions such as dikes, or weirs can exacerbate this problem by limiting scouring of macroalgae from the estuary and increasing hydraulic retention time.

Management Recommendations Include Nutrient Load Reductions and Restoration to Mitigate Eutrophication In Elkhorn Slough. In tidally mixed areas of the Slough, a 50% reduction of controllable sources is needed, equivalent to a loading capacity (all sources) of 7,463 tons/year of TN and 425 tons/year of TP. In tidally restricted areas, a 92% reduction in controllable sources would be needed if no restoration actions are taken. In addition to nutrient load reductions, four types of actions could directly address biostimulatory conditions (hydromodification and physical habitat alteration) and would likely improve eutrophication symptoms: 1) Improve tidal exchange and circulation to reduce hydraulic retention times and improve ability of estuary to flush out fine grained sediments; 2) Increase the area of intertidal habitat, in order to enhance wetting/drying that can drive greater rates of denitrification; 3) Removal of sediments high in sediment nitrogen, particularly those in tidal restricted areas and 4) Add buffers of riparian or other native vegetation designed to decrease nutrient loading from upland areas. These actions are entirely consistent with the Elkhorn Slough Tidal Restoration Project goals that are intended to increase ecosystem resilience to sea level rise and have been vetted with the local community.

Address Key Scientific and Modeling Uncertainties to Improve Capacity to Adaptively Manage Slough Over Time. The watershed loading and Slough water quality models are imperfect representations of the real world and should be considered as one line of evidence to combine with direct observation and scientific understanding to evaluate management plans. The scientific evidence of models and published studies point towards a compelling and immediate need to reduce nutrient loading to the Slough, while supporting the ongoing effort to restore it via hydrological and physical habitat restoration. Multiple data gaps exist that limit model applications. We recommend addressing these fundamental data gaps (see below) and refining coupled hydrodynamic and water quality models of the Slough, with the intent of improving these tools to adaptively manage the estuary over time.

Uncertainty Theme	Recommendation
Improve monitoring of eutrophication symptom	• Conduct comprehensive monitoring of macroalgal biomass in intertidal and subtidal habitats, conducted in conjunction with cost-effective monitoring (e.g., via drones) of macroalgal cover to fine tune its use as a screening level monitoring indicator. • Conduct monitoring to link sediment total nitrogen, total organic carbon, and grain size to macroalgal blooms and macroinvertebrate community composition. • Routinely document algal toxins, including both cyanotoxins and marine biotoxins in particulate suspended matter and in shellfish. • Document evidence of biological impairment from acidification.
Quantify eutrophication drivers	• Quantify groundwater contributions to nutrient loading of Elkhorn Slough. • Improve quantification of freshwater and nutrient loads to Elkhorn Slough surface water sources (Table 3.6), including improved understanding of nutrient contributions from agricultural land use types. • Quantify the contribution of anthropogenic nitrogen (e.g., from San Francisco Bay and Monterey coastal sources) that represent "ocean sources" of nutrients to Elkhorn Slough. • Quantify benthic nutrient and oxygen fluxes in the habitat types of Elkhorn Slough.
Improve modeling capabilities to simulate eutrophication from global and local drivers as well as management initiatives	• Assess the potential for seagrass restoration as a function of climate change (sea level rise) and eutrophication. • Quantify the effect of hydrological and physical habitat restoration in nutrient loading reduction required to meet biostimulatory targets.
Improve understanding of efficacy of management measures	• Establish the quantitative basis for a nutrient trading scheme based on in -kind (nutrients) or restoration as an alternative measure. • Quantify efficacy of best management practices and other nutrient load reduction measures, particularly for agricultural land uses.
Improve biostimulatory targets	• Improve the basis for compliance with dissolved oxygen objectives, with emphasis on spatial and temporal aggregation.

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1. INTRODUCTION

1.1 Purpose of Document

Eutrophication of estuaries and coastal waters, defined as "the accelerated accumulation of organic matter in aquatic habitats" (Nixon 1995), is a global environmental issue, with demonstrated links between anthropogenic changes in watersheds, altered freshwater flow and nutrient loading to coastal waters, hydromodification, and physical habitat change, all of which result in a suite of adverse ecosystem responses. These ecological impacts including harmful algal blooms, oxygen loss, acidification, and impacts on aquatic food webs (Valiela et al. 1992) can have far-reaching consequences, including fish-kills and lowered fishery production (Glasgow and Burkholder 2000), loss or degradation of seagrass beds (Twilley et al. 1985; Burkholder et al. 1994; McGlathery 2007), smothering of bivalves and other benthic organisms (Rabalais and Harper 1992), nuisance odors, and impacts on human and marine mammal health from increased frequency and extent of harmful algal blooms and poor water quality (Bates et al. 1989; Trainer et al. 2002). While nutrient pollution (including forms of nitrogen and phosphorus) is the leading cause of eutrophication, other factors can cause or significantly contribute to eutrophication. These factors include changes associated with conversion of natural landscapes to developed land uses, such as hydromodification, altered physical habitat, water temperature, and light availability, grazing pressure, etc. (Paerl et al. 2011).

The Elkhorn Slough estuary, a National Estuarine Research Reserve, is listed as impaired on the State of California's 2018 303(d) list for low dissolved oxygen, as well as other constituents (pH, nitrate) related to eutrophication. The Central Coast Regional Water Quality Control Board (Water Board) is developing a total maximum daily load (TMDL) to address these issues. To support this TMDL, a watershed loading model capable of predicting the runoff of freshwater flow, nutrients, and suspended sediments was developed and calibrated (Sakar et al. 2022). This model predicts the natural and human sources of nutrients to a newly developed and calibrated Elkhorn Slough estuary coupled hydrodynamic and water quality model (Butcher et al. 2022), which can simulate the response of the estuary to the influence of the watershed and the coastal ocean of Monterey Bay. At the same time, the State Water Board is developing an amendment to Water Quality Control Plan (SB Enclosed Bays & Estuaries) to update its biostimulatory water quality objective (WQO)[1]. While the policy is not anticipated for several years, available science has been synthesized to support that policy and to also develop appropriate biostimulatory numeric targets for Elkhorn Slough. These numeric targets help define healthy aquatic habitat and will be used as the goals from which to derive the TMDL, to identify what implementation actions can be taken to achieve those targets, including establishing load and waste load allocations.

This report summarizes technical activities to support the development of the TMDL by synthesizing literature to support decisions on biostimulatory targets for the estuary and demonstrating their application on existing monitoring data (Chapter 2) and applying the watershed loading and estuary water quality models to develop the scientific basis for total maximum daily loads (Chapter 3).

[1] The Water Board defines Biostimulatory substances (nitrogen and phosphorus) and conditions (hydromodification, temperature and habitat alteration, etc.) as those that contribute to the problem of eutrophication. Biostimulatory objectives are the water quality goals that protect beneficial uses against eutrophication.

1.2 Background

Elkhorn Slough, an estuary draining to Monterey Bay, is located in Monterey County, CA (Figure 1 1). Elkhorn Slough contains the largest tract of tidal salt marsh in California south of the San Francisco Bay and provides much-needed habitat for hundreds of species of plants and animals. The Slough was historically part of an interconnected, extensive estuary that included Elkhorn Slough proper, Bennett Slough, Moro Cojo Slough, Tembladero Slough, and the old Salinas River Channel (Figure 1.2). Connections to these channels still exist. The Elkhorn Slough channel is the largest of the channels and is the only one not obstructed by a water control structure at its mouth, and therefore the entire complex is generally referred to as the Elkhorn Slough estuary (Hughes et al. 2011). Now, all arms of the historic estuary except for the main Elkhorn Slough channel are diked and function mostly as freshwater impoundments. Thus, historic estuarine biodiversity is now only represented in this small area. As such, Elkhorn Slough is the only remnant of the historic estuary with abundant sea otters, eelgrass beds, fish nursery habitat, migratory shorebird foraging, and other numerous ecological habitats, which provide key ecosystem services to the region (Figure 1.3).

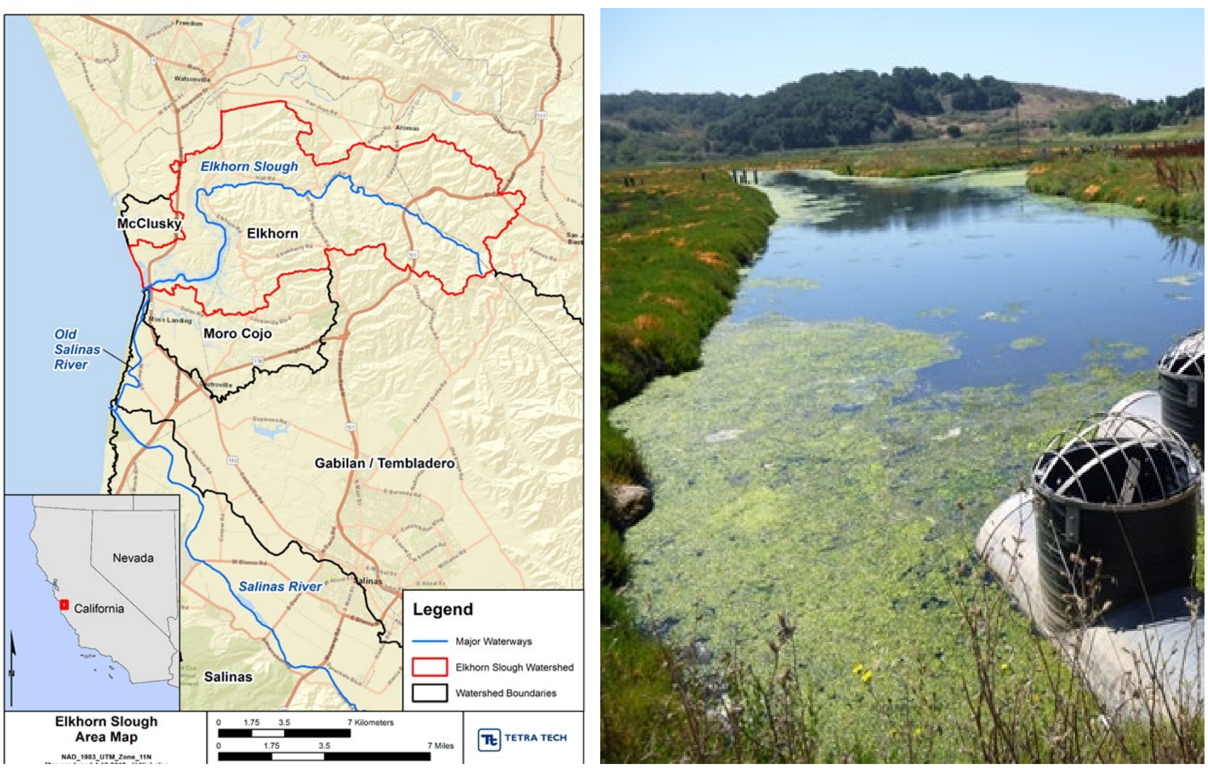


Figure 1.1. Left Panel: Location of Elkhorn Slough estuary and its contributing watershed. Right Panel: Macroalgal blooms in the Slough (Source: Kerstin Wasson).

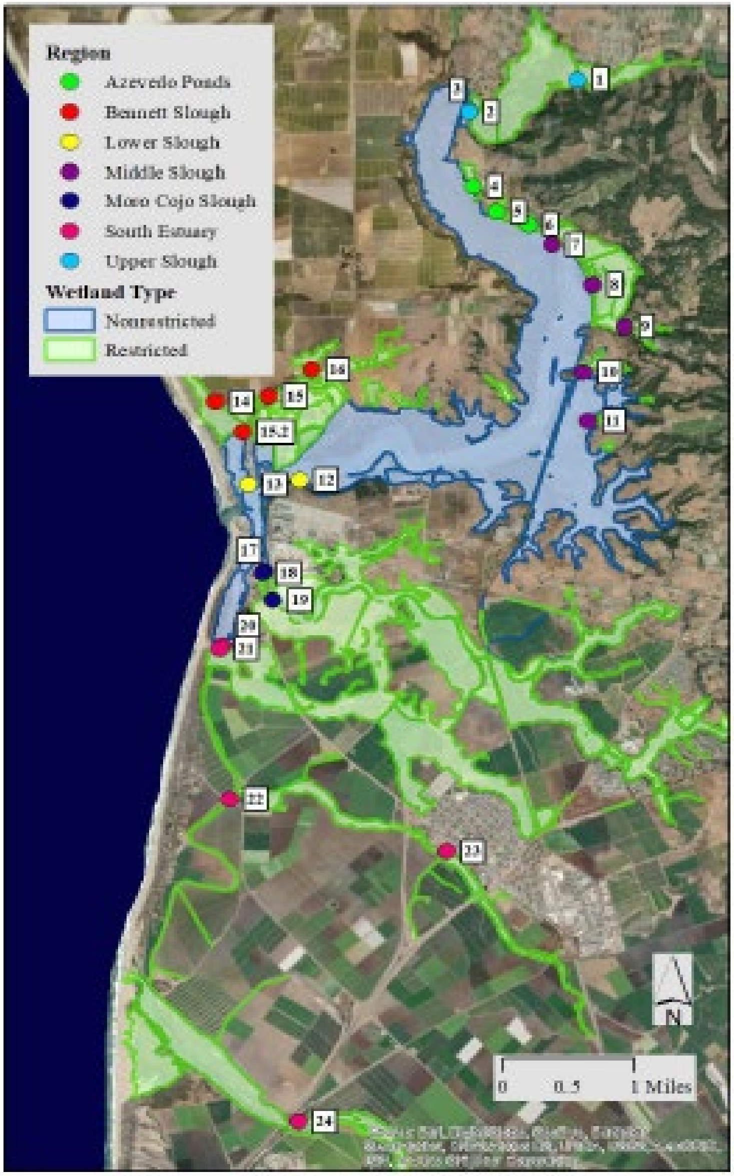


Table 1: Site codes and corresponding names of locations

ID	Site Name
1	Carneros Creek
2	Blohm Porter Marsh
3	Hudson Landing
4	Azevedo Pond, North
5	Azevedo Pond, Central
6	Azevedo Pond, South
7	Kirby Park
8	Reserve North Marsh
9	Strawberry Road
10	Whistlestop Lagoon
11	South Marsh
12	Vierra
13	Skipper's Landing
14	Bennett Slough, West
15	Bennett Slough, East
15.2	Jetty Road
16	Struve Pond
17	Moss Landing Road, North
18	Moss Landing Road, South
19	Moro Cojo Slough
20	Potrero Road, North
21	Potrero Road, South
22	Monterey Dunes Way
23	Tembladero Slough
24	Salinas River Bridge

Figure 1: Map of estuarine wetlands and significant landscape features. Elkhorn Slough National Estuarine Reserve monitoring sites are labeled.

Figure 1.2. Map of Elkhorn Slough habitats and adjacent waterbodies that influence eutrophication.

While Elkhorn Slough is the only undiked arm of the historic estuary, many of its peripheral wetlands have also been diked (Van Dyke and Wasson 2005). It has become increasingly clear that water quality and biodiversity are very different in the diked vs. undiked portions of the estuary (Ritter et al. 2008; Hughes et al. 2011). For this reason, restoration of tidal circulation in Elkhorn Slough is the major focus of conservation and restoration efforts. Agriculture is important in the Elkhorn Slough watershed, and erosion of sediments off steep adjacent farm fields was one conspicuous issue recognized decades ago. Reducing nutrients and sediment entering from adjacent farms has been a key priority motivating land acquisition and restoration by the Elkhorn Slough Foundation (Scharffenberger 1999), and local improvements have been documented as a result (Gee et al. 2010).

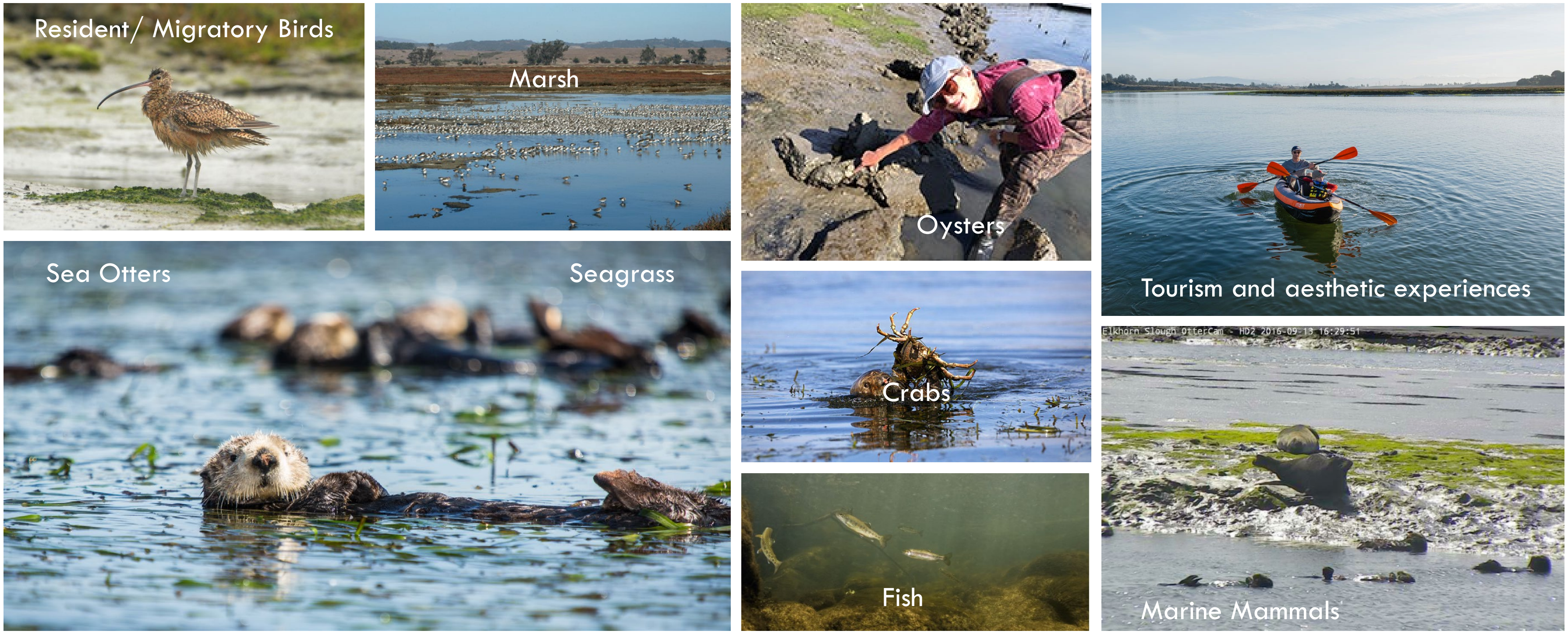


Figure 1.3. Examples of Elkhorn Slough prized habitats and inhabitants that represent the beneficial uses that the TMDL aims to address.

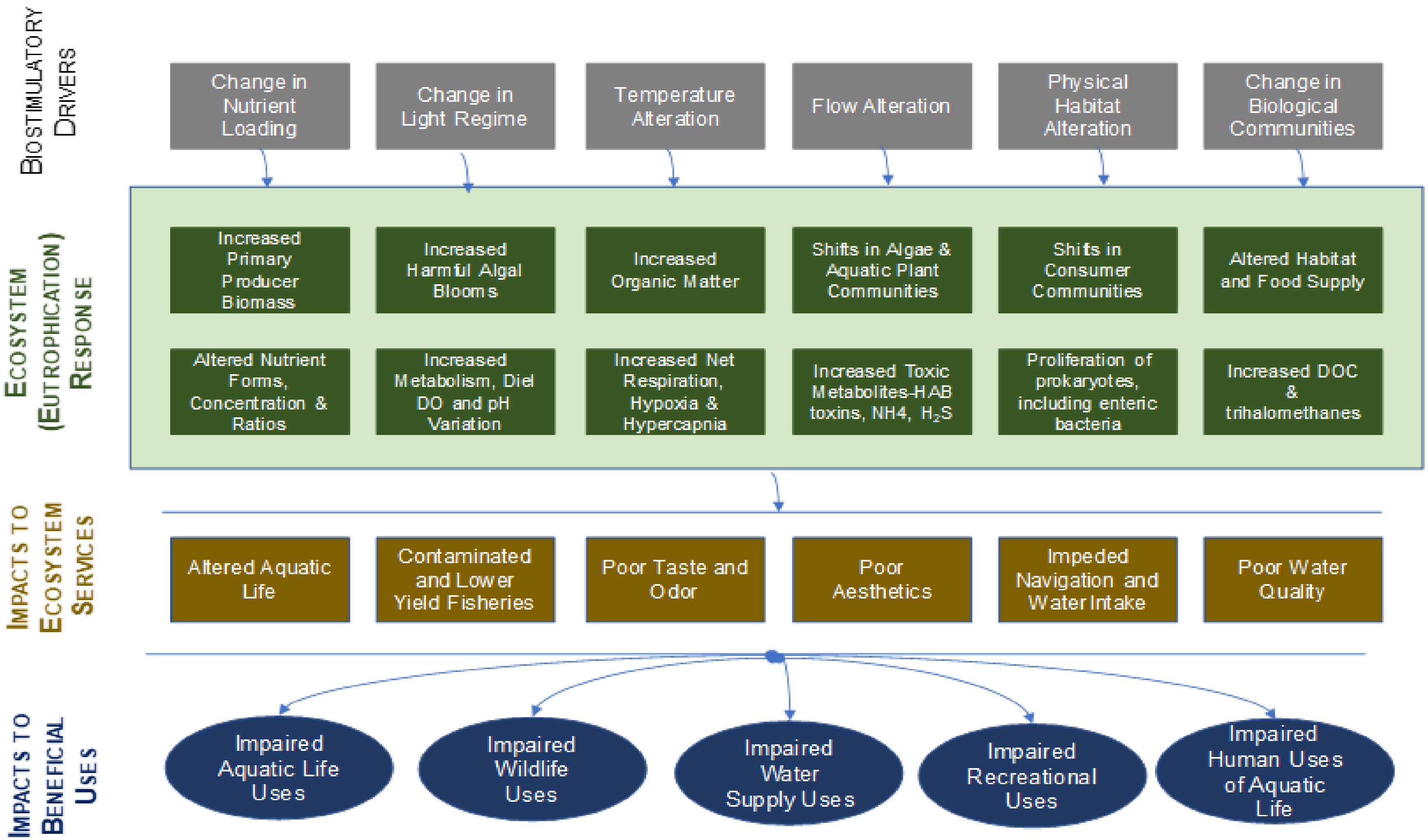


Figure 1.4. Conceptual representation of biostimulatory conditions and substances that result in eutrophication, which impacts ecosystem services and impair beneficial uses in water bodies.

Surrounding agricultural practices and tidally driven processes leading to nutrient loading have heavily influenced the estuary over the past 70 years, and long-term data suggest that nutrient levels have increased to the point that the estuary is home to some of the highest dissolved nutrient levels in comparison to other U.S. estuaries. Elkhorn Slough experiences routine microalgal and macroalgal blooms, which in turn are linked to increased fluctuations in pH, high occurrence of sulfate reducing bacteria, and chronic periods of daytime hyperoxia and nighttime hypoxia and anoxia. Over the decades, an accumulation of organic matter in sediments has altered benthic habitat quality. Hydrologic alterations in Elkhorn Slough, including dikes, culverts, and tide gates have also caused artificial dampening of the tidal range upstream of water control structures (Hughes et al. 2011). Extensive monitoring and research related to nutrient biogeochemical cycling has been conducted in the estuary over the past two decades, with several peer-reviewed papers identifying key nutrient-related processes (e.g., Caffrey et al. 2002; Hughes et al. 2011; Hughes et al. 2015; Jeppesen et al. 2016; Wasson et al. 2017). A summary of the data collected across the estuary is presented in a recent report by the Central Coast Water Board, including identification of relationships between different parameters, and description of temporal and spatial trends (Saiz and Keeling 2016). In addition, an interactive "Water Quality Report Card" has been published by the Elkhorn Slough Reserve, identifying sub-regions within the slough by their water quality (http://elkhornslough.org/water/). This report card consistently has graded Elkhorn Slough water quality as poor condition over the past decade.

1.3 Policy Framework: Biostimulatory Substances and Conditions

All California Regional Water Quality Control Boards (Regional Water Boards) have a narrative biostimulatory objective, e.g., "waters shall not contain biostimulatory substances in concentrations that promote aquatic growths to the extent that such growths cause nuisance or adversely affect beneficial uses" (Water Quality Control Plan for the Central Coastal Basin (Basin Plan) 2019); similar narrative language is used throughout California Water Board Basin Plans. While narrative biostimulatory objective theoretically covers a wide range of environmental drivers, no statewide consistent guidance exists to interpret this narrative objective to prevent eutrophication in specific waterbodies or to guide nutrient management actions across the state. To address this, the Water Board has funded the synthesis of science and a suite of studies to create the foundation for this guidance. In estuaries, this scientific synthesis has recently been summarized by Sutula et al. (in prep).

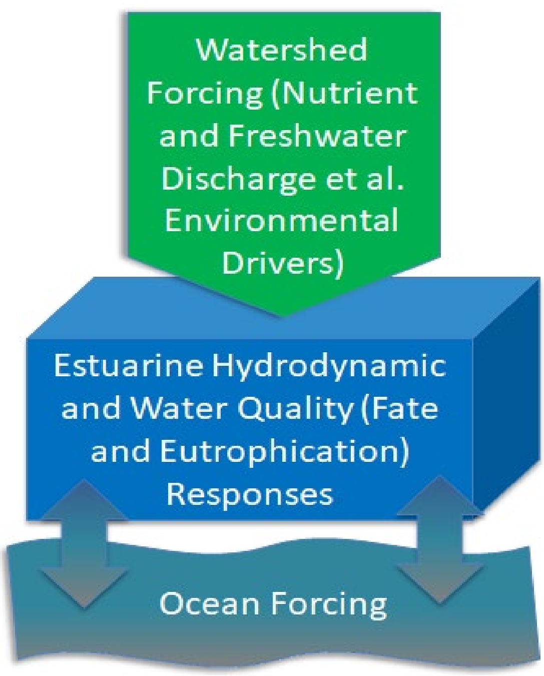
1.4 Brief Description of Scientific Approach and Document Organization

The Central Coast Water Board initiated work on the TMDL by reviewing the basis for nutrient targets for the estuary, develop initial estimates of nutrient sources to the Slough using available information and existing simplified modeling tools, and recommend an approach for further development of a TMDL and the establishment of load and waste load allocations. This project builds on the foundational work of Tetra Tech (2018) to set up, calibrate and apply a suite of watershed loading and receiving water models and synthesize literature to 1) investigate a range of biostimulatory targets that are protective of Slough beneficial uses under present day conditions, and 2) calculate the total maximum daily load of allowable nutrients needed to meet those conditions (Table 1.1). The project addressed four management questions, for which the methods and findings have been organized into two report chapters (Table 1.1.). A brief description of the approach is given here, supplemented by a more detailed description in each chapter. This synthesis of science will be considered by the Central Coast Water Board for their decisions on water quality management and/or policy decisions for the Slough.

Table 1.1. Summary of study questions and the report chapter in which their methods and findings can be found. SWAT= Soil and Water Assessment Tool; WASP = Water Quality Simulation Program (WASP), BBSRM = Biostimulatory-Biointegrity Stress Response Models.

Chapter	Question	Tools employed
2	• What are the ranges of biostimulatory targets that protect Slough beneficial uses? • How does existing monitoring data compare to these targets?	Literature review
3	• What is the attribution of nutrients et al. environmental drivers of eutrophication from different watershed, within Slough, and oceanic sources? • How do these nutrients drive eutrophication symptoms? • What are the maximum allowable loads of nitrogen and phosphorus that will meet these Slough biostimulatory targets?	SWAT EFDC-WASP Literature review

In Chapter 2, we identified from published literature the key eutrophication symptoms (indicators) and environmental drivers in Elkhorn Slough and selected candidate eutrophication indicators that represent key beneficial use protection endpoints. The synthesis of biostimulatory targets relevant for Elkhorn Slough is based on a recent comprehensive review of eutrophication thresholds for California's Mediterranean estuaries (Sutula et al. in prep), from which Water Board staff chose a set of provisional targets under consideration. These provisional targets were then applied to available monitoring data to illustrate how they would be used to assess beneficial use support.



Chapter 3 is based on an integrated modeling approach to support Slough eutrophication management discussions needs (Figure 1.5), consisting of 1) a means to estimate watershed nutrient loads and other environmental drivers, and 2) an estuarine hydrodynamic, biogeochemical, and lower trophic model that can capture inputs from ocean and the contributing watershed to simulate circulation, transport of nutrients and other environmental drivers, and eutrophication outcomes. A third validated coupled physical-biogeochemical model of the coastal ocean (Monterey Bay) was used to provide ocean forcing to the estuarine water quality model (Deutsch et al. 2021; Kessouri et al. 2021).

Figure 1.5. Conceptual approach to integrated modeling of the Slough, including watershed and ocean forcing of environmental drivers (biostimulatory substances and conditions) and estuarine hydrodynamic and eutrophication responses.

Tetra Tech (2018) reviewed available information on land use, soils and nutrient concentrations and watershed hydrology and estimated watershed nutrient loading using the STEPL – the Spreadsheet Tool for the Estimation of Pollutant Load (Mercado et al. 2014; Table 1). They used these estimates to compare with those from an uncalibrated version of Soil and Water Assessment Tool (SWAT; Nietsch et al. 2011) developed within the EPA Hydrologic and Water Quality System (HAWQS). HAWQS (Texas A&M 2017). They found that both STEPL and uncalibrated SWAT estimates of nutrient loading are within the same order of magnitude, and that a SWAT model, with appropriate calibration, can likely be a successful approach for refined estimates of loading from the watershed.

Therefore, we set up and calibrated a SWAT model for Elkhorn Slough contributing watersheds, building on Tetra Tech (2018, 2021), to estimate nutrient concentrations, temperature, and freshwater discharge entering the Elkhorn Slough from its contributing watershed (Figure 1.6). The model domain included direct drainage to the Slough including Porter Marsh, Gabilan/Tembladero Sloughs, and drainage to the Old Salinas River up to the head of tide.

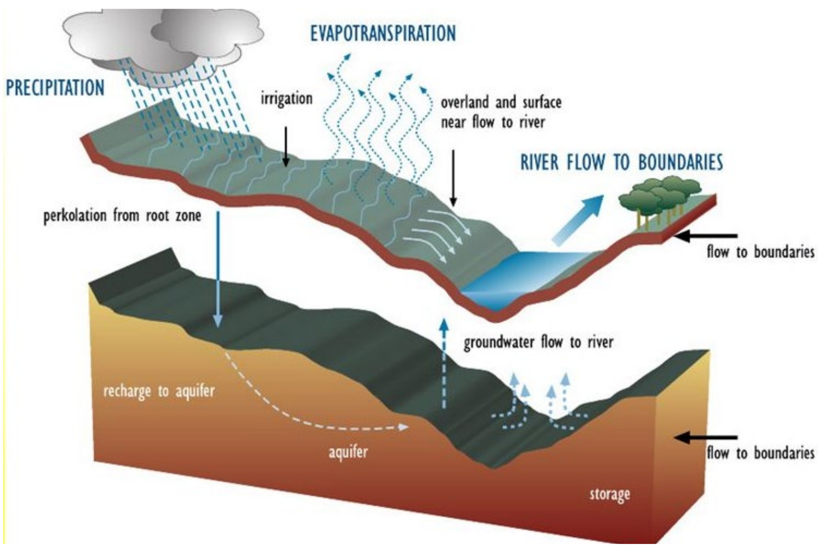


Figure 1.6. Representation of hydrologic processes captured in SWAT.

To predict eutrophication outcomes in Elkhorn Slough, we developed and calibrated a dynamic simulation model of estuarine physical circulation and eutrophication processes (Tetra Tech 2022). The estuarine hydrodynamic submodel, based the environmental fluid dynamic codes (EFDC), captures oceanic exchanges and complex tidal circulation within the Slough. Water quality is simulated using the Water Quality Analysis Simulation Program Version 8.4.0 (WASP) (https://www.epa.gov/ceam/wasp-model-documentation). WASP predicts a suite of eutrophication outcomes (e.g., algal biomass, dissolved oxygen, pH, total and dissolved inorganic nitrogen, and phosphorus) in the Elkhorn Slough as a function of watershed and nearshore oceanic physical and biogeochemical forcing (tidal forcing and circulation and freshwater discharge with inherent heat, nutrient concentrations, other environmental drivers) as well as internal cycling within the Slough (Figure 1.7).

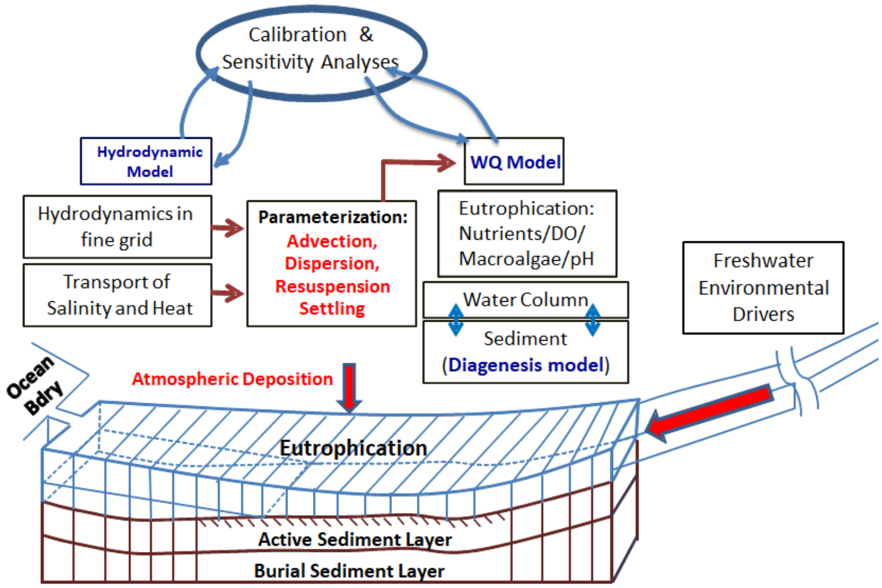


Figure 1.7. Generic schematic of coupling between estuarine hydrodynamic and eutrophication (water quality) model.

Ocean boundary conditions were provided by the 1-km resolution UCLA-SCCWRP Regional Ocean Modeling System (ROMS) with biogeochemical elemental cycling (ROMS-BEC) for the Monterey Bay nearshore (Deustch et al. in review). The model provides a full suite of physical, biogeochemical (N, P, and C), and lower ecosystem state variables, averaged on daily time steps. Outputs are available from 1997-2017. However, the version of the simulation utilized did not include land-based inputs to Monterey Bay (Sutula, personal communication).

The models were calibrated (Sakar et al. 2022; Butcher et al. 2022), then applied to investigate drivers of eutrophication in Elkhorn Slough (Chapter 3) and to derive the nitrogen (N) and phosphorus (P) allowable loading to the Slough.

2. Scientific Basis for Biostimulatory Targets

2.1 Elkhorn Slough Beneficial Uses

Understanding the beneficial uses provides guidance for developing numeric targets that must protect these uses. The specific beneficial uses for water bodies include municipal and domestic supply (MUN), agricultural supply (AGR), commercial and sport fishing (COMM), freshwater replenishment (FRESH), industrial process supply (PRO), groundwater recharge (GWR), preservation of rare and endangered species (RARE), water contact recreation (REC1), noncontact water recreation (REC2), wildlife habitat (WILD), cold freshwater habitat (COLD), warm freshwater habitat (WARM), fish migration (MIGR), and fish spawning (SPWN). In addition, coastal water body beneficial uses include industrial service supply (IND); navigation (NAV); marine habitat (MAR); shellfish harvesting (SHELL); commercial and sport fishing (COMM); wildlife habitat (WILD); and fish migration (MIGR). Elkhorn Slough uses are shown in Table 2.1.

Table 2.1. Designated beneficial uses of Elkhorn Slough versus that of nearby waterbodies.

Waterbody Name	MUN	AGR	PRO	IND	GWR	REC-1	REC-2	WILD	COLD	WARM	MIGR	SPWN	BIOL	RARE	EST	FRESH	NAV	POW	COMM	AQUA	SAL	SHELL
McClusky Slough					X	X	X	X		X		X		X					X			X
Elkhorn Slough						X	X	X	X	X	X	X	X	X	X		X		X	X		X
Los Carneros Creek	X					X	X	X	X		X	X		X		X			X			
Bennett Slough/Estuary						X	X	X	X	X		X	X	X	X				X			X
Parsons Slough						X	X	X	X			X	X	X	X				X			X

2.2 Conceptual Model of Eutrophication in Elkhorn Slough

Elkhorn Slough is a low inflow, Mediterranean estuary and, as such, shares characteristics with similar systems around the world. Conceptual models of the development and expression of eutrophication in Mediterranean estuaries provide a starting point to identify key symptoms of eutrophication that represent the environmental problem to be addressed in the Elkhorn Slough TMDL (Sutula et al. in prep).

In "minimally disturbed" condition (Figure 2.1, left panel), the subtidal and unvegetated habitat of Mediterranean estuaries typically have good light penetration and as such as are dominated by primary producers tolerant of low nutrient conditions, such as seagrass, and benthic diatoms living on the sediment surface that can contribute up to 50% of the primary production in an estuary (Underwood and Kromkamp 1999). Benthic macroalgae and phytoplankton are in low abundance, providing food and refuge for invertebrates, juvenile fish, crabs, and other species. In larger, well-flushed California estuaries, especially in northern California, intertidal to shallow subtidal portions are often dominated by the seagrass (e.g., *Zostera marina)*. A diversity and abundance of epibenthic and infaunal invertebrate communities bioturbate sediments at depths of 10 cm or more, resulting in a net autotrophic benthic environment with low sediment oxygen demand that efficiently metabolizes carbon and denitrifies inorganic nitrogen. Water column oxygen and pH are high with moderate diel variability that varies as a function of photosynthesis/respiration and diurnal and spring/neap cycles of the tide. Robust salt marsh exists in the upper intertidal habitat.

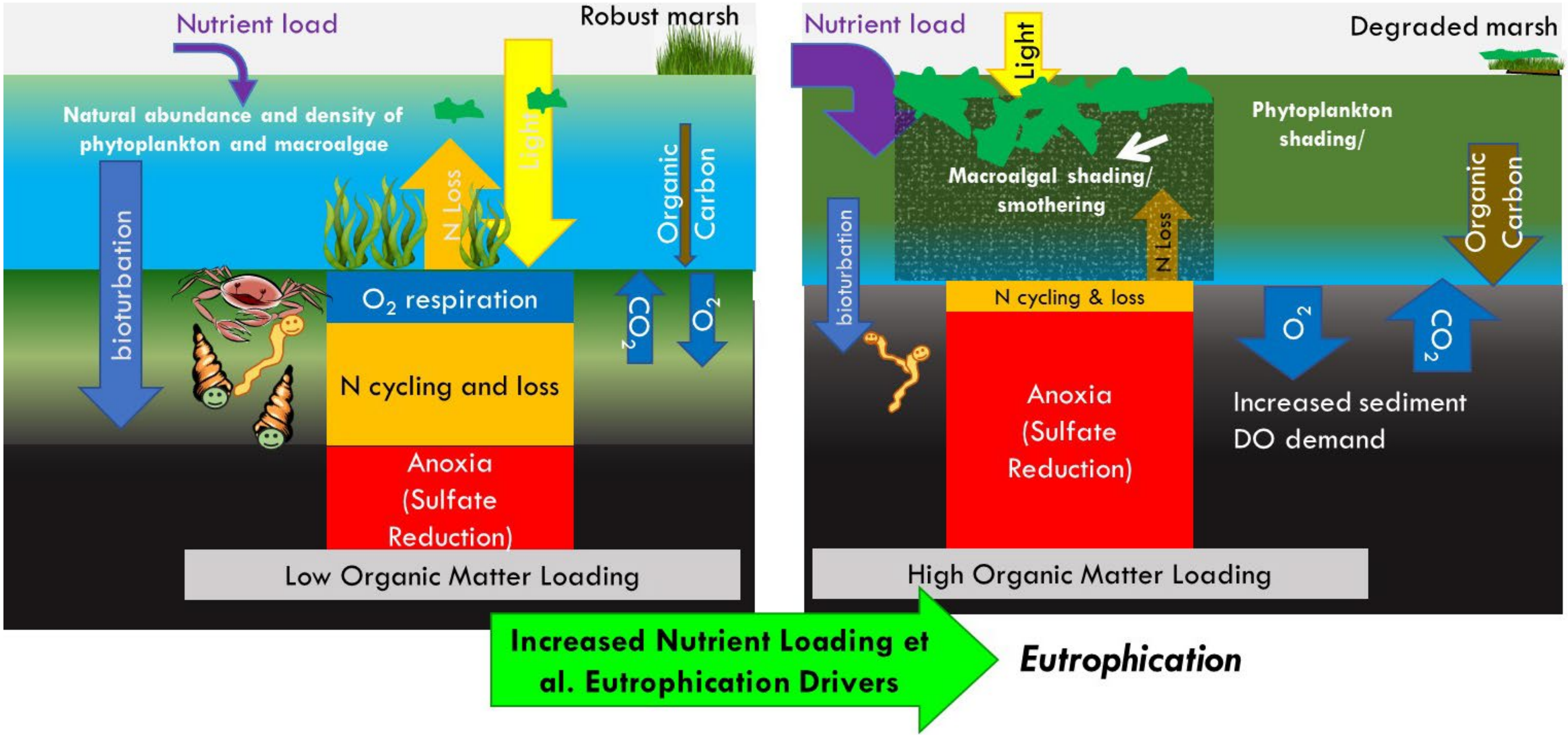


Figure 2.1. Conceptual model of the pathways of impacts of eutrophication on Mediterranean estuaries such as Elkhorn Slough, from Sutula (in prep). Left panel represents the characteristics of a minimally disturbed estuary. The right panel illustrates the characteristics or symptoms of eutrophication across habitats.

As nutrient availability and organic matter enrichment increases (Figure 2.1, right panel), the growth of certain species of epiphytic micro-, macroalgae and phytoplankton is stimulated by high nitrogen (N) and phosphorus (P) loading (Raffaelli et al. 1989; Valiela et al. 1992; Peckol and Rivers 1995; Pihl et al. 1999; Krause-Jensen et al. 2007), with N often the limiting nutrient in marine ecosystems. Macroalgae are somewhat unique for their ability to rapidly take up large pulses (luxury uptake) of inorganic nitrogen and store it for future growth (Fujita 1985; Bjornsater and Wheeler 1990; Fong et al. 1994; Lotze and Schramm 2000; Naldi and Viaroli 2002). This is especially important in Mediterranean-climate estuaries, 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 agricultural waters in both the wet and dry seasons (Zedler 1996). *Ulva spp* are particularly opportunistic due to their ability to shift the habitat niche they occupy (Kennison 2008). Early stages of the life cycle are tied to intertidal and shallow subtidal benthic habitat, where sufficient light penetrates. However, once the macroalgal thallus reaches a critical size, it detaches from the benthos and forms floating mats (Astill and Lavery 2001; Cummins et al. 2004; Kopecky and Dunton 2006). These mats are no longer restricted to intertidal or shallow subtidal regions; rather, they accumulate into floating rafts and can grow in virtually any portion of the estuary where the current transports them (Thomsen et al. 2006). As the biomass of both benthic algae and phytoplankton biomass increases, the shift favors nutrient tolerant and often, harmful algal bloom species that can produce toxins harmful to marine life and humans (Fong et al. 1993; Valiela et al. 1997; Viaroli et al. 2008; Sutula et al. 2017).

As these algal blooms increase in magnitude, frequency, and duration, they limit light penetration and trigger a host of adverse consequences including:

- Shading or smothering of benthic diatoms, seagrass, and salt marsh habitat;
- Shift towards benthic or planktonic harmful algal species that produce toxins that can cause acute or chronic illness, including death in humans and wildlife;
- Organic matter accumulation in the sediments and water column causes a fundamental alteration of biogeochemical cycling, changes in sediment and water column oxygen demand and CO2 flux, and weakening of the structural integrity of salt marsh sediments, making them susceptible to erosion;
- Decreased recruitment and survival of benthic invertebrates in mudflats and subtidal habitat and reduced carrying capacities for fishes and shorebirds;
- Increased frequency of water column and sediment hypoxia and heightening heterotrophic bacterial activity, resulting in poor water quality and increased frequency of diseases;
- Drift macroalgal blooms can become thick wrack that can smother salt marsh, mudflats, and oyster reefs;
- Poor aesthetics and an increase in odors relating to the decomposition of organic matter and increased sulfide production; and
- Shifts in both trophic and community structure of invertebrates, birds, and fishes.

In Elkhorn Slough, decades of scientific research and a dedicated effort of volunteer monitoring led by NERR staff have documented a multitude of these eutrophication symptoms (Hughes et al. 2011, 2012; Jeppesen et al. 2016). The NERRS has sponsored the use of a water quality report card (https://water.elkhornslough.org), through which various symptoms of eutrophication can be viewed over a 20-year time series. Surface water and groundwater nutrient concentrations in and around the Slough are elevated (Hughes et al. 2012). The concentrations of dissolved inorganic nitrogen in Slough waters place the estuary among the most nitrogen-enriched in the world (Hughes et al. 2011).

Elkhorn Slough has a long history of extensive and chronic macroalgal blooms, including well-known bloom species of both green *(Ulva spp.)* and red macroalgae species (*Gracilaria* spp; Allen 1992; Wasson et al. 2017). These blooms cover the mudflats, the subtidal habitat (as floating and benthic or attached blooms), and also wash up onto the marsh plain as wrack (Wasson et al. 2017; Figure 2.2). In an 80-year time series derived from aerial photographs, Wasson et al. (2017) documented that this macroalgal wrack increased exponentially over time and was highly correlated with nitrate concentrations in the Slough. They documented that this macroalgal wrack was linked to a decrease in salt marsh cover, flowering, and canopy height and also contributes to retreat of vegetation from the bank edge, increased bank erosion, and conversion of marsh to mudflat (Wasson et al. 2017). The combination of drift macroalgae and phytoplankton blooms limit light to seagrass beds (Sutula 2011) and have been linked to declines in eelgrass abundance in Elkhorn Slough (Hughes et al. 2012).

Elevated biomass of macroalgae and phytoplankton can lead to extreme fluctuations in levels of dissolved oxygen and pH. Dissolved oxygen levels peak during the day while algae photosynthesize, and hypoxia (low oxygen) becomes more extreme overnight while algae respire in the absence of light (Hughes et al. 2011). Strong swings in dissolved oxygen concentrations (hyperventilation) and pH are apparent in the upper areas of the Slough, particularly in areas with restrictions on tidal flow and high concentrations of nutrients (Chapin et al. 2004; Beck and Bruland 2000). Hypoxia has been shown to contribute to stress and lower survival in fish and oysters in the Elkhorn Slough (Jeppesen et al. 2016). Hughes et al. (2018) showed that even at DO concentrations below 4 mg/L, juvenile groundfish are extirpated from the Slough, thus impacting the important nursery habitat role that estuaries play.

Estuarine acidification (declining pH) accompanies oxygen stress on organisms because respiration is coupled to the production of dissolved inorganic carbon including carbon dioxide, which is an acid (Borges and Gypens 2010). At aragonite or calcite saturation of ~1.5, the biologically relevant form of pH, shells begin to dissolve. At values below 1, organisms are typically not able to make or repair shells. For a model saline estuary, the release of CO2 associated with the development of hypoxia is sufficient to reduce pH levels by more than 0.5 units and to decrease aragonite solubility to levels where dissolution of estuarine calcifiers (e.g., clams, oysters, crabs, snails, etc.) would be favored (Howarth et al. 2011). For this reason, nutrient-enriched estuaries such as Elkhorn are considered to be among the ecosystem's most vulnerable to ecological and biogeochemical perturbations from ocean acidification (Howarth et al. 2011).

Figure 2.2. Examples of macroalgal impacts to different habitats in Elkhorn Slough, showing floating algae (top left) and benthic (attached) algae (top right), both of which block light to seagrass and other beneficial benthic algae. Macroalgae can grow or be pushed onto mudflats (bottom right) and salt marsh habitats (bottom left), which degrade salt marsh plants, destroy the structural integrity of salt marsh soils, accelerating erosion, and reduce the diversity of epifauna and benthic infauna that are important forage for fish, birds, and marine mammals.

Chronic algal blooms accelerate organic matter accumulation in mudflat, and subtidal sediments. The accumulation of this large amount of labile organic matter stimulates microbial communities and increases the sediment oxygen demand and shallows redox potential (Cardoso et al. 2004). Zones of sediment anoxia and sulfate reduction, which are typically found at depths of 10 cm or more, become shallow, often extending throughout the sediment right under the algal mat (Dauer et al. 1981; Hentschel 1996). This leads to surficial pore water ammonia and sulfide concentrations that are toxic to epifauna and benthic invertebrates (Gianmarco et al. 1997; Kristiansen et al. 2002), which are important forage for fish, resident and migratory birds, and marine mammals (Green et al. 2013). Under extreme organic matter loading or macroalgal blooms, the zone of sulfate reduction rises to the surface, often extending throughout the sediment under the algal mat (Dauer et al. 1981; Hentschel 1996). While most benthic fauna has evolved to deal with organic rich conditions (Pearson and Rosenberg 1978; Hargrave et al.

2008), the amount of sediment nitrogen as well as the degree of hypoxia or anoxia in overlying water can be an important factor structuring the community composition (e.g., Rosenberg et al. 1991; Diaz and Rosenberg 1995; Baustin and Rabalais 2009). The depth to the apparent redox potential discontinuity (aRPD), below which zone of anoxia and sulfate reduction are encountered (Cardoso et al. 2004), is a visual indicator of this problem; a boot print in estuarine mudflats can reveal a thin grey layer that overtops the dark anoxic and sulfidic smell right below. In a survey of Elkhorn Slough subtidal and mudflats, Hughes et al. (2011) found that 10 of 18 sites had aRPD values < 2 cm (Hughes et al. 2012), a threshold below which translates to reduced habitat volume and quality for benthic infauna and an alteration in their community structure (Sutula et al. 2014).

Toxin-producing harmful algal blooms (HABs) can bioaccumulate in the food web and adversely affect humans and animals. Though toxins are not routinely monitored by the NERRS program, Elkhorn Slough has documented impacts by both freshwater (cyanobacteria) and marine toxic HABs. Cyanotoxins were ubiquitous and persistent in several Monterey Bay National Marine Sanctuary coastal watersheds over a 3-year time-series survey (Gibble and Kudela 2014). The deaths of more than 30 federally endangered sea otters in Elkhorn Slough and Monterey Bay were linked to ingestion of a freshwater cyanotoxin, microcystin, that bioaccumulated in bivalves (Miller et al. 2010; Kudela 2011). Blooms of the marine diatom Pseudo-nitzschia can produce domoic acid (DA), a toxin that most commonly causes neurological disease in humans and marine mammals. Moriarty et al. (2021) recently documented cardiovascular effects in southern sea otters in Elkhorn Slough due to acute and chronic exposures. They found that that clams and crabs, the favored prey of sea otters, were particularly high-risk prey because they are effective bioconcentrators of DA that also slowly excrete the toxin (Schultz et al. 2008, 2013). Sea otter postmortem examinations also indicated consumption of clams and crabs were associated with higher odds of fatal DA toxicosis (Miller et al. 2010). It certainly is an important question whether cyanobacterial or marine HAB blooms are originating in Elkhorn Slough; however, Sutula et al. (2016) noted that HAB blooms seeded from outside the estuary can proliferate within the estuary if conditions permit (e.g., abundant nutrient supply, light regime, etc.).

Cumulatively, these adverse effects result in a reduction in recreational use (REC1 and REC2) of Elkhorn Slough, poor water column and benthic habitat quality for estuarine and marine aquatic species (WARM, COLD, EST and MAR), direct impacts to populations of threatened and endangered (RARE), migratory (MIGR) birds and spawning (SPWN) fish and marine mammals, and reduction in the economic value of commercial and sports fisheries (COMM), aquaculture (AQUA), and shellfish (SHELL) harvesting.

2.3 Recommended Eutrophication Indicators for Elkhorn Slough

Sutula (in prep) conducted an extensive synthesis of the literature to evaluate candidate indicators and synthesis of thresholds to select biostimulatory targets for California's Mediterranean estuaries. This information is provided in summary here. For details, refer to Sutula (in prep).

Five primary and three supporting indicators are recommended for monitoring and as potential biostimulatory targets for Elkhorn Slough, based on the Sutula (in prep) evaluation of eutrophication response measures (Table 2). Five primary indicators met all five evaluation criteria and are recommended for Elkhorn Slough (Table 2.1): 1) has a clear link to ecosystem services and designated uses, 2) has 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 numerical simulation models, 3) has a scientifically sound and practical measurement process, 4) must be able to show a trend either towards increasing or/and decreasing eutrophication with an acceptable signal: noise ratio, and 5) literature exists to provide a basis for choosing a threshold. Supporting indicators failed to meet one or more of these criteria. Primary indicators included: measures of *planktonic and benthic microalgal and macroalgal abundance (as chlorophyll-a or bulk dry weight)*, *continuous DO and pH*, and *sediment nitrogen*. We note that of these primary indicators, most are currently routine employed in the NERRS water quality monitoring and report card, except for macroalgal biomass and sediment nitrogen.

Four indicators were recommended as "supporting:" 1) macroalgal percent cover, 2) HAB cell density and toxins, 3) benthic invertebrate taxonomy and/or aRPD, and 4) water column concentrations of phosphate, ammonium, and total dissolved inorganic nitrogen (including nitrate, nitrite, and ammonium).

Macroalgal cover was designated as supporting, because low biomass blooms can see up to 100% coverage on mudflats (McLaughlin et al. 2014). However, the risk of high biomass blooms increases with increasing cover, so percent cover becomes an effective screening tool and one that is already employed through various approaches (remote sensing and field-based) by the Elkhorn Slough NERR.

HAB cell density and toxin concentration were designated as supporting by Sutula (in prep) because the status of science in modeling relationships with environmental drivers is in its infancy. However, because of the extreme threat that toxic HABs pose to humans and wildlife, incorporation of this indicator is strongly recommended.

Both *aRPD* and *benthic invertebrate taxonomy and interpretive indices* (e.g., M-AMBI, Gillett et al. 2015) that rely are on these data respond to a large suite of stressor gradients and are not specific to eutrophication. However, the Water Boards Surface Water Ambient Monitoring Program (SWAMP) has routine sampling protocols for sediment chemistry and benthic invertebrate taxonomy as well as interpretive guidance to support it sediment quality objectives (SQO). These protocols can be used to identify toxic contaminants that are impairing benthic habitat quality in Elkhorn Slough (e.g., pesticides). With recent work by Gillett et al. (in prep), eutrophication can be integrated into a robust causal assessment of benthic habitat quality. Because of the expense of the SQO protocols (~$6000/sample), aRPD can be a useful screening variable to assess benthic habitat quality and is one that has already been used by the NERRS monitoring program.

Table 2.2. Summary of indicators for which a synthesis of thresholds and associated assessments framework are presented for Elkhorn Slough.

<table>
<tr><th rowspan="2">Habitat</th><th rowspan="2">Indicators</th><th colspan="4">Habitat</th><th rowspan="2">Suggested Sampling Protocols for California Mediterranean Estuaries</th></tr>
<tr><th>Wetland</th><th>Mudflat</th><th>Seagrass</th><th>Subtidal</th></tr>
<tr><td colspan="7">Primary Indicators</td></tr>
<tr><td>Macroalgae</td><td>Biomass</td><td>X</td><td>X</td><td>X</td><td>X</td><td>McLaughlin and Sutula 2021</td></tr>
<tr><td>Sediment Quality</td><td>Sediment TN Nitrogen (N)</td><td></td><td></td><td colspan="2">X</td><td>McLaughlin et al. (2012)</td></tr>
<tr><td>Phyto-plankton</td><td>Chlorophyll-a (link to water clarity/ seagrass, DO, and HAB toxins)</td><td></td><td></td><td colspan="2">X</td><td rowspan="3">McLaughlin et al. (2012)</td></tr>
<tr><td rowspan="2">Physio-chemistry</td><td>Continuous dissolved oxygen</td><td></td><td></td><td colspan="2">X</td></tr>
<tr><td>Continuous pH</td><td></td><td></td><td colspan="2">X</td></tr>
<tr><td colspan="7">Supporting Indicators</td></tr>
<tr><td>Macroalgae</td><td>Percent cover</td><td>X</td><td>X</td><td colspan="2">X</td><td>McLaughlin et al. (2021)</td></tr>
<tr><td>HAB</td><td>cell density and toxin</td><td></td><td>X</td><td colspan="2">X</td><td>CalHABMAP.org</td></tr>
<tr><td rowspan="2">Sediment quality</td><td>Taxonomic composition</td><td></td><td></td><td colspan="2">X</td><td>Bight Coastal Ecology Program 2013</td></tr>
<tr><td>aRPD</td><td></td><td>X</td><td colspan="2">X</td><td>Sutula et al. 2014</td></tr>
<tr><td rowspan="2">Dissolved inorganic nutrients</td><td>Dissolved inorganic nitrogen</td><td></td><td></td><td colspan="2" rowspan="2">X</td><td rowspan="2">Bight 2008 Eutrophication Assessment Protocol</td></tr>
<tr><td>Dissolved inorganic phosphorus</td><td></td><td></td></tr>
</table>

Dissolved inorganic nutrients (phosphate and total dissolved inorganic nitrogen - DIN) are recommended as supporting lines because alone they do not represent robust diagnostic variable for eutrophication (Bricker et al. 2003; Sutula (in prep) and are prone to false negatives or positives. However, they are an excellent indicator of eutrophication risk (Cloern et al. 2001) and, more specifically, indicative of anthropogenic sources of nitrogen and phosphorus. Ammonium is called out separately because of its toxicity to aquatic organisms. The USEPA and CalEPA have water quality objectives for ammonium, which were set at 0.1 mg/L (USEPA 1999).

2.4 Scientific Basis for Biostimulatory Targets

Sutula (in prep) conducted a review of existing literature to synthesize data from experiments and field studies that describe the stressor-response relationships and associated ecological thresholds[2]. Data derived from California experiments and field studies was preferred over that generated outside this region and thus tracked separately in the review. This information was

[2] Generally, ecological thresholds have been defined as “the point at which there is an abrupt change in an ecosystem property or where small changes in an environmental driver produce large responses in the ecosystem” (Grossman et al. 2006). Cuffney et al. (2010) further distinguish between resistance thresholds (e.g., a sharp decline in ecosystem condition following an initial no effect zone) and exhaustion thresholds (a sharp transition to zero slope at the end of a stressor gradient at which point the response variable reaches a natural limit). In contrast to resistance or exhaustion thresholds, a “reference envelope" is the stressor range in sites found in a minimally disturbed condition (Stoddard et al. 2006).

then synthesized into a set of simplified diagnostic assessment framework that classifies ecological condition of a given estuarine segment into five categories from very high to very low ecological condition, consistent with the approach of the E.U. Water Directive Framework (e.g., Best et al. 2007; Scanlan et al. 2007). This translation from continuous response along a stressor gradient to categorical assessment framework loses information but is intended to simply the translation to routine management application (e.g., policy decisions on biostimulatory targets). The review attempted to be explicit as to how the magnitude, extent, and duration of the effects should be assessed, what appropriate methods are available to measure the indicator and the temporal frequency and spatial density of data required to make that assessment and provide specific provide guidance on how the data should be analyzed to categorize estuarine segments. The development of these assessment frameworks assumed that the estuary could be divided into segments, corresponding to natural gradients in freshwater flow, tidal mixing, and hydraulic residence time, three major controls on eutrophication response.

Macroalgal Blooms

Sutula (in prep) proposed a macroalgal assessment framework (Table 2.3). The California SWAMP Macroalgal Assessment Protocol (McLaughlin et al. 2021) provides a standardized field monitoring, laboratory processing, and data management protocol, a quality assurance project plan, and training materials for this assessment. Briefly, biomass is typically measured in subtidal habitat using a randomized grid approach and a benthic sampler or for intertidal habitat using a 30- to 50-m transects at three sites established in the lower intertidal and upper subtidal zone (< 1 m at MLLW) parallel to the water line, or where no intertidal habitat exists, in a grid fashion in subtidal habitat. Typically, 15 biomass samples are harvested from a known surface area at random points along the 3 parallel transect or grids. Once algae are collected, algal biomass samples are cleaned of intercalated mud, associated fauna, and entangled debris in the laboratory (Boyle et al. 2004; Kennison 2008). For purposes of standardization and consistency across the state, the assessment thresholds are expressed in dry weight, though cleaned wet weight and other means of measuring biomass (e.g., biovolume) are possible, with a conversion between cleaned wet weight to dry weight of roughly 10:1. Macroalgal blooms are highly variable both seasonally and interannually. For many estuaries, the critical period of peak macroalgal abundance will be in the growing season (April – October), but blooms can occur year-round, depending on the timing of peak nutrient loads, tidal inlet closure, etc. (Kennison 2008). For this reason, monthly or bimonthly assessments throughout the year are highly recommended.

Table 2.3. Proposed assessment framework to diagnose eutrophication using macroalgae in seagrass dominated and unvegetated intertidal flat and subtidal habitat for California estuaries. Assessment is based on average biomass (g dw m-2) of the two highest consecutive sampling periods if sampled monthly; if sampled bi-monthly, the assessment is based on the maximum segment-averaged biomass from any single sampling event. In habitats in which seagrass beds are distributed into the intertidal zone, the seagrass density is sparce or intermixed with unvegetated habitat, the framework for unvegetated intertidal and shallow subtidal habitat should be employed. Percent cover is a supporting line of evidence, recommended to be used as a screening variable.

Ecological	Macroalgal Biomass (g dw m^{-2})	Percent Cover

Condition Category	Unvegetated	Seagrass	Percent chance of exceeding 70 g dw m^{-2}	Percent Cover Screening Targets
Very Low	≥140	≥170	> 80%	≥80
Low	70 to < 140	100 to < 170	45 to 80%	50 to < 70
Moderate	30 to < 70	75 to < 100	12 to 45%	20 to < 50
High	15 to < 30	15 to ≤ 75	5 to 12%	5 to < 20
Very High	≤ 15	≤ 15	<5%	≤ 5

Biomass and percent cover can be used to categorize an estuarine segment into five categories from very high to very low ecological condition in seagrass-dominated and unvegetated intertidal and subtidal habitats, with thresholds for the bins are tied explicitly to field and experimental evidence (see Table 6, Sutula (in prep). Basis for thresholds are as follows: 1) high ecological condition: reference envelope (< 15 g dw m^{-2}, Sutula et al. 2014, 2017), 2) moderate ecological condition: no observed effect level (30 g dw m^{-2}, benthic invertebrates, Cardoso et al. 2004; 75 g m^{-2}, seagrass, Huntington and Boyer 2008a,b), 3) low ecological condition: resistance threshold, benthic invertebrates, 70 dw m^{-2} (Bona 2006), 84-94 g dw m^{-2} lowest observed effect level, benthic invertebrates, Green et al. (2013), 100 g dw m^{-2} lowest observed effect level, seagrass, Bittick et al. (2018)., and 4) very low ecological condition: Azoic sediments correlated with high porewater sulfide 140 g dw m^{-2}, Green (2011), severe impacts on shoot density, 170 g dw m^{-2}, Huntington and Boyer (2008b). Percent cover is proposed as a screening level indicator. Sutula (in prep) derived the probability of exceeding the 70 g dw m^{-2} biomass thresholds from synoptically collected percent cover data.

Bloom duration is explicitly accounted in frequency of sampling. Both Green et al. (2013) and Bittick et al. (2018) found that severe effects occurred within 4 -10 weeks of macroalgal biomass treatment. For this reason, sampling frequency is strongly recommended to be monthly, but under circumstances of restricted resources, bi-monthly at minimum, to categorize temporal variability and capture peak bloom periods. Macroalgal framework should be applied to estimate the average of biomass across the three sites (15 samples per site) in the intertidal/subtidal shoreline method or the subtidal grid (15 samples). The annual categorization should be based on the frequency of sampling. If monthly sampling events are conducted, the assessment framework should be applied to the segment average biomass of the two consecutive sampling periods with the highest biomass. If bi-monthly monitoring events are made, then the framework should be applied to the maximum biomass for any single event for one year, representing the segment average for that period.

Dissolved Inorganic Nutrients

Sutula (in prep) deferred from developing assessment frameworks for ambient estuarine dissolved inorganic nutrient concentrations because these values are inherently site-specific. However, because of their utility as a screening level indicator, we provide additional analyses that may contribute towards their use in Elkhorn Slough. We provide these analyses to support decisions on screening level analyses for nitrate+nitrite (N+N) and phosphate (PO4).

First, median and 75th percentiles of six minimally disturbed reference estuaries have been summarized by Sutula et al. 2016, Figure 2.3). The 75th percentile of total nitrogen (TN; including N+N, ammonium, and organic nitrogen) and total phosphorus (TP; including phosphate, particulate, and organic phosphorus) is 0.43 mg/L TN and 0.09 mg/L TP. Of note, total dissolved inorganic nitrogen in these systems was 0.03 mg/L, while ortho-phosphate was 0.07 mg/L

Estuary	TN		TP		NO3+NO2		NH4		PO4	
	Mean (Range)	75thile	Mean (Range)	75thile	Mean (Range)	75thile	Mean (Range)	75thile	Mean (Range)	75thile
Navarro River	0.18 (0.01-0.34)	0.29	0.04 (0.01-0.32)	0.02	0.005 (0.003-0.01)	0.006	0.008 (0.005-0.01)	0.01	0.005 (0.003-0.01)	0.006
Salmon Creek	0.34 (0.20-0.62)	0.45	0.20 (0.03-0.30)	0.11	0.005 (0.002-0.28)	0.043	0.009 (0.004-0.024)	0.009	0.13 (0.0002-0.28)	0.23
Laguna Creek	0.28 (0.12-0.51)	0.42	0.13 (0.07-0.22)	0.19	0.04 (0.007-0.09)	0.06	0.02 (0.008-0.04)	0.02	0.08 (0.01-0.17)	0.11
Waddell Creek	0.27 (0.12-0.50)	0.33	0.08 (0.04-0.19)	0.08	0.009 (0.002-0.03)	0.009	0.013 (0.006-0.04)	0.01	0.03 (0.01-0.04)	0.03
Topanga Creek	0.41 (0.24-0.62)	0.50	0.04 (0.02-0.05)	0.05	0.003 (0.007-0.08)	0.004	0.006 (0.002-0.01)	0.006	0.009 (0.003-0.04)	0.009
San Onofre Creek	1.16 (0.37-2.4)	1.4	0.28 (0.04-0.91)	0.28	0.007 (0.0007-0.002)	0.009	0.01 (0.004-0.03)	0.01	0.09 (0.01-0.44)	0.22
Median ± SD All Estuaries	0.31 ± 0.33	0.43 ± 0.38	0.09 ± 0.10	0.09 ± 0.10	0.006± 0.01	0.021± 0.004	0.01± 0.004	0.01± 0.004	0.01± 0.005	0.07 ± 0.09

Figure 2.3. Median and 75th percentile of total nitrogen and total phosphorus in six California intermittently closed reference estuaries. From Sutula et al. (2016).

Second, Wasson et al. (2017) found historic records of ambient Slough nitrate associated with historic imagery of macroalgal wrack on adjacent mudflats and salt marsh. Their work shows an inflection point of accelerated risk of wrack accumulation at 1 mg/L nitrate, while 0.3-0.5 mg/L nitrate was a baseline of low number of grids cells with macroalgal wrack percent cover (Figure 2.4).

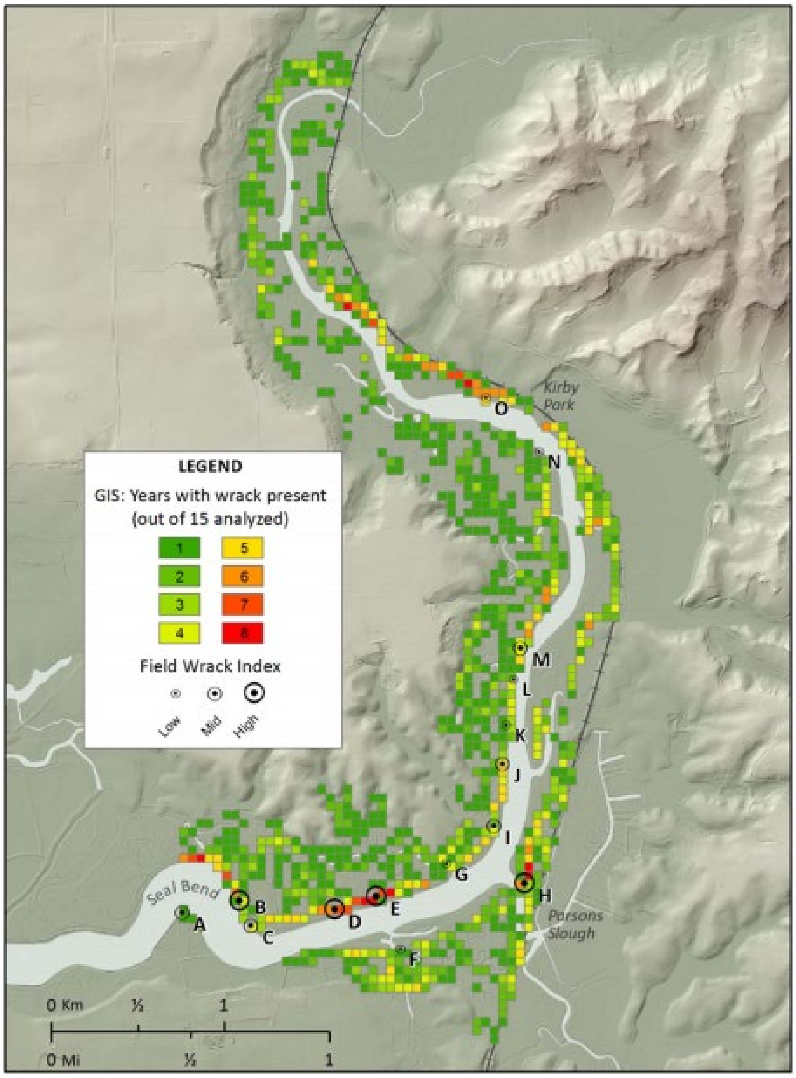
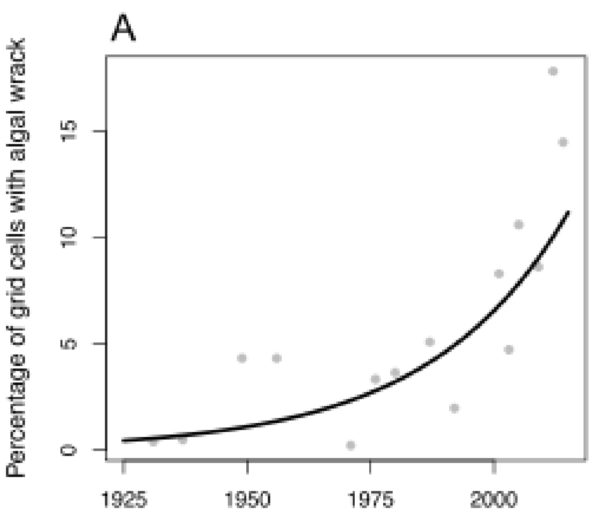
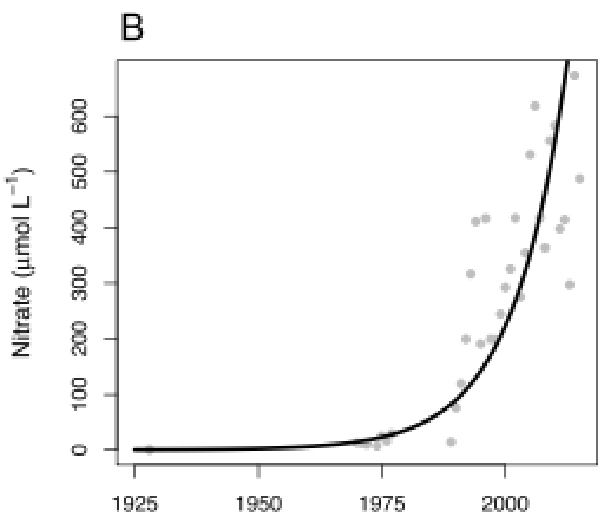


Figure 2.4. From Wasson et al. (2017) showing temporal changes in algal wrack, nitrate concentrations over the last century. (A) Results of analyses of photographs from 15 year s between 1931 and 2014, showing the percentage of grid cells with algal wrack present. (B) Trend in nitrate concentrations sent out from 1928, 1971-77, 1985 – 2015.





Finally, nitrate values can be constrained by information that we have on concentrations associated with nitrate in upwelled waters from Monterey Bay. Using 1 km horizontal resolution output from the OPC-sponsored Regional Ocean Modeling System (ROMS) with biogeochemical elements cycling (BEC; Deustch et al. 2021), we found mean to maximum values of 0.4 to 0.6 at 50 m depth, and values of 0.2 to 0.4 mg/L nitrate at 25 m depth (Figure 2.5).

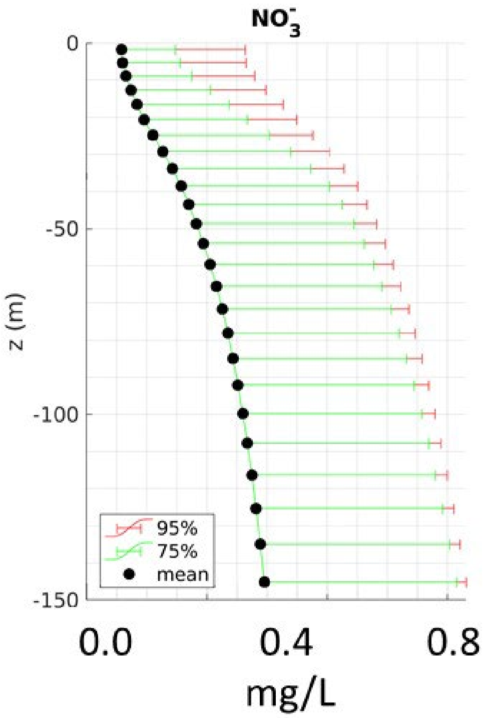
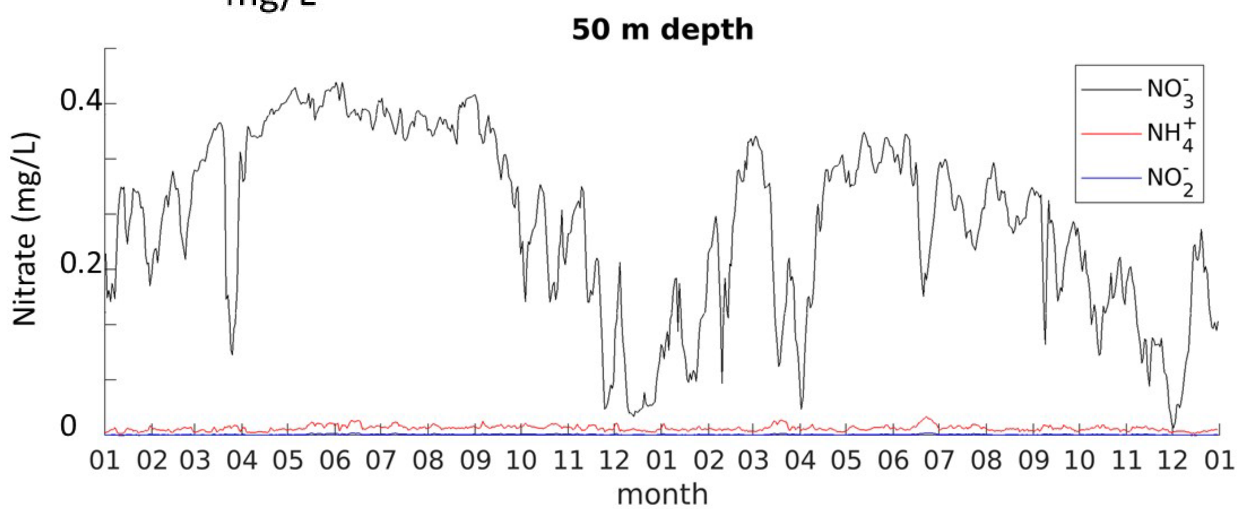


Figure 2.5. Synthesized information on nitrate concentrations in Monterey Bay at the mouth of Elkhorn Slough from the ROMS-BEC 1 km output (Deutsch et al. 2021). Left panel shows concentrations of nitrate with depth in Monterey Bay, with black circles indicating mean and the red and green lines indicating the upper 95th and 75th percentiles. Bottom panel shows a seasonal time series for this same location showing maximum concentrations of nitrate that occur during upwelling events (and characteristic seasons, e.g., March – September).



Phytoplankton Chlorophyll-a

Chlorophyll-a has formed a cornerstone of standardized approaches to assess eutrophication (Bricker et al. 2003; Zaldivar et al. 2008) and to support regulatory water-quality goals in estuaries (Harding et al. 2014) because it is a well-recognized indicator that integrates nutrient loadings and represents adverse effects to ecosystems. Decisions based on quantitative endpoints can be based on deviations from "reference" conditions when data prior to degradation are available (Andersen et al. 2011), or on quantitative relationships with ecosystem impairments such as low DO, increased risk of toxic HABs, or decreased water clarity (e.g., Harding et al. 2014; Sutula et al. 2017). Sutula (in prep) found that phytoplankton-based assessments for an estuary should ultimately be developed from site-specific monitoring and modeling analyses. Pragmatically, however, data and resources may not be available for these analyses and thus existing frameworks could be considered, including (Table 2.4): 1) ASSETs (Bricker et al. 2003) European Union Water Framework Directive (EU WFD) for Mediterranean lagoons (Zaldivar et al. 2008), 3) California reference studies of bar-built estuaries (Sutula et al. 2018), and 4) site-specific assessment framework for San Francisco Bay (Sutula et al. 2017). The once important source of differences in thresholds between these frameworks is due to the metric used to assess them. For example, phytoplankton assessment frameworks that utilize annual average or growing season median (Sutula et al. in prep; Zaldivar et al. 2008) have understandably lower values that those that utilize the 90th percentile of the spring blooms (ASSETS, Bricker et al. 2003) or single event samples related to HAB events (Sutula et al. 2017). Second, bar-built estuaries appear to have much lower values in general than those of deep, well flushed estuaries. For these reasons, ample caution is urged in applying these values for data-poor Mediterranean estuaries.

Table 2.4. Existing frameworks for evaluating eutrophication class based on water column chlorophyll-a concentration.

Assessment Framework	Reference	Metric	Classification				
			Very Low	Low	Moderate	High	Very High
EU WFD	Zaldivar et al. 2008	Annual Average	> 30	> 10-30	> 7 - 10	> 5 - 7	< 5
ASSETS	Bricker et al. 2003	Bloom Maximum	> 60	> 20-60	> 5-20	< 5	
CA Bar-built estuary reference study	Sutula et al. (in prep)	Apr- Oct median North Coast South Coast	>16 > 24		> 6- 16 > 9-24	< 6 < 9	
San Francisco Bay	Sutula et al. 2017	Linked to HAB toxins, single event	> 25		> 13- 25	< 13	
		Linked to DO WQC. Feb-Sept. mean for South SFB	> 58	> 44-58	> 32-44	> 14-32	< 14

In most Mediterranean estuaries, records of chlorophyll-a prior to human disturbance are not available, complicating development of reference chlorophyll-a ranges. Sutula et al. (in prep) assessed chlorophyll-a in minimally disturbed (and intermittently closed) bar-built estuaries in California over a one-year period. Chl-a concentrations as captured by in situ fluorescence show these bar-built estuaries to be generally characterized by episodic blooms that occur periodically throughout the season for a week or two, rather than one extended productive period. Latitude and ocean forcing matter, particularly on upwelling-dominated coastlines. The mean of discrete chl-a samples found in the colder and foggy northern and central California coast was significantly lower than that of southern California bar-built estuaries (p-value – 0.002). Bar-built estuaries open to tidal exchange can be influenced by seasonal periods of high productivity associated with upwelling and El Nino Southern Oscillation events. The 90th percentile of chlorophyll-a in these reference estuaries (northern/central coast = 15.7 mg m^{-3}; south coast = 23.9 mg m^{-3}), typically used by CalEPA to designate "impaired" waterbodies using a reference-based approach, while the 70th percentile would be indicative of sustaining aquatic life uses (northern/central coast = 5.7 mg m^{-3}; south coast = 8.5 mg m^{-3}). Because of the short duration (1-yr) and low sample size of this dataset, however, use of these values should be appropriately caveated and when possible, used as a supporting line of evidence with other site-specific approaches (see below).

Table 2.5. Examples of light requirements and associated chlorophyll-a thresholds for seagrass habitats in various estuaries. These values are typically derived based on modeling and site-specific analyses and are for illustrative purposes only.

Location	Chla (ug L^{-1})	Light Requirement	Source
Yaquina Bay, OR	< 3-5 [1]	0.8– 1.5 m^{-1}, expressed as water clarity	Brown et al. 2007
Tampa Bay, FL	< 3.8-9.8	0.65-1.04 m^{-1}, expressed as water clarity	Janicki et al. 2000
Sarasota Bay, FL	< 6.1-11.0	Not given	Janicki et al. 2009
Maryland Coastal Bays	< 15 in lower bays < 60 in upper tributaries	15% attenuation of surface irradiance	Wazniak and Hall 2005

[1] Light attenuation is primary requirement, chlorophyll-a and turbidity are secondary

Derivation of chlorophyll-a thresholds through estuary-specific relationships with either low DO, increased risk of toxic HABs, or decreased water clarity that impact seagrass and other benthic primary producers is well established and one of the mostly commonly used in large well-studied estuaries across the globe (e.g., Chesapeake Bay, Tampa Bay, and San Francisco Bay, Sutula 2011). In either case, it requires an extensive monitoring dataset that captures the full gradient in these relationships (e.g., Sutula et al. 2017) or numerical water quality models, validated with monitoring data. For that reason, management of data limited bar-built estuaries often face this roadblock. Numerical models have the advantage or statistical analyses of observations in that they are able to pull out confounding and interactive relationships of chlorophyll-a with ocean forcing, tidal dynamics and freshwater flow. As an example of this type of analysis, Sutula et al. (2017) analyzed a 20-year dataset that included chlorophyll-a (1993-2014), phytoplankton species composition (1993-2014), DO (1993-2014), and algal toxins (2012-2014) to quantify chlorophyll-a thresholds and related uncertainty that correspond to categories of "protected" and "at risk" in the context of current DO water quality criteria and HAB cell density and toxin alert levels (Sutula et al. 2017). Use of the Sutula et al. (2017) developed for San Francisco Bay or ASSETS generic thresholds across all U.S. estuaries should be considered with caution, because these relationships are not likely to hold with precision outside of the waterbody for which they were developed.

Sediment Nitrogen

Benthic macroinvertebrates (BMI) 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. BMI communities provide: 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, and 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. California EPA have recognized the intrinsic value of BMI and as such, have policies that includes estuarine BMI as a primary indicator of aquatic life in streams and estuaries. The thresholds assembled here are based on the response of the macrobenthic community to the production and accumulation of excessive amounts of organic matter in soft sediment habitats of California's Mediterranean estuaries (Figure 2.6). Assessments of sediment and macrobenthos are based on 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). The thresholds discussed here are therefore tied to this protocol (Table 2.6). Sediment organic matter accumulation is a time-integrated measure, so suggested sample frequency is 1-2 times per year, preferably at the onset of the growing season (April- May) and just after (September-October). Sutula et al. (2018) noted a draw down in %TN associated with development and persistence of a chronic macroalgal bloom, then a fall increase as the bloom senesced. Because sediment grain size and organic matter enrichment can be spatially patchy, a grid sampling approach is recommended to characterize the variability in sediment in the subtidal habitat (McLaughlin et al. 2021), but the spatial representation can be improved if multiple samples or a composite over a larger surface area can be made.

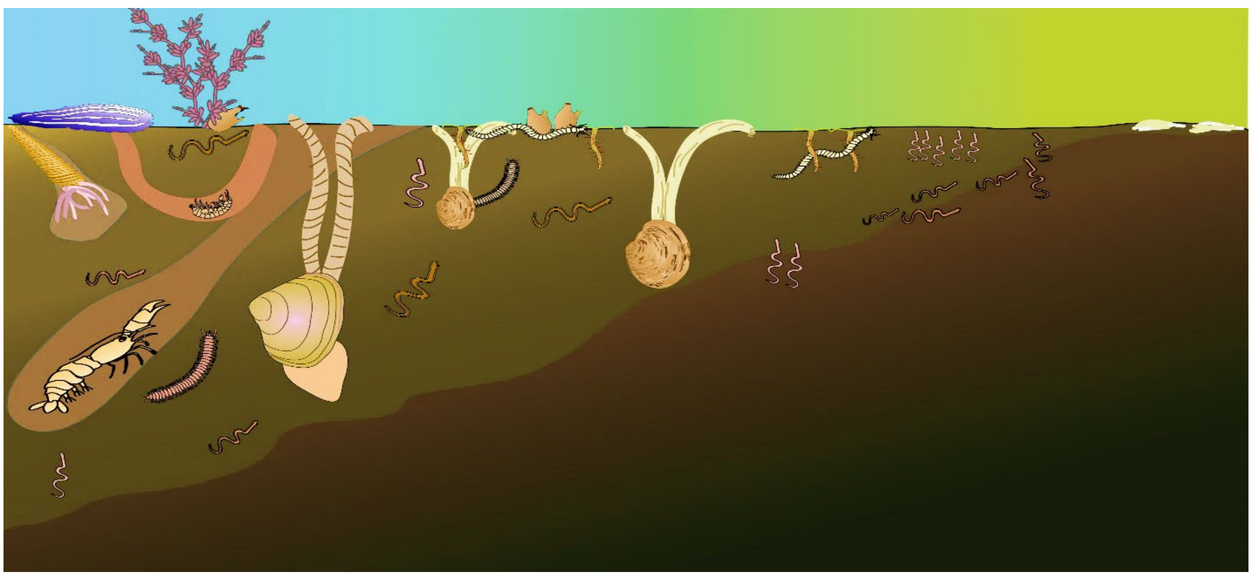


Figure 2.6. From Pearson and Rosenberg (1978), depiction of loss of functional groups of benthic invertebrates with increasing organic matter accumulation (from left to right) and associated zones of aRPD. Left side of figure shows a deep aRDP (> 10 cm) associated with deep burrowing filter feeders. With increasing sediment organic matter accumulation, depth to which infauna can live shallows and the diversity of those infauna decline, until the sediments eventually come azoic (devoid of infauna).

Table 2.6. Assessment framework based on derived thresholds of sediment %OC and %N developed from logistic regressions with M-AMBI (Gillett et al. in prep) and comparison to thresholds developed from a separate relationship based on depth to apparent redox potential discontinuity (aRPD; Sutula et al. 2014).

Condition Category	Thresholds Associated with M-AMBI		Sediment TN (%) Thresholds Associated with aRPD
	Probability of Minimally Disturbed Benthos	Sediment TN (%)	
Very Low	< 0.60	≥ 0.37	> 0.29
Low	0.60 to < 0.70	>0.25 to <0.37	
Moderate	0.70 to < 0.80	0.12 to <0.25	0.14 to ≤ 0.29
High/Very High	≥0.80	< 0.12	< 0.14

Gillett et al. (in prep) utilized the relationships between gradients in sediment organic carbon (%OC) and percent nitrogen (%N) to response of the M-AMBI index (Gillett et al. 2015; Pelletier et al. 2018), a national index of ecological based on benthic invertebrate community composition that has good performance across salinities gradients (a key attribute for application to bar-built estuaries). Logistic regression was used to identify tipping points of increasing concentrations of %OC and %N on M-AMBI. Sutula (in prep) used the Gillett et al. (in prep) analyses to identify thresholds, based on the probability from 50% (very low condition) to 80% (very high condition) of meeting a M-AMBI score of 1 (defined as within the natural variability of benthic infauna communities. Sediment percent N varies as a function of grain size. Thus, analyses were initially classified by sediment type (muddy versus sandy) to reduce covariance. Thresholds for sandy sediments did not pass AIC criteria for model significance and were dropped, so derived thresholds are applicable to muddy sediments with % fines > 40%. Thresholds that link to aRPD are qualitatively different than M-AMBI derived thresholds. M-AMBI is a whole community assessment method and does not value certain functional groups over others; aRPD however changes significantly from deep to shallow with the loss of deep burrowing filter feeders (e.g., clams, shrimp, etc.), which are key for oxidizing sediment organic matter and maintenance of biogeochemical functioning (e.g., denitrification).

Surface Water DO and pH

Estuarine organisms have adapted to "life on the edge" by virtue of their tolerance to frequent and highly variable exposure to low pH, carbonate saturation state, temperature (T), and dissolved oxygen concentrations (DO). Physiological optima in these intrinsic properties define a natural envelope of suitable habitat. When physiological tolerance thresholds in these properties are exceeded, beyond which the tolerance turns into vulnerability, pH, DO, and T can become stressors. Spatial and temporal patterns in these stressors limit suitable habitat for organisms and ultimately control the distribution, interaction among species, and ecosystem productivity (García-Reyes et al. 2013). Acceleration of climate change is causing low pH (acidification) and low dissolved oxygen (deoxygenation), which can be exacerbated in estuarine and nearshore coastal waters due to eutrophication, which enhances biological respiration (Duarte et al. 2013). Low oxygen and pH co-vary, so organisms experience suboxic or hypoxia conditions are also experiencing acidification stress. Temperature further exacerbates the problem because it increases biological activity, decreases oxygen solubility, increases aerobic oxygen demand, and exacerbates stress on organisms, so warming can greatly limit available potential habitat (Deutsch et al. 2020).

The US 1970 Clean Water Act established the basic tenets for establishing DO and pH water quality criteria and derivation of site-specific criteria. All coastal Water Board basins have DO and pH numeric objectives established decades ago or more recent updates that are typically based on the review and use of the Virginia Province approach (e.g., Supplemental Table 2, USEPA 2000). These criteria, developed to protect estuarine and marine life in the 20th century, are in many cases ill-suited to address the challenges of climate change and its effects on water quality (Tomasetti and Gobler 2020). Six major problems exist with the scientific approach underpinning these criteria:

1. pH criteria were developed to be largely "end of pipe" and are not protective of marine and estuarine calcifiers (Weisberg et al. 2016; Bednarsek et al. 2019), which can occur at pH ranges > 7.5.
2. Criteria that are based on a deviation from a naturally occurring range (either DO or pH) are very difficult to evaluate, particularly given lack of monitoring data and shifting baselines from climate change.
3. DO criteria only considers dissolved oxygen demand based on experiments done at fixed temperatures, while in nature, organismal demand fluctuates through wide ranges of temperature extremes (Deutsch et al. 2015).
4. DO criteria are typically derived based on a limited set of USEPA database of acute and chronic oxygen tolerance experiments. Data available are U.S. east coast centric and rarely have adequate information the chronic tolerances of native species to suboxic conditions. Furthermore, these experiments used nitrogen gas to establish low-DO conditions, an approach that artificially elevates pH values, creating unrealistic, basified conditions. Realistic conditions would have lower pH associated with low oxygen.
5. DO and pH criteria give information on thresholds, but not appropriate averaging to apply them, and this nuance in application creates a tremendous inconsistency in their application.
6. Percent saturation incorporates temperature effects on DO solubility but does not account for effects of variable respiratory demand under different temperature regimes. A classic example is the work of Davis (1975), which showed the response of freshwater salmonid populations to variable DO, ranging from 7.75 (no adverse effects) to 4.25 mg/L (widespread effects; Supplemental Table 2). Percent saturations of 85% are protective of salmonid populations at 20ºC, but not at 25ºC, a value which is reached in many Mediterranean bar-built estuaries with limited summertime exchange (McLaughlin et al. 2013).

Fundamentally changing the scientific basis for DO and pH criteria is a tall order, given that the endeavor requires both water manager staff time and USEPA approval for TMDLs and site-specific criteria. For Elkhorn Slough, it is beyond the scope of the present project. It is not readily clear to managers what they will gain in terms of environmental protection if the resources are spent in changing the DO and pH criteria. Acknowledging that issue, this section of the review focuses on the scientific basis for pH and DO criteria in California estuaries, focused on effects on aquatic life. This information can be used as an additional line of evidence to guide interpretation of existing WQO.

Bailey et al. (2014) derived DO criteria for California estuaries and is illustrative of the pitfalls in the current scientific approach. In California, most estuarine DO criteria found in Water Board basins utilize the conceptual equivalent of chronic continuous criterion (CCC) as their basis, either for salmonid use designations (COLD) or non-salmonid uses (WARM). Utilizing this standardized methodology, Bailey et al. (2014) found the CCC as 6.2 mg/L when salmonids are present and 5.2 mg/L when absent. However, because most of the species in the EPA database are from US east coast estuaries, derivation is rarely based on native species. In the California example, the 4 fish and 7 invertebrate species used to derive the CCC contained no native California species and, therefore, the criterion calculation is based on a combination of introduced and surrogate species. This data limitation generally results in the same CCC, and CMC calculated across all US estuaries, because of the database used to derive CCCs. Furthermore, as noted above, concentration-based criteria ignore the influence of temperature and salinity.

European Water Directive Framework proposed assessment of DO in transitional waters (Best et al. 2007) attempts to correct for salinity (where high-quality waters scale from 7 mg/L in freshwater to 5.8 mg/L in full strength marine; Table 2.7). Data from Hughes et al. (2015) showed that a predicted probability of flatfish occurrence as a function of the DO increased above 0.8 at > 5.8 mg/L, while below 4.0 mg/L flatfish were essentially extirpated.

Table 2.7. Best et al. (2007). Proposed EU Water Framework Directive classification system for dissolved oxygen in subtidal habitats for Estuaries and Coastal Waters.

Ecological Condition Category	Freshwater (mg/l)	Marine (mg/l)	Protection level
Very Low	< 2	< 1.6	No salmonids present
Low	> 2 to < 3	> 1.6 – < 2.4	Presence of non-salmonids, poor survival of salmonids
Moderate	> 3 to < 5	> 2.4 – < 4.0	Most life stages of non-salmonid adults
High	> 5 to < 7	> 4 – > 5.7	Presence of salmonids, but not all life stages and flat fish
Very High	≥ 7	≥ 5.8	All life stages of salmonids and flat fish

Existing CalEPA pH WQO were intended to evaluate end of pipe effects and as such as not protective of aquatic life; and no EPA guidance on derivation of biologically relevant criteria (Weisberg et al. 2016). There have been a number of studies that have described processes by which marine and estuaries calcifiers are vulnerable to acidification (Kroeker et al. 2013; Calosi et al. 2019; Bednaršek et al. 2019; Bednaršek et al. 2021). Recent studies have targeted syntheses of these experimental and studies with the express intent to develop thresholds of ecological effects of low pH or saturation state on pteropods (calcifying zooplankton, Bednaršek et al. 2019), echinoderms (principally, sea urchins, Bednarsek et al. 2021), bivalves (Barton et al. 2012; Gimenez et al. 2018; Waldebusser et al. 2015), and decapods (principally, crabs, Bednarsek et al. 2021). Synthesis of the thresholds across these four taxa carries an important caveat. First, the carbonate system parameter (e.g., pH, pCO2, or calcium or aragonite saturation state) that is the most relevant varies by taxa and specific pathway of impairment. Application of the thresholds must utilize the parameter reported that represents the major axis of biological response, rather than converting all into a single parameter for convenience (Waldebusser et al. 2015).

Thus, interpretation and application of this information requires an understanding of the native species composition of the estuary, their life history, and the import of the impact pathway and life stage. The general guidance provided here is to discern the applicable species and thresholds of interest and to apply those specific thresholds, rather than a composite assessment framework. However, water quality managers must choose which species, which can be confusing, and therefore interpretation of the gradient in responses across different taxa is illustrative (acknowledging the caveats above, and for comparative purposes only). This cross-taxa synthesis illustrates that pteropods and bivalve larvae are among the most sensitive endpoints documented to date (Table 2.8; Bednarsek et al. 2019). In general, as would be expected, larval life stages (pelagic) are more sensitive than adults, particularly for species that benthic adult stages. Duration is an important factor; assuming equal magnitude, short-duration thresholds will be more sensitive to acidification effects than one of longer duration. At pH ~< 7.90 (duration of 2-5 days), mild dissolution of adult pteropod and larval Mytilus spp. shells has been observed in both field and laboratory studies (Bednarsek et al. 2019; Waldebusser et al. 2015). Mild dissolution is considered an early warning response (Bednaršek et al. 2019). At ~> 7.80 to < 7.90 and at durations of 5-7 days, physiological effects across taxa are more common; moderate dissolution occurs and is considered an important effect because it imposes energetic costs for repair and maintenance (Bednarsek et al. 2019). Below 7.80, severe dissolution occurs (and can be lethal in adult pteropods and larval life stages). Larval mortality can be found at higher durations. Below 7.70, mortality of larvae is more common at shorter duration (7-14 days) and physiological effects are pervasive and severe. This information can generally be summarized into an assessment framework of pH values (Table 13), with important caveats, mentioned below.

Treatment of duration merits specific discussion. Bednarsek et al. (2019) provides specific guidance for application of these thresholds that includes the instruction that a threshold is triggered and assessed only for those continuous periods longer than the stated duration. However, in high variability coastal environments such as estuaries, such an approach is extremely problematic. Instead, a running average of the data is recommended, with a count of the number of days below the given threshold. Even then, application of these thresholds to bar-built estuaries that have a strong diel variability will be problematic, because those high and low values will be lost in the averaged. If this is the case, the guidance should be to consider utilizing a percent of time below the threshold as an additional metric of assessment.

Other caveats are important. As with DO, each of these thresholds are essentially tied to a specific experimental temperature that typically mimics ambient conditions. Higher temperatures can exacerbate the effects of acidification but many of the experiments from which these temperatures have been derived cannot account for that effect. Second, most experiments are kept at a stable treatment state with low variation. The dynamic exposures that are typical of nearshore and estuarine habitats, which present an opportunity for recovery and local acclimatization, are not represented in these thresholds.

Table 2.8. Effects of pH on marine and estuarine calcifying organisms in subtidal habitat. For specific organisms for which saturation state is a better predictor, thresholds should be translated from monitored pH to aragonite or calcite saturation state. Duration varies, but for the purpose of assessment is given at 5 days, assessed as a rolling average.

Condition Category (subtidal)	pH*	Endpoint
Very Low	< 7.60	Adult mortality of some sensitive decapod species
Low	> 7.60 to < 7.70	Larval mortality of some sensitive species, adult pteropods
Moderate	> 7.70 to < 7.80	Severe shell dissolution, growth effects, physiological impacts
High	> 7.80 to < 7.90	Mild – severe dissolution, physiological impacts
Very High	≥ 7.90	Life stages supported

2.5 Demonstration of Target Application to Elkhorn Slough

An overarching purpose of this chapter is to provide the evidence to support decisions on biostimulatory targets, which are policy decisions. However, stakeholders are better able to engage the Water Board on the substantive bases for these targets if they understand how they are applied. In this section, we provide an analysis of existing Elkhorn Slough NERRS monitoring data, based on provisional targets under consideration by the Central Coast Water Board staff (Table 2.9). Final decisions on targets are pending refinement based on stakeholder feedback and adoption by the Board.

Two types of data sources were used for the demonstration (Figure 2.7): 1) four NERRS system long term monitoring sites located in or along the main channel (continuous DO and pH and nutrient grab samples), and 2) 24 volunteer monitoring sites (nutrients, chlorophyll-a grab samples, DO field measurements and % cover of floating macroalgae. From each, data from January 1, 2010, through October 23, 2021, were utilized, because this represents a time period in which the field and analytical protocols are likely to be the most consistent.

Continuous DO and pH data are logged at 15-minute intervals, so a one-hour running average was applied to remove sensor spikes, then calculations of the 7-day average daily DO minima and daily DO mean were calculated (the latter for comparisons sake). The 10th percentile of each year was calculated and compared against the target (7 mg/L DO and 7.8 pH), then the number of years in which the target was not met was calculated.

Macroalgal biomass is not routinely monitored by Elkhorn NERRS staff or volunteers, though % cover of floating macroalgae is assessed at their monthly volunteer sites. For this reason, floating % macroalgae was assessed at the annual maximum value. Chlorophyll-a and nutrient concentration targets were assessed at the 90th percentile for each year.

The results are presented for continuous data (DO and pH) and nutrient grab samples for the four main NERRS water quality sites, while only a summary table is presented for the 24 volunteer monitoring sites. Results and graphical depiction of data as box and whiskers plots for each of these sites can be viewed in Appendix 2.

DO and pH

At Azevedo Ponds, North and South Marsh, the Slough sites where continuous data are available, the target of 7 mg/L was never met over the twelve-year period, regardless of the temporal statistic (daily minima or mean). At Vierra Mouth, the target was never met if a DO minima statistic was applied but was met 6 of 12 years if a daily average was applied. For sites in which discrete DO samples represent an instantaneous snapshot, only the Salinas River site consistently met the threshold when applied against the 10th percentile of the daily DO minima (Table 2.10).

For pH, application of a 5-day rolling average against a target pH of 7.8 made for more infrequent thresholds excursions, in comparison with the DO 7-day average of daily minima (Table 2.10). Application of this target identifies locations where the mean values were chronically depressed below the threshold, such as was the case in Azevedo Ponds and South Marsh. North Marsh and Vierra Mouth had fewer excursions of the pH target.

Table 2.9. Provisional Elkhorn Slough biostimulatory numeric targets under consideration by the Central Coast Water Board staff based on their understanding of Water Board precedent (e.g., basin plan WQO), Sutula (in prep), and Sections 2.2-2.4 (and cited references therein).

Indicator	Metric	Type	Threshold	Temporal statistic	Justification
DO	Concentration	Primary	7 mg/L	10th percentile of 7 day running average of daily DO minima	Central Coast Water Board Basin Plan, State Water Board Impaired Waters policy
pH	Unitless	Primary	7.8	10th percentile of 7 day running average of daily pH minima	
Macroalgae	Biomass	Primary	30 g dw m^{-2}	Average of two consecutive highest biomass periods (if monthly) or annual maximum (if bimonthly)	No effects benchmark outside range of reference, Sutula et al. 2014, with temporal averaging consistent with Sutula (in prep).
	% Cover	Screening	20 %	Average of two consecutive highest biomass periods (if monthly) or annual maximum (if floating macroalgae only)	Threshold associated with low risk (< 12%) of exceeding lowest observed adverse effect of 70 g dw $^{-2}$). Temporal statistic based on McLaughlin et al. (2014). Recommend
Phytoplankton	Chlorophyll-a	Primary	15 µg/L	90th percentile of annual monthly grab samples (non-vegetated sites only)	Applicable to non-vegetated subtidal habitats.
Sediment Quality	Sediment %TN	Primary	0.1%	Composite of multiple sites within a segment, once per year at the end of the dry season (post algal blooms senescence).	95th percentile of aRPD exhaustion threshold, signaling loss of deep burrowing filter feeders.
	Sediment aRPD	Screening	> 2 cm	Single sample (or if multiple cores) Geomean of cores or benthic camera images in a segment	Loss of deep burrowing filter feeders.
Dissolved Inorganic Nutrients	DIN (N+N and NH4)	Screening	0.5 mg/L	90th percentile of annual monthly grab samples	Twice the 95th percentile of reference in Central Coast streams, within the mean to 95th percentile of upwelling nitrate in Monterey Bay (F. Kessouri, personal communication)
	Ammonium	Primary	0.1 mg/L		EPA Toxicity thresholds for total ammonia nitrogen (USEPA 1999)
	Phosphate	Screening	0.09 mg/L		95th percentile of reference in Central Coast streams.

Table 2.10. Summary of frequency of years (out of 12) in which the 24 NERRS volunteer monitoring sites did not meet the provisional biostimulatory target (Table 2.9) for discrete DO, floating macroalgal % cover, surface water chlorophyll-a, DIN, and total phosphorus. Table is color coded for ease of reading where green shade = 0-1 sites, white shade is 2-3 sites and red shade is 4 or more sites over 12 years. "Restricted" refers to the tidal hydrology as a result of transportation infrastructure, dikes, or weirs.

Region	Restricted?	Site	DO	FloatAlg%Cov	Chl-a	DIN	Nh4	TP
Azevedo Ponds	Y	APC	12/12	10/12	11/12	4/12	10/12	5/12
Azevedo Ponds	Y	APN	10/12	5/12	0/12	5/12	11/12	11/12
Azevedo Ponds	Y	APS	12/12	4/12	12/12	7/12	8/12	9/12
Bennett Slough		JR	12/12	1/12	0/12	12/12	12/12	12/12
Bennett Slough	Y	SP	8/8	5/8	5/8	7/8	4/8	8/8
Freshwater Pond	Y	CAT	11/11	4/11	11/11	11/11	10/11	11/11
Freshwater Pond	Y	ROK	8/9	4/9	8/9	9/9	9/9	9/9
Lower Slough	Y	BSE	11/12	7/12	12/12	3/12	7/12	9/12
Lower Slough		NERRVM	8/12	0/12	0/12	6/12	6/12	12/12
Middle Slough		KP	11/12	0/12	5/12	6/12	9/12	11/12
Middle Slough	Y	NERRNM	11/12	3/12	7/12	5/12	9/12	12/12
Middle Slough	Y	RBR	11/12	2/12	1/12	7/12	10/12	12/12
Middle Slough	Y	RSM	11/12	0/12	5/12	3/12	7/12	12/12
Middle Slough		SKL	7/12	0/12	1/12	12/12	11/12	12/12
Middle Slough	Y	STB	12/12	12/12	12/12	9/12	12/12	6/12
Moro Cojo Slough	Y	MCS	7/12	10/12	9/12	12/12	12/12	12/12
Moro Cojo Slough	Y	MCS2	7/7	0/11	7/12	7/12	7/12	6/12
Moro Cojo Slough		MLN	5/12	1/12	5/12	12/12	12/12	12/12
Moro Cojo Slough		MLS	11/12	9/12	9/12	12/12	12/12	12/12
Salinas River	Y	SRB	0/12	5/12	8/12	12/12	9/12	12/12
Tembladero Slough	Y	MDW	10/12	1/12	8/12	12/12	11/12	12/12
Tembladero Slough		PRN	8/12	0/12	9/12	12/12	12/12	12/12
Tembladero Slough	Y	PRS	9/12	1/12	9/12	12/12	12/12	12/12
Tembladero Slough	Y	TS	4/12	2/12	9/12	12/12	12/12	12/12
Tembladero Slough	Y	TS2	6/10	0/10	9/10	10/10	8/10	10/10
Upper Slough	Y	CC	12/12	6/12	12/12	12/12	12/12	12/12
Upper Slough		HLE	11/12	5/12	11/12	12/12	12/12	12/12
Upper Slough		HLW	12/12	3/12	11/12	10/12	11/12	12/12

Macroalgal Floating % Cover

The macroalgal floating % cover target was only met consistently in about half of the volunteer monitoring sites and most consistently in Tembladero Slough. Although macroalgal floating % cover is easily assessed visually by volunteers, its inherent flaw is that it is possible to have 100% of attached (benthic) macroalgal bloom that is not visible from the surface. Moreover, prevailing winds can drive the cover to one side of the assessment site, which will cause a false negative. Given these caveats, at minimum a paired assessment of macroalgal biomass, benthic and floating algal cover is recommended to ground truth the floating algal volunteer protocol and determine whether refinements are needed.

Chlorophyll-a

Discrete chlorophyll-a grabs at the volunteer monitoring sites show only a handful of sites met the provisional target consistently. Sites that were noted to meet the floating macroalgal % cover target in Tembladero and Middle Slough generally tend to consistently not meet the chlorophyll-a target, with the exception of sites at Vierra Mouth (NERRVM), the Reserve Bridge (RBR), Skippers Landing (SKL), and JR in Bennett Slough.

Dissolved Inorganic Nutrients

Figure 2.10 gives graphical visualization of the NERRS water quality sites and the corresponding frequency of not meeting the provisional target. In general, screening values for DIN were infrequently met at all sites. In contrast, the three downstream sites (South and North Marsh, Vierra Mouth) meeting TP screening values (monitored by NERRS as phosphate) for most years. None of the four sites met the ammonium target. At the volunteer monitoring sites, most sites did not meet any of the DIN, NH4 or TP targets under consideration, though temporal statistic does greatly influence the frequency (see Appendix 2 for site-by-site examples).

Sediment Nitrogen and HAB Toxins

Sediment nitrogen is not assessed by the NERRS program. Hughes et al. (2011) demonstrated that aRPD was less than 2 cm in 10 of 18 sites in their study.

Similarly, toxins are not routinely monitored. Ample evidence exists that toxins are present and affecting marine mammals (see Section 2.2 for details).

2.6 Summary

This chapter provided an updated conceptual model of Elkhorn Slough, building on the strong foundation of decades of research and synthesis of the NERRS staff and collaborators.

This conceptual model was used as the basis to choose indicators of eutrophication that are relevant for Elkhorn Slough habitats, which span from salt marsh, mudflats to subtidal habitat. These consisted of primary indicators (macroalgal biomass, phytoplankton chlorophyll-a, sediment nitrogen, surface water DO and pH) and supporting indicators (dissolved inorganic nutrients, HAB toxins and macroalgal % cover).

The science supporting selection of thresholds was reviewed, based largely on the work of Sutula (in prep) and literature cited therein. This review was the basis for Central Coast Water Board selection of provisional biostimulatory numeric targets that it is considering. The targets were applied to 12 years of NERRS water quality and volunteer monitoring data to illustrate their application and to encourage discussion among stakeholders on the final selection of targets that they will present in their staff report.

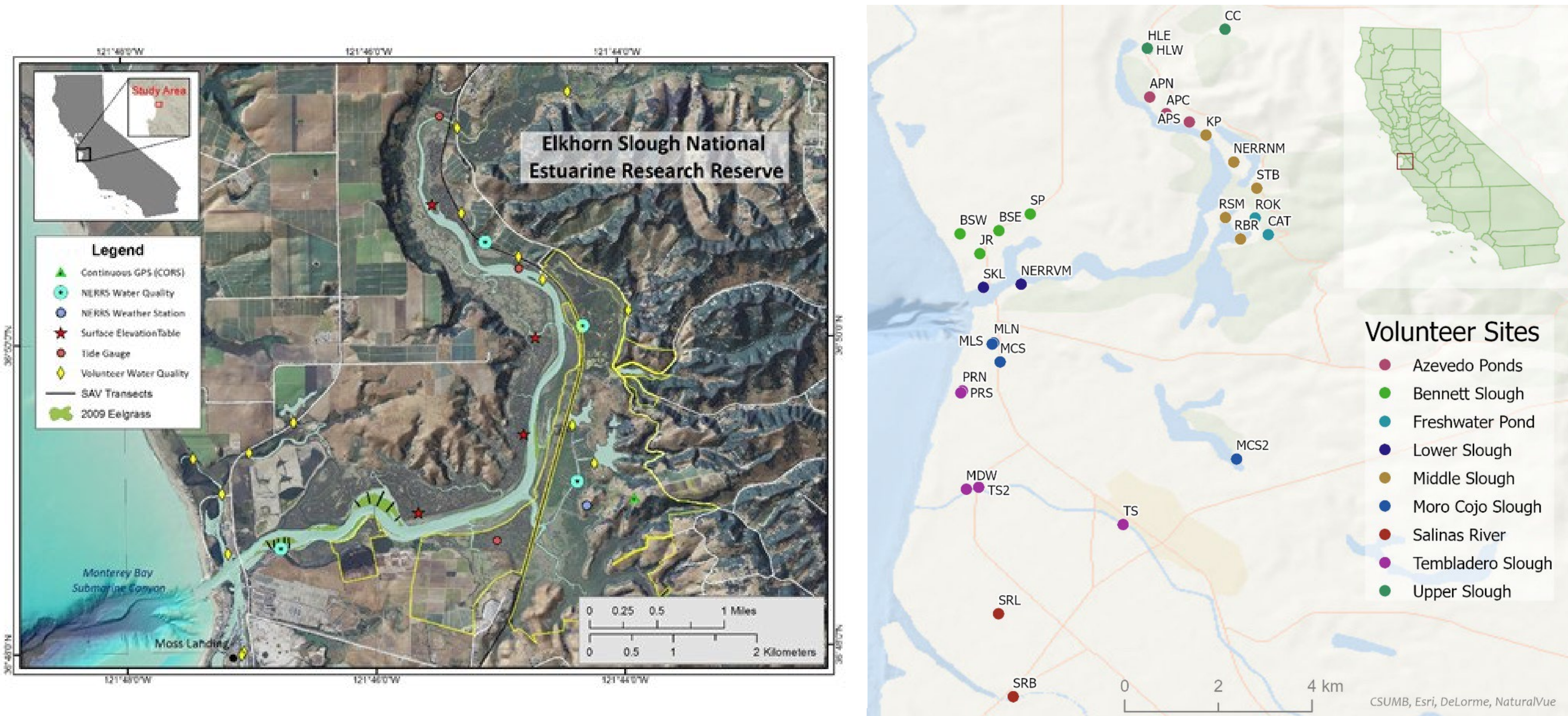
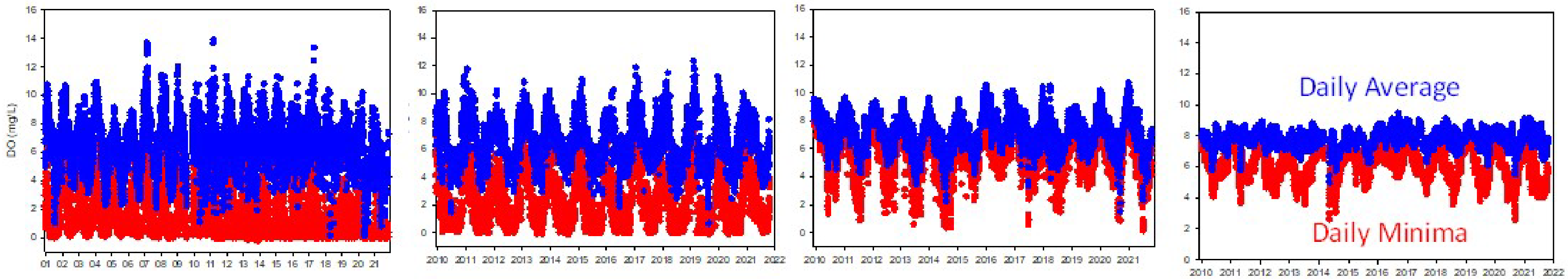


Figure 2.7. Top Panel: Location of NERRS water quality monitoring stations along main channel of the Slough, designated in green dots. Bottom Panel: location of 24 NERRS volunteer monitoring data.



Azevedo Ponds (ELKAP)

Year	10th Percentile of	
	DODailyMin	DODailyAve
2010	0.3	4.2
2011	0.0	3.6
2012	0.2	4.5
2013	0.1	3.7
2014	0.2	4.9
2015	0.1	4.7
2016	0.2	5.0
2017	0.1	4.9
2018	0.0	4.5
2019	0.1	5.0
2020	0.0	2.3
2021	0.0	2.7
Frequency	12/12	12/12

North Marsh (ELKNM)*

Year	10th Percentile of	
	DODailyMin	DODailyAve
2010	0.4	4.1
2011	0.4	4.3
2012	0.4	4.6
2013	0.3	4.5
2014	0.4	4.4
2015	0.3	5.0
2016	0.3	4.6
2017	0.4	4.8
2018	0.3	4.9
2019	0.2	4.0
2020	0.1	4.1
2021	0.3	4.5
Frequency	12/12	12/12

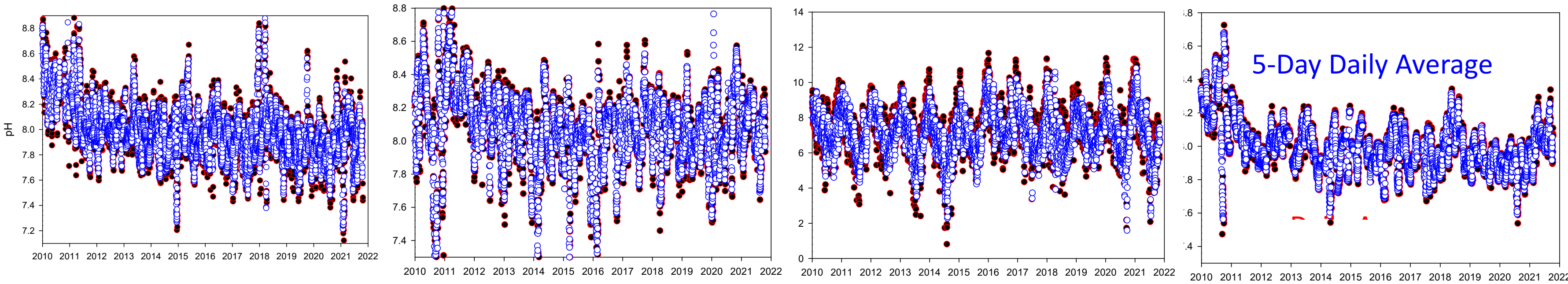
South Marsh (ELKSM)*

Year	10th Percentile of	
	DODailyMin	DODailyAve
2010	2.5	4.6
2011	1.9	4.6
2012	1.9	5.2
2013	1.7	4.8
2014	1.6	4.6
2015	3.8	6.1
2016	2.3	5.2
2017	3.2	5.7
2018	2.3	5.1
2019	1.7	4.7
2020	4.0	5.8
2021	4.7	6.4
Frequency	12/12	12/12

Vierra Mouth (ELKVM)

Year	10th Percentile of	
	DODailyMin	DO Daily Ave
2010	5.22	6.93
2011	5.26	7.41
2012	4.83	7.12
2013	4.59	6.84
2014	4.80	6.54
2015	5.01	6.96
2016	5.81	7.42
2017	5.21	7.02
2018	4.80	7.17
2019	4.81	7.05
2020	4.25	6.91
2021	4.35	6.73
Freqency	**12/12**	**6/12**

Figure 2.8. Top Panel: Graphical output of daily averaged (in blue) and daily DO minima (red) for the four Elkhorn Slough NERRS sites. The Azevedo Ponds towards the head of the estuary show pervasive low DO, which lessens in downstream sites within increasing mixing of highly oxygenated ocean waters. Bottom Panel: the 10th percentile of daily DO minima (the provisional target) versus the daily average over each year, which when assessed against the threshold (7 mg/L) can be used to calculate the frequency in which the target is not met (shaded red and blue boxes at the bottom).



Year	10th Percentile of pHDailyAve	10th Percentile of 5-DayDailyAve
2010	8.08	8.13
2011	7.87	7.91
2012	7.82	7.84
2013	7.73	7.75
2014	7.71	7.74
2015	7.75	7.77
2016	7.73	7.75
2017	7.63	7.66
2018	7.74	7.76
2019	7.64	7.66
2020	7.63	7.66
2021	7.56	7.60
Frequency	9/12	9/12

Year	10th Percentile of pHDailyAve	10th Percentile of 5-DayDailyAve
2010	7.48	7.55
2011	8.13	8.17
2012	7.84	7.86
2013	7.86	7.88
2014	7.77	7.79
2015	7.75	7.76
2016	7.60	7.59
2017	7.90	7.91
2018	7.85	7.84
2019	7.85	7.87
2020	7.77	7.80
2021	7.88	7.89
Frequency	5/12	4/12

Year	10th Percentile of pHDailyAve	10th Percentile of 5-DayDailyAve
2010	7.78	7.79
2011	7.60	7.61
2012	7.80	7.81
2013	7.70	7.70
2014	7.63	7.66
2015	7.60	7.65
2016	7.80	7.80
2017	7.60	7.65
2018	7.70	7.70
2019	7.63	7.65
2020	7.65	7.66
2021	7.47	7.45
Frequency	10/12	10/12

Year	10th Percentile of pHDailyAve	10th Percentile of 5-DayDailyAve
2010	8.00	8.00
2011	7.93	7.93
2012	7.58	7.84
2013	7.87	7.87
2014	7.77	7.79
2015	7.87	7.87
2016	7.78	7.79
2017	7.80	7.80
2018	7.88	7.88
2019	7.82	7.82
2020	7.78	7.79
2021	7.90	7.93
Frequency	4/12	3/12

Figure 2.9. Top Panel: Graphical output of 5-day running average (in blue) and daily averaged pH for the four Elkhorn Slough NERRS sites. Bottom Panel: the 10th percentile of daily average versus 5 day running average over each year, which when assessed against the threshold (7.8) can be used to calculate the frequency in which the target is not met (shaded red and blue boxes at the bottom).

Azevedo Ponds (ELKAP)

Year	DIN	NH4	PO4
2010	0.50	0.14	0.89
2011	0.82	0.16	0.09
2012	0.47	0.16	0.13
2013	0.20	0.10	0.09
2014	0.42	0.19	0.08
2015	0.19	0.16	0.07
2016	0.34	0.26	0.14
2017	1.10	0.32	0.19
2018	0.49	0.28	0.19
2019	1.74	0.49	0.07
2020	0.56	0.26	0.31
Frequency	5/12	12/12	8/12

North Marsh (ELKNM)*

Year	DIN	NH4	PO4
2010	0.47	0.09	0.10
2011	0.50	0.08	0.09
2012	0.30	0.16	0.07
2013	0.32	0.12	0.07
2014	0.29	0.11	0.05
2015	0.21	0.13	0.06
2016	0.37	0.16	0.08
2017	1.26	1.10	0.08
2018	0.41	0.29	0.09
2019	0.74	0.18	0.06
2020	0.58	0.43	0.05
Frequency	4/12	9/12	3/12

South Marsh (ELKSM)*

Year	DIN	NH4	PO4
2010	0.58	0.12	0.10
2011	0.93	0.10	0.07
2012	0.44	0.14	0.06
2013	0.46	0.12	0.07
2014	0.46	0.14	0.06
2015	0.41	0.11	0.05
2016	0.45	0.12	0.06
2017	0.82	0.31	0.08
2018	0.73	0.34	0.05
2019	0.92	0.18	0.05
2020	0.51	0.33	0.04
Frequency	6/12	12/12	1/12

Vierra Mouth (ELKVM)

Year	DIN	NH4	PO4
2010	1.29	0.14	0.11
2011	0.75	0.08	0.05
2012	0.48	0.09	0.05
2013	0.77	0.08	0.06
2014	0.52	0.11	0.06
2015	0.85	0.08	0.05
2016	0.42	0.09	0.05
2017	1.37	0.25	0.12
2018	0.53	0.28	0.05
2019	1.64	0.14	0.06
2020	0.55	0.23	0.03
Frequency	9/12	6/12	2/12

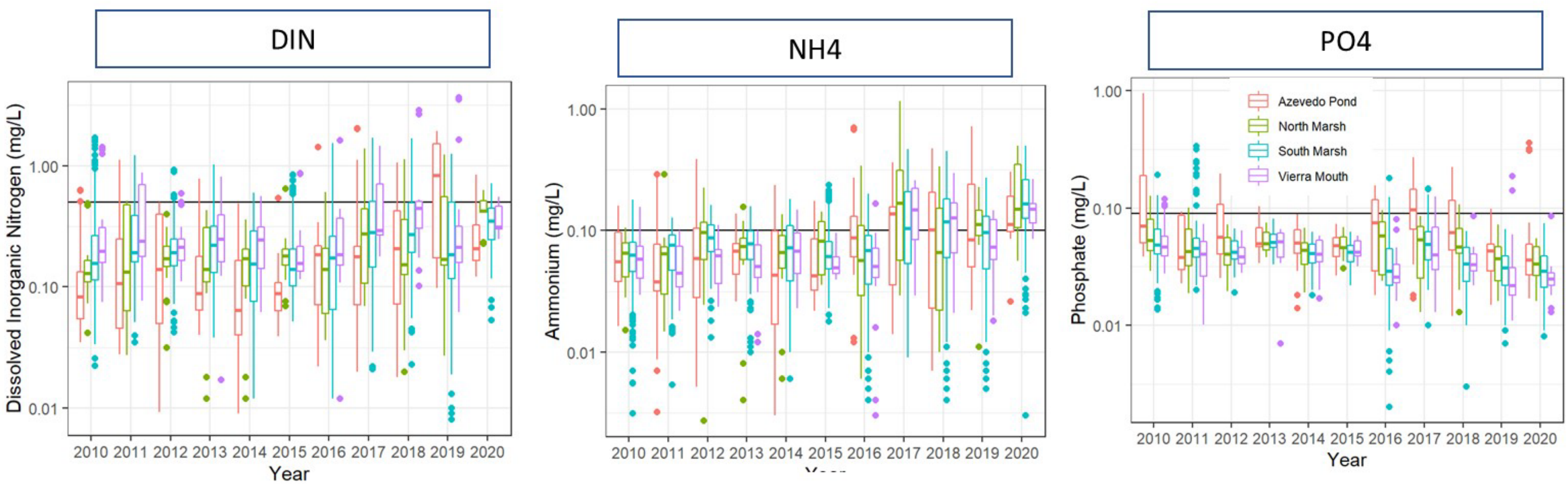


Figure 2.10. Top Panel: Summary of the annual 90th percentile of nutrient concentrations at the four NERRS water quality sites over the 12 years, with the fraction of years in which the provisional targets, including DIN (0.5 mg/L), NH4 (0.1 mg/L) and phosphorus targets (0.09 mg/L), were not met. Bottom panel shows box-and-whiskers plots of the distribution of data for each site by analyte.

3. Maximum Allowable Loads and Other Strategies To Mitigate Eutrophication

3.1 Introduction

The Elkhorn Slough estuary exhibits eutrophic conditions caused by exceedances of water quality objectives for dissolved oxygen, pH, un-ionized ammonia, and chlorophyll-a, as well as nutrient-related problems caused by high levels of nitrate, orthophosphate, and algal biomass, referred to as biostimulatory substances. To support water quality improvement activities, the Water Board is considering a suite of biostimulatory targets to restore water quality in the estuary. Elkhorn Slough is a dynamic and complicated estuarine system with nutrient inputs from multiple sources: ocean water inflow, Old Salinas River, and freshwater tributaries in the upper watershed. This chapter synthesizes drivers of eutrophication, provides analyses that can support the establishment of total nitrogen (TN) and total phosphorus (TP) maximum allowable loads, and strategies to establish load allocations.

The approach to this endeavor relies on two linked and calibrated models (Figure 3-1): 1) the calculation of watershed loads to the Slough, for which a Soil & Water Assessment Tool (SWAT; Tetra Tech 2021) is used. The second component consists of coupled hydrodynamic and water quality receiving water models that provide the linkage from external loads of nutrients (whether from the watershed, from Monterey Bay and the Old Salinas River, or internally generated) to the nutrient distribution within the waters of the Slough and associated biologic responses. The hydrodynamic model of mixing and transport within the Slough is based on the Environmental Fluid Dynamics Code or EFDC (Hamrick 1992, 1996; Tetra Tech 2007). The water quality model, the EPA-supported Water Analysis Simulation Program (WASP; version 8.4.0) has an advanced eutrophication (EUTRO) module that simulates pollutant fate and transport, along with biogeochemical reactions and eutrophication (Tetra Tech 2022).

The reader is referred to the individual Tetra Tech (2021, 2022) reports for details on the development, implementation, and calibration of the modeling tools. A brief summary is provided below.

An aim of the modeling is to summarize knowledge of the system and its interlinked parts. The models provide a process-based representation of the connection between stressor sources (e.g., nutrient loads) and responses of interest in the Slough (e.g., dissolved oxygen (DO) and algal growth). To the extent that the process-based representation correctly represents reality, the models can then be used to conduct scenario tests of how the system may respond to reductions in nutrient loads, for the purpose of estimating maximum loading rates consistent with supporting beneficial uses as defined by the targets defined in Chapter 2.

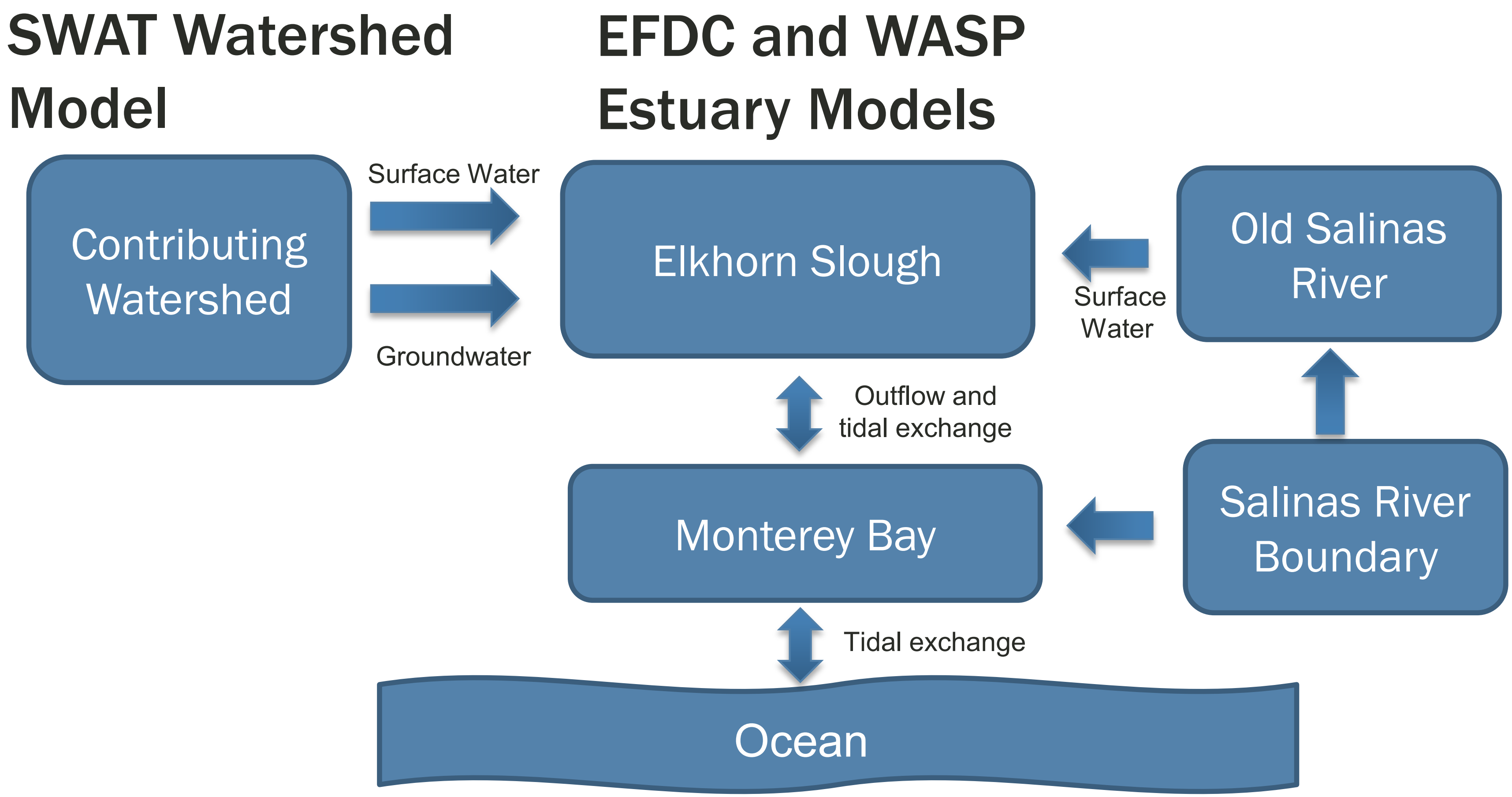


Figure 3.1. Conceptual view of tools supporting the synthesis of environmental drivers and calculation of maximum allowable loads.

3.2 Methods

3.2.1 Description of tools

SWAT Model

Flow and loads from the Elkhorn Slough watershed are estimated with the SWAT+ model (Bieger et al. 2017). The SWAT+ model provides estimates of flow and water quality loads directly to Elkhorn Slough as well as to its tributary, the Old Salinas River, which receives watershed runoff from the Moro Cojo and Gabilan/Tembladero watersheds, as well as releases from the Salinas River Lagoon. The model calibration summary (Table 3.1) provides an overview of SWAT model performance, which is generally acceptable based on a suite of performance criteria (very good: ≤ 20%, good: 20-30 %, fair: 30-45%, poor: > 45%). Watershed loading from discharge of groundwater from deep aquifers is not fully simulated by SWAT.

Table 3.1. From Tetra Tech (2021). SWAT model performance for water quality.

Name	Constituent	Paired Concentration (Mean, mg/L)			Paired Load (Mean, lbs/d)		
		Obs	Sim	Percent Error (Obs - Sim)	Obs	Sim	Percent Error (Obs - Sim)
Carneros Creek	NO_3+NO_2	1.424	1.044	26.7	100.4	84.1	16.4
	TN	1.453	1.229	15.4	128.6	124.9	2.3
	PO_4	0.425	0.347	18.4	24.7	27.0	-7.9
Moro Cojo Slough	NO_3+NO_2	0.914	1.200	-31.3	14.4	23.6	-62.1
	TN	1.100	1.024	6.9	53.7	29.1	45.7
	PO_4	0.621	0.631	-1.7	19.7	11.2	43.0
Tembladero Slough	NO_3+NO_2	28.296	26.426	19.6	3,897.6	3,865.0	31.0
	TN	32.870	27.902	1.4	5,604.7	3,421.7	14.2
	PO_4	0.499	0.488	2.2	94.1	78.1	17.0
	TP	0.598	0.608	-1.8	66.5	65.4	1.6

EFDC+ WASP Models

The EFDC simulation focuses on the conservation of mass and momentum of water in Elkhorn Slough. In addition to tidal forcing and external flow, the hydrodynamics are influenced by water density, which in turn depends on temperature and salinity. Therefore, both heat and salt content of water are simulated within EFDC and passed forward to the water quality model. EFDC calibration results are summarized in Table 3.2. Generally, hydrodynamic model performance is good in the main slough and Azevedo Pond, but less adequate in the North and South Marsh areas.

Simulation of water quality involved targeted nutrients, oxygen, pH, and algae, represented by both phytoplankton (generalized as one group) and macroalgae, which was conceptualized as attached to represent what appears to be the dominant form *(Ulva spp.)* found on intertidal and subtidal habitat

(Figure 3.2). Calibration was conducted for the year 2012, based on high availability of both the MBARI LOBO moorings (Table 3.3.) and the NERRS data (Table 3.4). We note that statistics of calibration compare point-in-space measurements to spatial averages over the relatively large WASP segments and that the calibration is a compromise to obtain the simultaneous best fit over multiple parameters and multiple stations. Generally, dissolved oxygen and dissolved inorganic nutrients are adequately simulated in the main Slough and in the tidally restricted areas. Error was larger in predicted chlorophyll-a, but no systematic bias was noted. Because data were not available to calibrate the WASP model for macroalgal biomass, then model tuning to balance algal uptake by phytoplankton versus macroalgae could not be achieved.

Table 3.2. EFDC hydrodynamic model performance. Nash-Sutcliffe = NS. Average absolute error = AAE.

Statistic	LOBO 1	LOBO 4	LOBO 2		ELKVM	ELKSM	ELKNM	ELKAP	LOBO 3
Water Surface Elevation (m)									
Average error	0.004	-0.007	-0.003		-0.005	0.048	-0.001	-0.029	-0.003
AAE	0.077	0.097	0.183		0.095	0.270	0.050	0.047	0.086
NS	0.940	0.906	0.756		0.992	0.462	-0.082	0.759	0.912
Salinity (psu)									
Average error	-0.020	-0.178	-1.623		0.021	-0.589	-1.722	-2.523	-1.677
AAE	0.519	0.536	1.110		0.595	0.846	2.354	2.849	9.900
NS	0.538	0.612	0.994		0.235	0.522	0.152	-0.451	0.181
Water Temperature (°C)									
Average error	-0.144	0.448	0.163		0.122	1.045	0.817	-2.080	0.970
AAE	0.901	1.146	0.798		0.862	1.343	1.414	3.266	1.008
NS	0.645	0.744	0.807		0.620	0.699	0.796	0.035	0.614

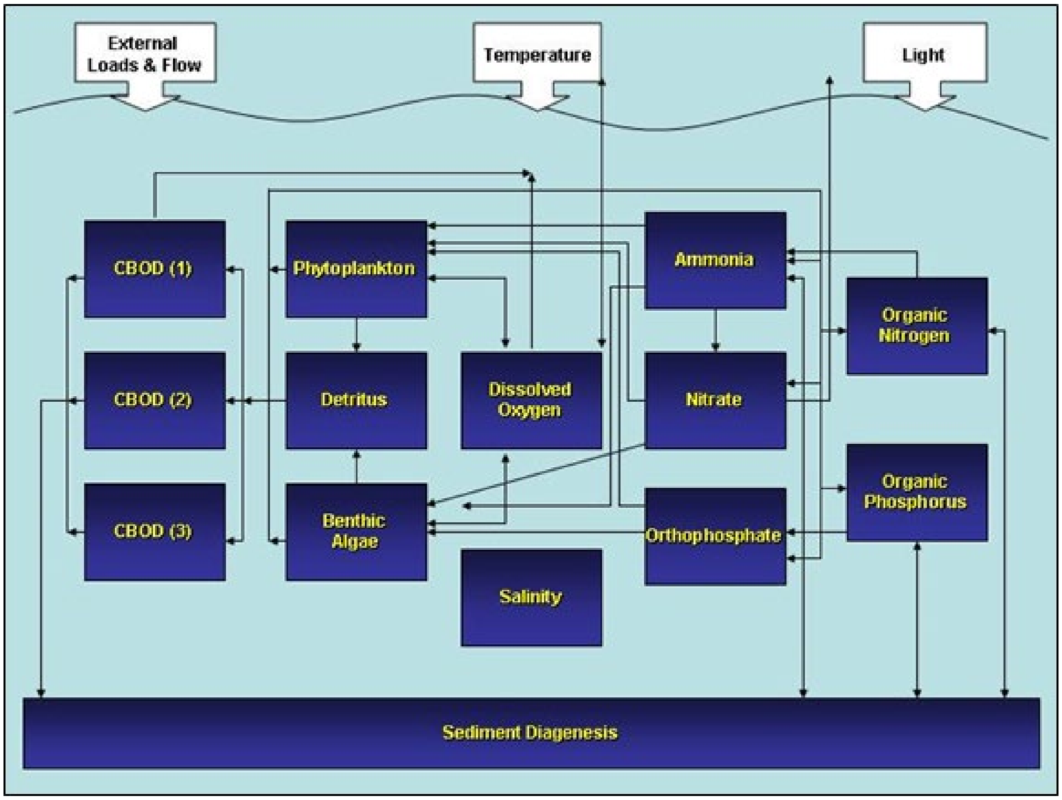


Figure 3.2. Schematic of the WASP Eutrophication (EUTRO) Module.

Table 3.3. Water Quality Calibration Statistics for LOBO Moorings 2012.

	L01								
	NO_3 (mg/L)			DO (mg/L)			Chlorophyll *a* (µg/L)		
	Observed	Simulated	Error	Observed	Simulated	Error	Observed	Simulated	Error
Count	8404			8617			8620		
Average	0.353	0.385	9.0%	7.312	7.423	1.5%	2.691	3.435	27.7%
Median	0.204	0.272	33.2%	7.507	7.553	0.6%	1.970	3.015	53.1%
25%le	0.118	0.142	20.2%	6.419	6.928	7.9%	1.610	2.032	26.2%
75%le	0.369	0.494	33.8%	8.313	8.068	-2.9%	2.470	4.784	93.7%
	L04								
	NO_3 (mg/L)			DO (mg/L)			Chlorophyll *a* (µg/L)		
	Observed	Simulated	Error	Observed	Simulated	Error	Observed	Simulated	Error
Count	8302			7995			8301		
Average	0.258	0.273	5.9%	6.569	6.590	0.3%	4.008	3.375	-15.8%
Median	0.171	0.156	-8.7%	6.725	6.809	1.2%	2.530	2.868	13.3%
25%le	0.091	0.042	-54.2%	5.657	5.919	4.6%	1.990	1.915	-3.8%
75%le	0.330	0.365	10.4%	7.650	7.386	-3.5%	3.580	4.760	33.0%
	LO2								
	NO_3 (mg/L)			DO (mg/L)			Chlorophyll *a* (µg/L)		
	Observed	Simulated	Error	Observed	Simulated	Error	Observed	Simulated	Error
Count	4963			5515			5487		
Average	0.279	0.275	-1.6%	5.914	5.184	-12.3%	5.450	5.118	-6.1%
Median	0.187	0.161	-13.9%	6.367	5.455	-14.3%	3.640	4.344	19.3%
25%le	0.085	0.076	-10.7%	4.186	4.025	-3.9%	2.750	2.311	-16.0%
75%le	0.398	0.350	-11.9%	7.797	6.392	-18.0%	7.745	7.855	1.4%

Table 3.4. Water Quality Calibration Statistics for NERR Continuous Monitoring Sites, 2012. O = observed. S = Simulated. E = Error.

	O	S	E	O	S	E	O	S	E	O	S	E	O	S	E
	NO_3 (mg/L)			DO (mg/L)			Chlorophyll *a* (µg/L)			PO_4 (mg/L)			NH_4 (mg/L)		
Elkhorn Slough, Vierra Mouth															
Count	13			9063			13			13			13		
Average	0.181	0.300	65.8%	7.782	7.724	-0.7%	2.910	3.199	9.9%	0.042	0.048	14.5%	0.064	0.084	31.7%
Median	0.122	0.245	100.2%	7.850	7.833	-0.2%	2.834	2.213	-21.9%	0.038	0.048	27.0%	0.066	0.089	34.3%
25%le	0.097	0.101	5.0%	7.100	7.300	2.8%	2.032	2.108	3.8%	0.037	0.036	-2.7%	0.048	0.049	1.7%
75%le	0.186	0.348	87.6%	8.467	8.262	-2.4%	3.296	4.718	43.2%	0.045	0.060	32.1%	0.078	0.113	44.9%
	Elkhorn Slough, Azevedo Pond														
Count	13			9006			13			13			13		
Average	0.077	0.032	-57.6%	6.198	6.938	11.9%	4.712	52.751	1019%	0.076	0.128	69.1%	0.078	0.192	145.4%
Median	0.048	0.005	-89.7%	6.100	5.613	-8.0%	4.438	12.339	178%	0.056	0.063	11.5%	0.042	0.071	68.9%
25%le	0.007	0.002	-65.7%	2.900	1.074	-63.0%	3.334	1.780	-46.6%	0.035	0.041	17.1%	0.023	0.024	6.5%
75%le	0.073	0.009	-87.2%	8.850	11.59	31.0%	5.355	92.497	1627%	0.107	0.204	90.3%	0.098	0.123	24.8%
	Elkhorn Slough, North Marsh														
Count	14			9125			14			14			14		
Average	0.084	0.084	0.4%	6.409	7.295	13.8%	13.60	20.007	47.1%	0.045	0.093	107.8%	0.094	0.474	404.8%
Median	0.040	0.024	-41.2%	6.125	7.117	16.2%	10.10	8.168	-19.2%	0.045	0.063	39.3%	0.088	0.333	277.4%
25%le	0.032	0.009	-72.4%	4.300	4.706	9.4%	5.131	1.016	-80.2%	0.033	0.052	57.2%	0.051	0.106	109.0%
75%le	0.078	0.039	-50.6%	8.075	9.680	19.9%	20.53	21.134	2.9%	0.054	0.103	88.7%	0.116	0.495	327.7%
	Elkhorn Slough, South Marsh														
Count	149			8954			149			149			149		
Average	0.139	0.220	58.3%	7.176	7.140	-0.5%	4.708	3.373	-28.4%	0.044	0.052	16.5%	0.092	0.121	31.6%
Median	0.087	0.105	20.8%	7.275	7.212	-0.9%	3.842	2.731	-28.9%	0.042	0.051	21.2%	0.089	0.124	40.0%
25%le	0.047	0.018	-61.4%	6.350	6.461	1.8%	2.399	2.002	-16.5%	0.037	0.041	10.8%	0.069	0.051	-26.2%
75%le	0.172	0.206	19.7%	8.100	7.899	-2.5%	5.947	5.101	-14.2%	0.053	0.062	16.5%	0.110	0.173	56.5%

3.2.2. Numeric Targets Evaluated

Models were applied to evaluate the drivers of eutrophication and synthesize maximum allowable loads, specifically with respect to those that are required to achieve the following numeric targets under consideration by the Water Board for the TMDL (see Chapter 2 for synthesis of the basis for these targets):

- Dissolved oxygen: 10th percentile of 7-day average of daily minima (7DADMin) > 7 mg/L
- Macroalgae biomass: < 30 g DW/m2 as average over 2 highest density months
- Chlorophyll-a: 90th percentile < 15 μg/L
- Water column DIN: 90th percentile < 0.5 mg/L
- Water column Ortho P: 90th percentile < 0.09 mg/L

3.2.3 Summary of Eutrophication Sources and Cycling

The SWAT and EFDC-WASP models were used to estimate the total annual contributions of the watershed, Monterey Bay, sediment diagenesis, and atmospheric deposition during model calibration year 2012 to budgets of total and dissolved inorganic nitrogen and phosphorus. These loads can then be used to link to indicators that impair aquatic life use, specifically: 1) macroalgal biomass and 2) low dissolved oxygen.

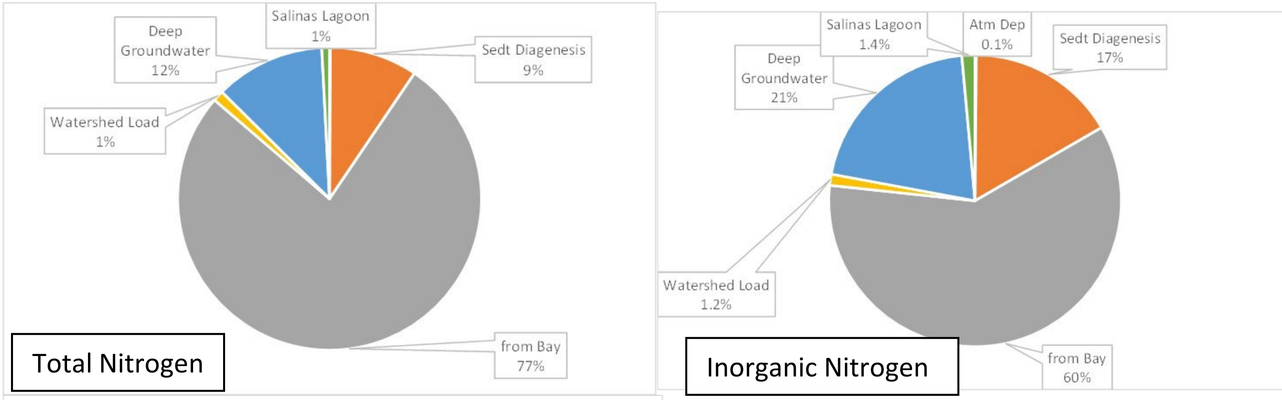
3.2.4 Load Reductions to Achieve Targets

We tested reductions in controllable sources of N and P loads jointly and separately. “Controllable sources” omits loads from Monterey Bay and atmospheric deposition. The results of joint reductions, where total N and P are reduced by the same amount, were very similar to N reductions only and thus are the focus of the analysis presented in this report. The joint reductions include reductions in nutrient regeneration from organic sediment because these ultimately derive from the watershed. This simplified approach was used as a means to gauge the load reduction needed, but in using this approach we do not assume that all sources can be reduced to the same extent. SWAT and nutrient regeneration from sediments estimated for 2012 were reduced by 10, 25, 50, 75, 90, and 95%. Concentrations of TN, TP, chlorophyll-a, macroalgal biomass and DO were plotted as a function of the percent reduction, using the targets under consideration as reference. Full details of the load reduction experiments are provided in Tetra Tech (2022).

3.3. Results and Discussion

Summary of Eutrophication Sources and Cycling in Elkhorn Slough

The results of the modeling synthesis of Elkhorn Slough eutrophication drivers supports the conceptual model provided in Section 2.2. Land-based loads of nutrients augment natural loads from atmospheric and Bay sources. Estuary-wide TN and inorganic N loads are summarized in Figure 3.3. The largest source of TN and TP loading is tidal flux from Monterey Bay (~60-77%). Watershed and releases from the sediment (sediment diagenesis) contribute roughly an equivalent amount of TN while sediment diagenesis is the dominant controllable source of TP (36%).



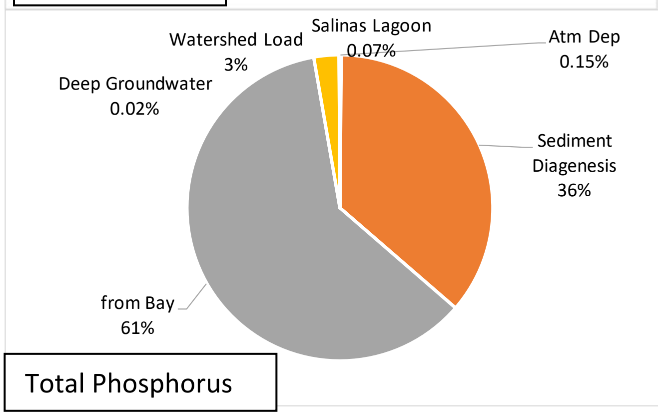


Figure 3.3. Summary of annual loads to Elkhorn Slough during the 2012 calibration year for total N, inorganic N and total P. Inorganic N watershed sources include deep groundwater and Salinas River Lagoon.

This relative contribution changes when focused on the restricted mixing areas within the estuary (Figure 3.4), where sediment diagenesis is the major source of TN and TP (~65%) and sources of TN and TP from the main Slough (including loads advected from the Bay) and from the watershed are minor (Figure 3.4). WASP does not provide a convenient means to tabulate all the internal cycling of N species between organic and inorganic forms within the water column, but the overall inorganic N balance ultimately depends on the direct loading of inorganic N (including releases of ammonia N from organic sediments).

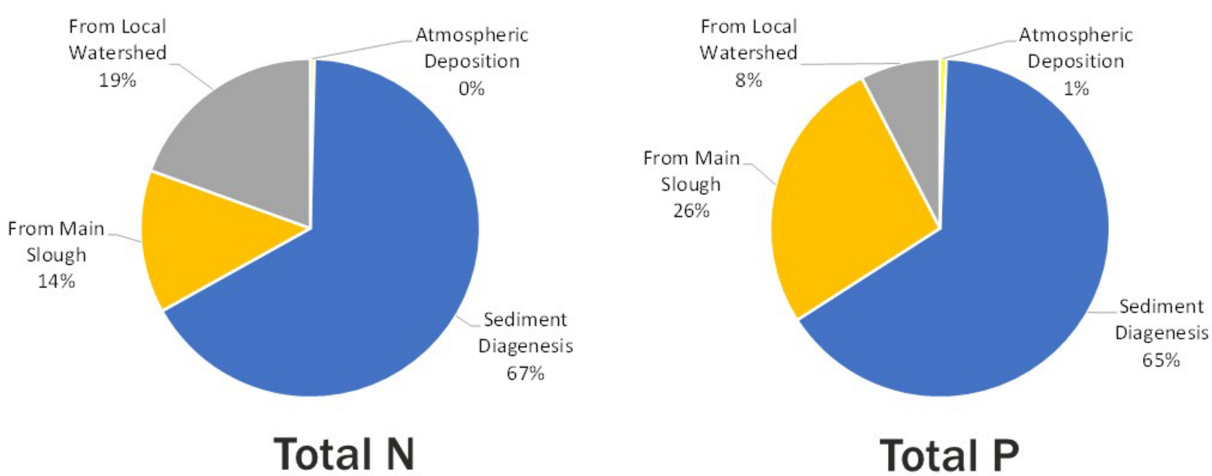


Figure 3.4. Summary of annual TN and TP loads to tidally restricted areas of Elkhorn Slough.

The overall inorganic N mass balance (Table 3.5) indicates that the Slough serves as a reactor that converts inorganic N to organic material (primarily as macroalgal biomass). As these extensive macroalgal blooms senesce and settle, the N-rich algal tissues replenish the supply available from the sediment, thus creating a chronic, self-sustaining eutrophication problem. McLaughlin et al. (2012) found that macroalgal sequestration of inorganic N diverts N away from denitrification (the only permanent means to eliminate nitrogen from the system) and thus sets up a cycle of macroalgal sequestration and regeneration that can cause rapid accumulation of sediment N over time. Furthermore, attached Ulva mats have been shown to drive concentrations gradients across the sediment water interface that promote the efflux of ammonium and phosphorus, accelerating macroalgal growth. Thus, while Bay source of N appear to dominate the whole estuary budget, macroalgal blooms can begin and thrive solely on the basis of enrichment sediment nitrogen sources. Tidal restrictions such as dikes or weirs can exacerbate this problem by limiting scouring of macroalgae from the estuary and increasing hydraulic retention time.

Table 3.5. Inorganic N Mass Balance for 2012.

Inorganic N Inputs: 4,346 Mg/yr	Total N Inputs: 7,652 Mg/yr
Inorganic N to Bay: 3,265 Mg/yr	Total N to Bay: 7,130 Mg/yr
Converted to Organic N or Lost: 1,081 Mg/yr	Net Retention and Loss: 522 Mg/yr

Note: 1 Mg = 1.1023 English tons

Nutrient loading from land-based sources is estimated to be highly variable by water year with loads that are much greater than 2012 in the wet years of 2009, 2016, 2017, and 2019. This suggests that the role of watershed versus Bay sources may be underestimated if calculated on the basis of 2012 alone. In addition, estimates of Bay sources currently include anthropogenic nitrogen that is discharged and mixed into Monterey Bay. Coastal modeling studies ongoing now will provide an opportunity to disentangle these complex sources but cannot currently inform our modeled nutrient mass balances.

For ongoing watershed sources, SWAT+ provides a breakdown of the estimated relative contribution of major source areas or inputs to the Slough (Table 3.6) and by land use (Figure 3.5). Old Salinas River is the largest source of TN and TP, followed by Upper Elkhorn Slough. Agriculture is the watershed land use that provides the greatest load of TN and TP (Figure 3.6).

3.6. Summary of calendar year 2012 TN and TP loads and percent contribution from source areas to the Slough.

Source Area	Total N (lbs./yr)	% Contribution TN	Total P (lbs./yr)	% Contribution TP
Old Salinas River	1,974,024	86%	15,355	53%
Lower Elkhorn Slough (to S. Marsh)	781	0.03%	303	1.0%
Bennett/Parsons	6,635	0.29%	1,182	4.1%
South Marsh	10,407	0.45%	1,078	3.7%
Upper Elkhorn Slough	275,682	12%	9,685	33%
North Marsh, Azevedo Ponds	40,721	1.8%	1,335	4.6%
Watershed Total	***2,308,251***	100%	***28,938***	100%

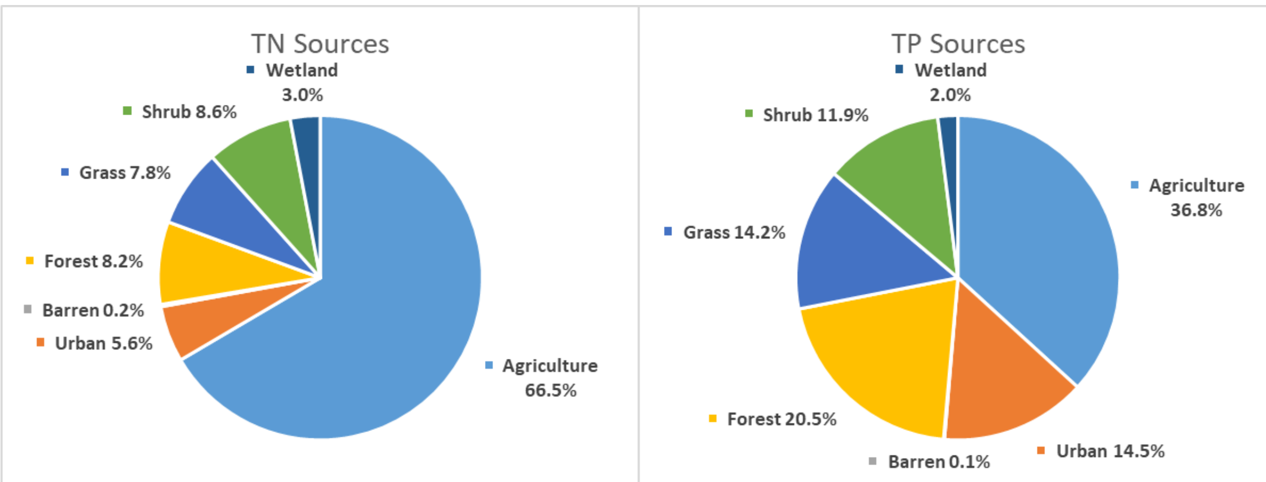


Figure 3.5. Calendar year estimates of Inorganic N Load from watershed to slough.

Linkage to Impaired Uses

Model analyses demonstrate that inorganic nitrogen from water column and sediment sources drives proliferation of macroalgae in the Slough. Macroalgal and episodic phytoplankton blooms increase in magnitude, frequency and duration, extinguishing light to and which trigger a host of adverse consequences including: 1) Shading or smothering of benthic diatoms, seagrass, and salt marsh habitat; 2) shift towards benthic or planktonic harmful algal species that produce toxins that can cause acute or chronic illness, including death in humans and wildlife; 3) organic matter accumulation in the sediments and water column causes a fundamental alteration of biogeochemical cycling, changes in sediment and water column oxygen demand and CO2 flux, and weakening of the structural integrity of salt marsh sediments, making them susceptible to erosion; 4) decreased recruitment and survival of benthic invertebrates in mudflats and subtidal habitat and reduced carrying capacities for fishes and shorebirds; 5) increased frequency of water column and sediment hypoxia and heightening heterotrophic bacterial activity, resulting in poor water quality and increased frequency of diseases; 6) drift macroalgal blooms can become thick wrack that can smother salt marsh, mudflats, and oyster reefs; 7) poor aesthetics and an increase in odors relating to the decomposition of organic matter and increased sulfide production; and 8) shifts in both trophic and community structure of invertebrates, birds, and fishes.

Summary of the monitoring data points to dissolved oxygen problems with respect to both strong diel variation and chronic low dissolved oxygen (Chapter 2). Dissolved oxygen levels peak during the day while algae photosynthesize, and hypoxia (low oxygen) becomes more extreme overnight while algae respire in the absence of light (Hughes et al. 2011). Strong swings in dissolved oxygen concentrations (hyperventilation) and pH are apparent in the upper areas of the Slough, particularly in areas with restrictions on tidal flow and high concentrations of nutrients (Chapin et al. 2004; Beck and Bruland 2000). Hypoxia has been shown to attribute to stress and lower survival in fish and oysters in the Elkhorn Slough (Jeppesen et al. 2016). Hughes et al. (2018) showed that even at DO concentrations below 4 mg/L, juvenile groundfish are extirpated from the Slough, thus impacting the important nursery habitat role that estuaries play.

DO concentrations in Elkhorn Slough represent the net effects of a number of different processes (Figure 3.6). This indicates that the dominant process increasing DO is production by macroalgae. This production is approximately balanced, in nearly equal parts, by sediment oxygen demand and macroalgal respiration. The contributions of other processes (respiration and production by phytoplankton, nitrification of ammonia, and oxidation of carbonaceous CBOD) are relatively minor contributors to the DO mass balance. Note that the net reaeration term represents the sum of influx of oxygen from the atmosphere to the water column (when water in the Slough is below saturation for DO) and efflux of oxygen from the water column to the atmosphere (when water in the Slough is above saturation for DO).

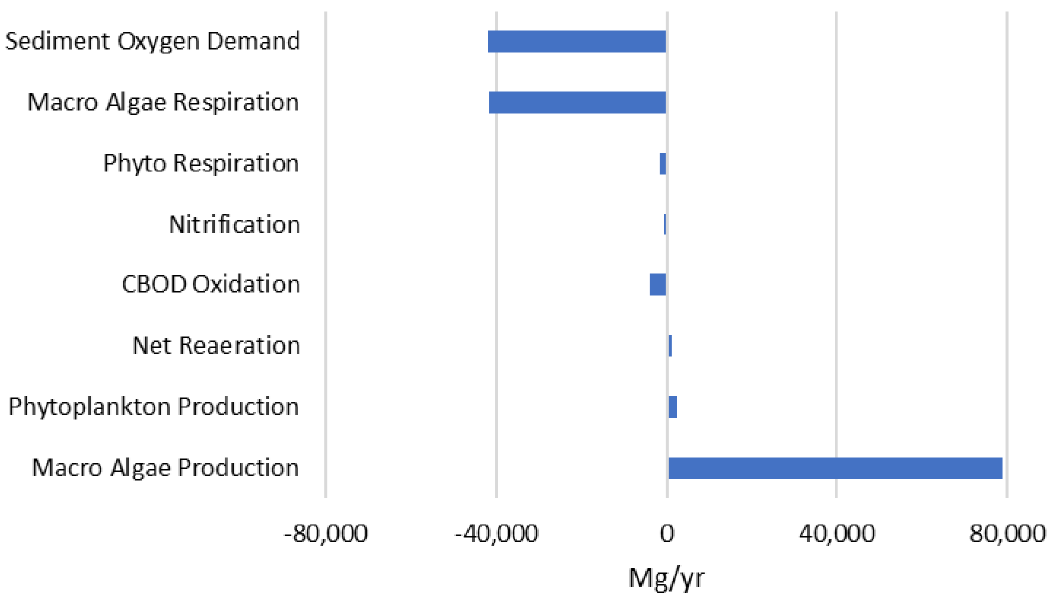


Figure 3.6. Model-estimated Sources and Sinks of DO for 2012

The WASP model was not able to simulate long-term interannual accumulation rates of sediment TN, another target under consideration by the Water Board. Chronic algal blooms accelerate organic matter accumulation in mudflat, and subtidal sediments. The accumulation of this large amount of labile organic matter stimulates microbial communities and increases the sediment oxygen demand and shallow redox potential (Cardoso et al. 2004). Zones of sediment anoxia and sulfate reduction, which are typically found at depths of 10 cm or more, become shallow, often extending throughout the sediment right under the algal mat (Dauer et al. 1981; Hentschel 1996). This leads to surficial pore water ammonia and sulfide concentrations that are toxic to epifauna and benthic invertebrates (Kristiansen et al. 2002), which are important forage for fish, resident and migratory birds, and marine mammals (Green et al. 2013). Under extreme organic matter loading or macroalgal blooms, the zone of sulfate reduction rises to the surface, often extending throughout the sediment under the algal mat (Dauer et al. 1981; Hentschel 1996). While most benthic fauna has evolved to deal with organic rich conditions (Pearson and Rosenberg 1978, Hargrave et al. 2008), the amount of sediment nitrogen as well as the degree of hypoxia or anoxia in overlying water can be an important factor structuring the community composition (e.g., Rosenberg et al. 1991; Diaz and Rosenberg 1995; Baustin and Rabalais 2009). The depth to the apparent redox potential discontinuity (aRPD), below which zone of anoxia and sulfate reduction are encountered (Cardoso et al. 2004), is a visual indicator of this problem; a boot print in estuarine mudflats can reveal a thin grey layer that overtops the dark anoxic and sulfidic smell right below. In a survey of Elkhorn Slough subtidal and mudflats, Hughes et al. (2011) found that 10 of 18 sites had aRPD values < 2 cm (Hughes et al. 2012), a threshold below which translates to reduced habitat volume and quality for benthic infauna and an alteration in their community structure (Sutula et al. 2014).

Nutrient Loading Capacity Analyses and Strategies to Allocate Load Reductions

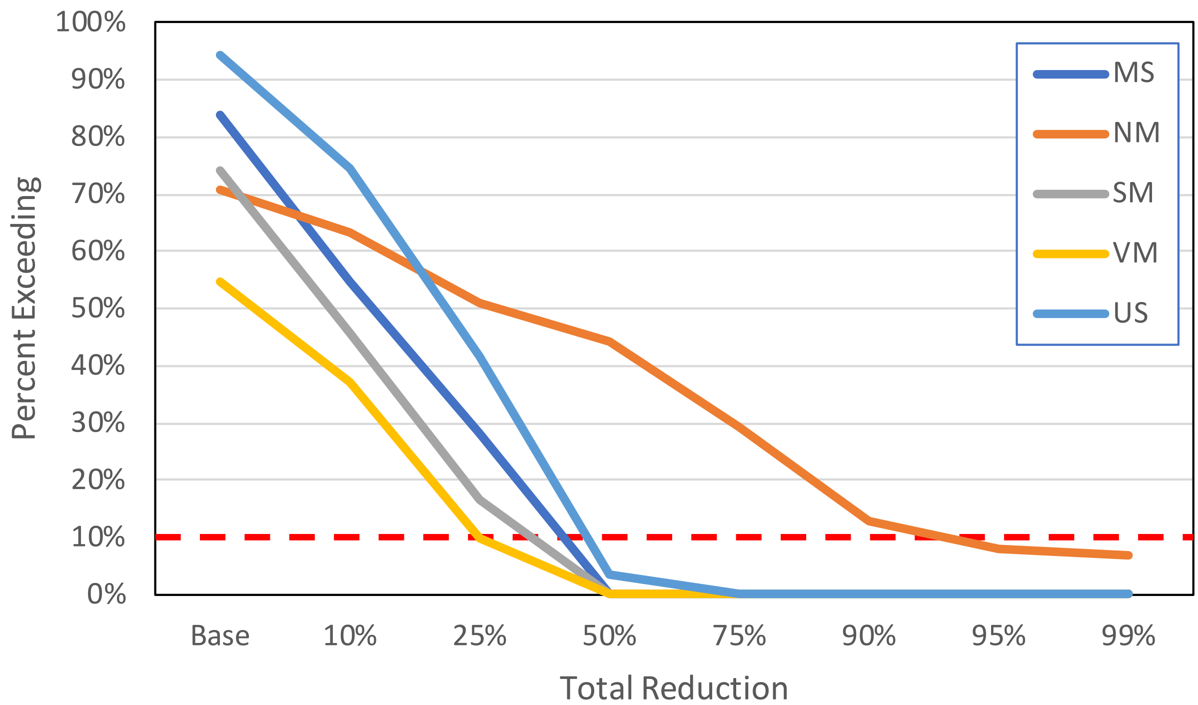
Simulations of nutrient reductions from 10-95% provided an integrated estimate of load reductions required for tidally mixed versus restricted mixing regions of the Slough (Table 3.7-3.8).

In tidally mixed areas, the target driving the most stringent reductions was DO (requiring up to 50% reduction of TN and TP to meet the 7-day average of daily minima or 7DADMinDO; Figure 3.8). To meet the dissolved inorganic N and P targets, reductions of 40% would be required, while the planktonic chlorophyll-a target was met under the current loading scenario. For 2012 model year, a 50% reduction of controllable sources is equivalent to a loading capacity (all sources) of 7,463 tons/year of total nitrogen and 425 tons/year of total phosphorus (Table 3.8). A 50% reduction in watershed and sediment loads equals about a 12% reduction in N load and 19% reduction in P load from all sources, including a margin of safety.

In contrast, in the area designated as “restricted mixing” (e.g., North Marsh, Figure 3.8), reductions of N and P loads of up to 92% would be needed to meet the 7DADMinDO target, while 45% reduction in loads would be required to meet the phytoplankton chlorophyll-a target and 15-20% required to meet the DIN and phosphate targets. For such a target to be achieved, options such as restoration, which could bring in either increased tidal flushing and/or removal of the thick layer of organically enriched sediments, could be considered.

Modeled macroalgal densities never exceeded the target of 30 g dw m^{-2} and thus could not inform load reductions. This could be due in part because: 1) data were not available to calibrate biomass, and 2) the model averages over the spatial patchiness that occurs with the current resolution of model grid, whereas the target was developed from observations of densities within patches. Thus, uncertainty exists in the efficacy of load reduction to address impairments due to macroalgal blooms.

Nutrient load reductions could be achieved in a number of ways including, 1) proportional, in which reductions are assigned according to the contribution of jurisdiction and land use to the total



load, and 2) disproportional, in which one sector or jurisdiction receives greater share, based on criteria such as their role in legacy or nutrient organic matter accumulation. At the time at which this report was drafted, Water Board staff are still considering policy options and thus further analyses was not requested.

Figure 3.7, DO target (7DADMin) response to N and P reductions by location in Elkhorn Slough, where MS = main slough, NM = North Marsh, SM = South Marsh, VM = Vierra mouth, and US = Upper Slough. NM has highly restricted tidal mixing.

Table 3.7. Load reductions required to meet individual biostimulatory numeric targets under consideration for tidally mixed versus restricted mixing areas.

Target	Tidally Mixed	Restricted Mixing
7DADMin DO	25 – 50%	92%
Planktonic Chlorophyll	0%	45%
Dissolved Inorganic N	30 – 40% (not achievable in lower Slough due to elevated concentrations in Monterey Bay)	15%
Ortho-phosphorus	0 – 30%	20%

Table 3.8. Summary of controllable loads to tidally mixed areas from watershed and sediment diagenesis.

Source Area	Total N (ton/yr)	Total P (ton/yr)
Total Load (all sources)	***8,435***	***528***
Watershed Total	***1,154***	***14.5***
Sediment Diagenesis	790	191
Total Controllable Loads	1,944	205.5
Total with 50% Reduction in Controllable Loads	7,463	425

Other Strategies to Mitigate Eutrophication

Ecosystem restoration is a viable and appropriate management action to consider in the implementation plan, given the fact that eutrophication problems in the tidally restricted areas are likely to continue, even if watershed nutrient loads are drastically reduced. Four types of actions could directly address biostimulatory conditions (hydromodification and physical habitat alteration) and would likely improve eutrophication symptoms:

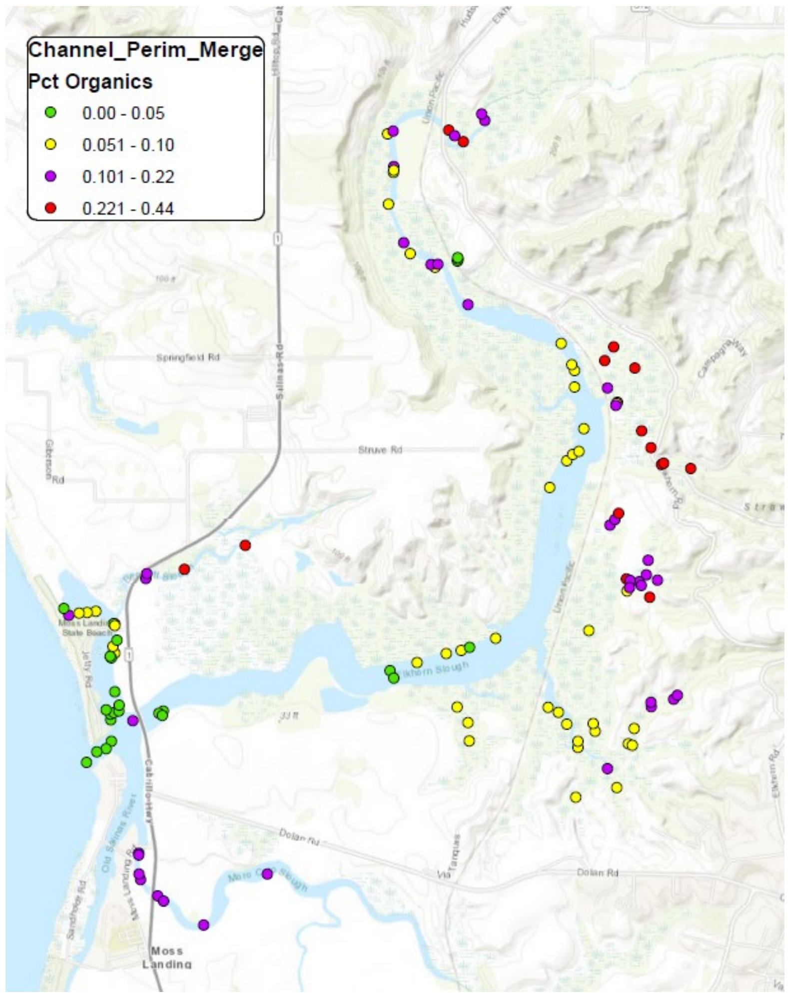


Figure 3.8. Distribution of organic rich sediments in Elkhorn Slough, as measured by ash-free dry mass. Figure courtesy of L. Harlan.

1) Improve tidal exchange and circulation to reduce hydraulic retention times and improve ability of estuary to flush out fine grained sediments;

2) Increase the area of moderate to high elevation salt marsh, in order to enhance wetting/drying that can drive greater rates of denitrification;

3) Remove sediments high in sediment nitrogen, particularly those in tidal restricted areas (Figure 3.8); and

4) Add buffers of riparian or other native vegetation designed to decrease nutrient loading from upland areas.

A combination of these actions, in combination with watershed nutrient load reductions, are anticipated to be the most effective approach to address eutrophication in the system, as it would remove legacy sediment enrichment and reduce accumulation of new sources of organic matter. These actions are also consistent with the Elkhorn Slough Tidal Restoration Project goals (see inset box), which noted (ESA 2014):

> Elkhorn Slough Tidal Restoration Project Goals (ESA 2014)
>
> 1) Reduce tidal scour by adding sediment to historically diked and drained areas.
> 2) Protect and improve surface water quality in Elkhorn Slough by establishing a permanent vegetative buffer.
> 3) Increase the extent of tidal marsh of sufficient elevation to be resilient to moderate sea level rise.

"Fifty percent of the tidal salt marsh in Elkhorn Slough has been lost in the past 70 years. This habitat loss is a result of increased tidal flooding, which "drowns" the vegetation, caused by past diking and draining of the marsh and construction of a harbor at the mouth of the Slough in 1947. The loss of riverine sediment inputs, subsidence of marsh areas, sea level rise, increased salinity, and increased nutrient inputs may also contribute to marsh drowning (Watson et al. 2010, Deegan et al. 2012). Bank and channel erosion in Elkhorn Slough are deepening and widening tidal creeks, causing salt marshes to collapse into the channel, and eroding sediments that provide important habitat and support estuarine food webs. "

"Habitat loss is expected to become more severe with accelerating sea level rise."

The TMDL implementation plan could provide incentives to contribute to the ongoing restoration of Elkhorn Slough. For example, the Water Board could offer flexibility to update the allowable loads to Elkhorn Slough pending an analyses of required load reductions post-restoration. Formal nutrient trading schemes could be established, in which nutrient reduction credits could be "traded" for contributions to restoration.

4. Summary of Findings, Uncertainties, and Science Recommendations

Summary of Findings

Eutrophication Conceptual Model Provides Options for Multiple Biostimulatory Targets. We developed a conceptual model of the drivers and impacts of eutrophication on human and aquatic life beneficial uses in Elkhorn Slough, building on the strong foundation of decades of research and synthesis of the NERRS staff and collaborators. The conceptual model was used to identify indicators and review the scientific basis for biostimulatory targets. Targets under consideration for the policy have now been selected by Water Board staff and, when applied to a decade of Elkhorn Slough monitoring data, we found that every subregion of the Slough systematically failed to meet most targets for most years, but conditions are worst in the tidally restricted areas.

Modeling Analyses Link Nutrient Loading and Biostimulatory Conditions to Harmful Algal Blooms and Hypoxia. This conceptual model of eutrophication drivers and responses was explicitly linked and quantified through calibrated watershed loading and Slough water quality models. These modeling analyses identified that: 1) inorganic nutrients are driving dense macroalgal blooms in the estuary, and 2) macroalgal blooms are the primary driver of DO diel swing and also contribute to sediment organic matter accumulation, which is another important sink for DO in the system. Published studies have documented that DO conditions are contributing to stress and lower survival in fish and oysters in the Elkhorn Slough, including documentation that existing conditions are causing juvenile groundfish to be extirpated from the Slough, thus impacting the important nursery habitat role of the Slough. Macroalgal blooms are smothering salt marsh, oyster reed and seagrass habitat and causing weakening and erosion of marsh banks, decreasing recruitment and survival of benthic invertebrates in mudflats and subtidal habitat and reduced carrying capacities for fishes and shorebirds, and creating poor aesthetics and an increase in odors relating to the decomposition of organic matter and increased sulfide production that can negatively affect ecotourism in this National Estuarine Research Reserve. Modeling analyses do not extend specifically to toxic HABs, but ample published literature links nutrient loading and eutrophication in the Salinas river valley to cyanotoxin related disease and mortality in sea otters and California sea lions.

While Modeling Analyses Suggest N Loading is Dominated By Natural Ocean Sources, Chronic Macroalgal Blooms Are Likely Fueled By Slough Sediments Enriched In Nitrogen. The Slough serves as a reactor that converts inorganic N to organic material (primarily as macroalgal biomass). As these extensive macroalgal blooms senesce, this N-rich algal tissues replenishes the supply available from the sediment, thus creating a chronic, self-sustaining eutrophication problem. In tidally mixed areas of Elkhorn Slough, the dominant source of TN and TP is Monterey Bay (~60-77%). Watershed and releases from the sediment (sediment diagenesis) contribute roughly an equivalent amount of inorganic N while sediment diagenesis is the dominant controllable source of TP (36%). Within tidally restricted areas, sediment diagenesis becomes the major source of TN and TP (~65%). Macroalgal sequestration of inorganic N diverts N away from denitrification (the only permanent means to eliminate nitrogen from the system) and thus sets up a cycle of macroalgal sequestration and regeneration that can cause rapid accumulation of sediment N over time. Furthermore, attached Ulva mats have been shown to drive concentrations gradients across the sediment water interface that promote the efflux of ammonium and phosphorus, accelerating macroalgal growth. Thus, while Bay source of N appear to dominate the whole estuary budget, macroalgal blooms can begin and thrive solely on the basis of enrichment sediment nitrogen sources. Tidal restrictions such as dikes, or weirs can exacerbate this problem by limiting scouring of macroalgae from the estuary and increasing hydraulic retention time. In tidally mixed areas of the Slough, a 50% reduction of controllable sources is needed, equivalent to a loading capacity (all sources) of 7,463 tons/year of total nitrogen and 425 tons/year of total phosphorus. In tidally restricted areas, a 92% reduction in controllable sources would be needed if no restoration actions are taken.

Management Recommendations Include Nutrient Load Reductions and Restoration to Mitigation Eutrophication In Elkhorn Slough. In addition to nutrient load reductions, four types of actions could directly address biostimulatory conditions (hydromodification and physical habitat alteration) and would likely improve eutrophication symptoms: 1) Improve tidal exchange and circulation to reduce hydraulic retention times and improve ability of estuary to flush out fine grained sediments; 2) Increase the area of intertidal habitat, in order to enhance wetting/drying that can drive greater rates of denitrification; 3) Remove sediments high in sediment nitrogen, particularly those in tidal restricted areas; and 4) Add buffers of riparian or other native vegetation designed to decrease nutrient loading from upland areas. These actions are entirely consistent with the Elkhorn Slough Tidal Restoration Project goals that are intended to increase ecosystem resilience to sea level rise and have been vetted with the local community.

Address Key Scientific and Modeling Uncertainties to Improve Capacity to Adaptively Manage Slough Overtime. The watershed loading and Slough water quality models are imperfect representations of the real world and should be considered as one line of evidence to combine with direct observation and scientific understanding to evaluate management plans. The scientific evidence of models and published studies point towards a compelling and immediate need to reduce nutrient loading to the Slough, while supporting the ongoing effort to restore it via hydrological and physical habitat restoration. Multiple data gaps exist that limited model applications. We recommend addressing these fundamental data gaps (see below) and refining coupled hydrodynamic and water quality models of the Slough over time, with the intent of improving the tool to adaptive manage the estuary overtime.

Uncertainties and Science Recommendations

Below are major science recommendations are intended to address uncertainties that underpin the Elkhorn Slough TMDL.

- Improve monitoring of eutrophication symptoms.
- Perform comprehensive monitoring of macroalgal biomass in intertidal and subtidal habitats, conducted in conjunction with cost-effective monitoring (e.g., via drones) of macroalgal cover to fine tune its use as a screening level monitoring indicator.
- Conduct monitoring to link sediment total nitrogen, total organic carbon, and grain size to macroalgal blooms and macroinvertebrate community composition.
- Routinely document algal toxins, including both cyanotoxins and marine biotoxins in particulate suspended matter and in shellfish.
- Quantify eutrophication drivers.

    - Quantify groundwater contributions to nutrient loading of Elkhorn Slough.
    - Improve quantification of freshwater and nutrient loads to Elkhorn Slough surface water sources (Table 3.6), including improved understanding of nutrient contributions from agricultural land use types.
    - Quantify the contribution of anthropogenic nitrogen (e.g., from San Francisco Bay and Monterey coastal sources) that represent "ocean sources" of nutrients to Elkhorn Slough.
    - Quantify benthic nutrient and oxygen fluxes in the habitat types of Elkhorn Slough.
- Improve numerical modeling capabilities to simulate eutrophication in response to local drivers and climate change as well as management initiatives. Specific applications include:
    - Assess the potential for seagrass restoration as a function of climate change (sea level rise) and eutrophication.
    - Quantify the effect of hydrological and physical habitat restoration in nutrient loading reduction required to meet biostimulatory targets.
- Improve understanding of efficacy of management measures.
    - Establish the quantitative basis for a nutrient trading scheme based on in -kind (nutrients) or restoration as an alternative measure.
    - Quantify efficacy of best management practices and other nutrient load reduction measures, particularly for agricultural land uses.
- Improve biostimulatory targets.
    - Improve the basis for compliance with dissolved oxygen objectives, with emphasis on spatial and temporal aggregation.
    - Improve the basis for phytoplankton biomass (chlorophyll-a) targets, presumably through a linkage with 1) protection of seagrass habitat, and 2) linkage with algal toxins.

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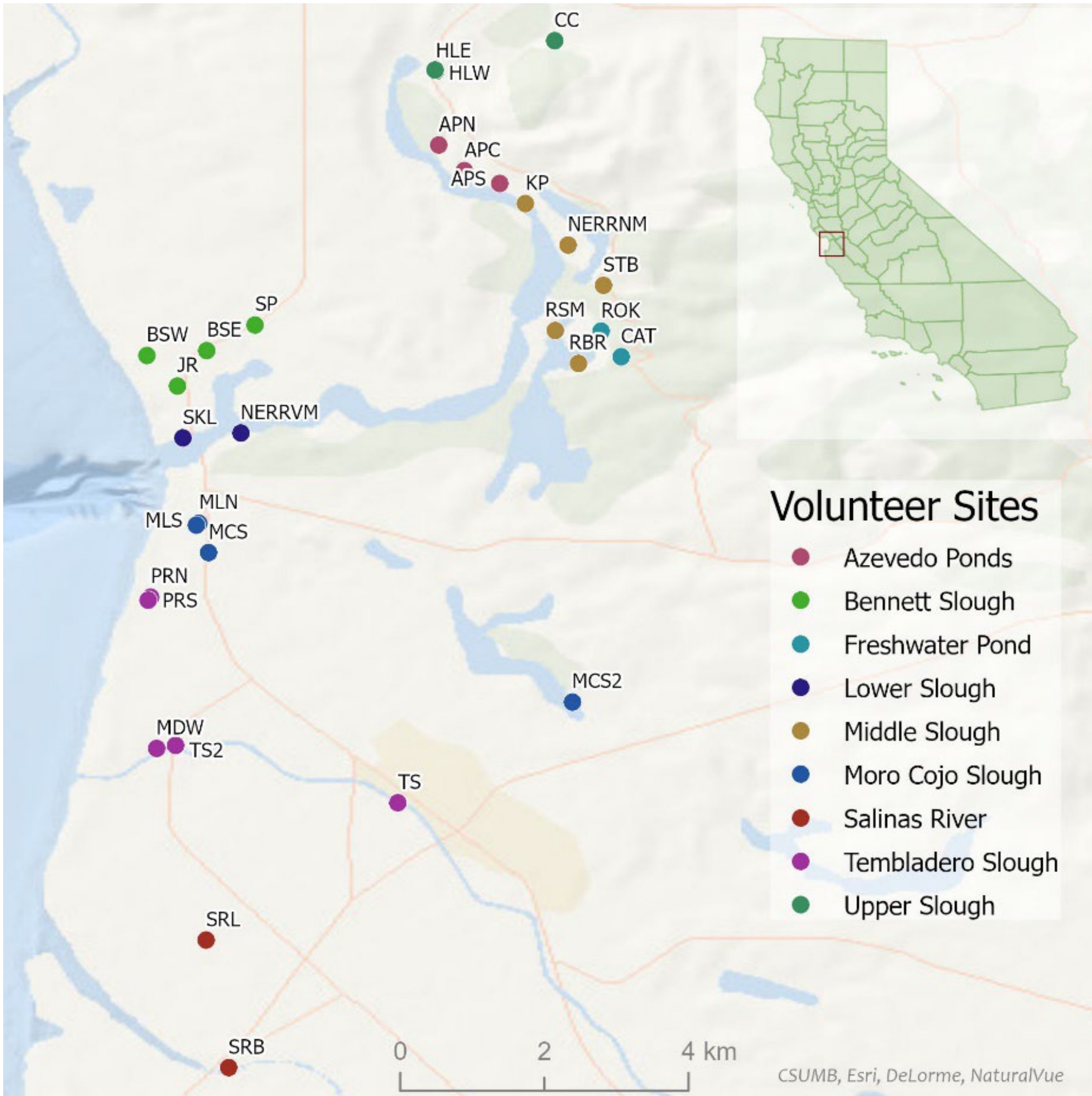
APPENDIX 1. APPLICATION OF PROVISIONAL TARGETS TO VOLUNTEER MONITORING SITES



Number of Years over 12-year Record in Which Target is Not Met

Upper Slough

- Carneros Creek (CC)*
- Hudson Landing East (HLE)
- Hudson Landing West (HLW)

Site	Statistic	DO	oatAlg%	Chl-a	DIN	Nh4	TP
CC	Eval. Statistic	12/12	6/12	12/12	12/12	12/12	12/12
HLE	Eval. Statistic	11/12	5/12	11/12	12/12	12/12	12/12
HLW	Eval. Statistic	12/12	3/12	11/12	10/12	11/12	12/12
Site	Statistic	DO	oatAlg%	Chl-a	DIN	Nh4	TP
CC	Median	12/12	0/12	6/12	11/12	11/12	12/12
HLE	Median	6/12	0/12	5/12	9/12	7/12	10/12
HLW	Median	8/12	0/12	1/12	2/12	6/12	7/12

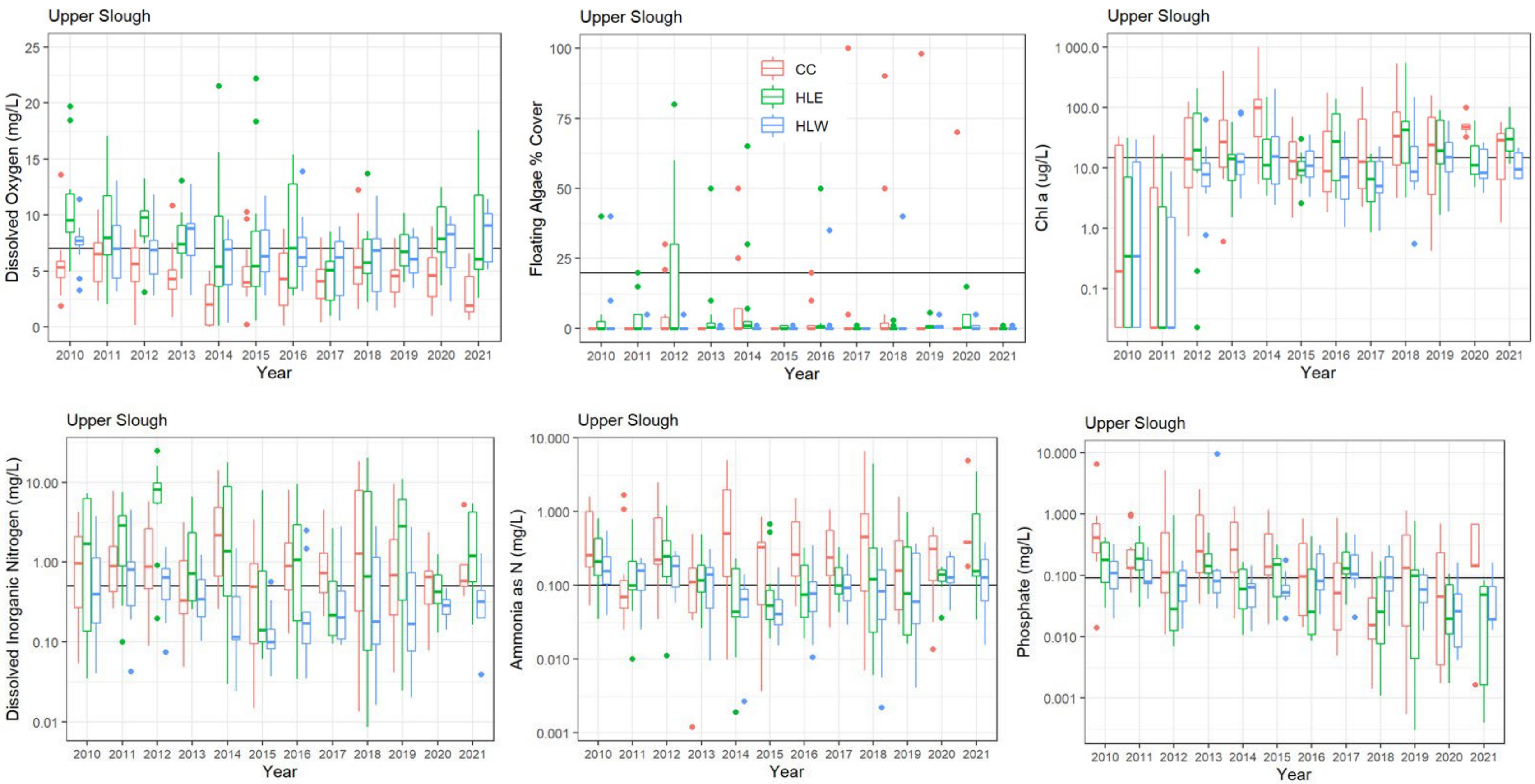


Figure A2-1. Top panel: number of years in which provisional biostimulatory targets under consideration were met in Upper Slough sites. Middle and bottom panels of box-and-whiskers plots give statistical distribution of data by year by analyte.

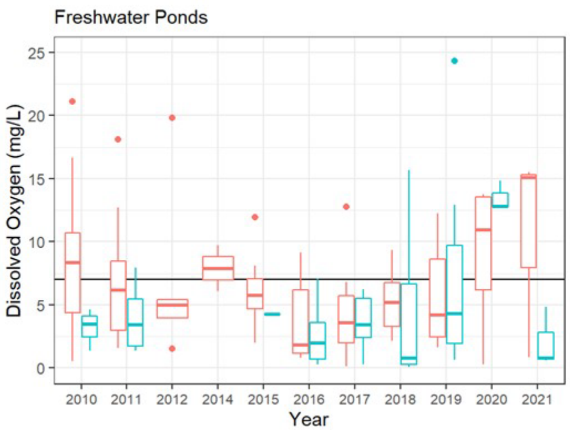
Freshwater Ponds*

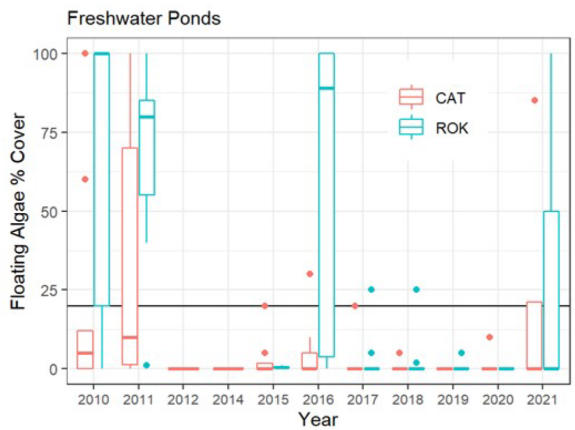
- ROK
- CAT

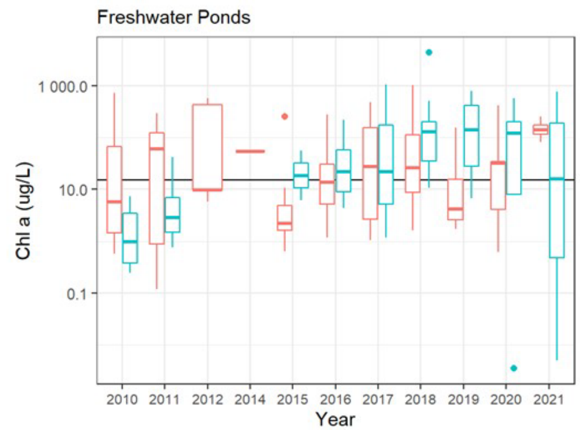
Number of Years over 12-year Record in Which Target is Not Met

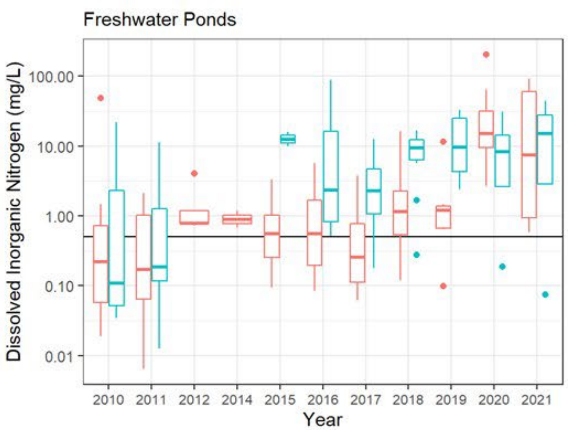
Site	Statistic	DO	FloatAlg%Cov	Chl-a	DIN	Nh4	TP
CAT	Eval. Statistic	11/11	4/11	11/11	11/11	10/11	11/11
ROK	Eval. Statistic	8/9	4/9	8/9	9/9	9/9	9/9

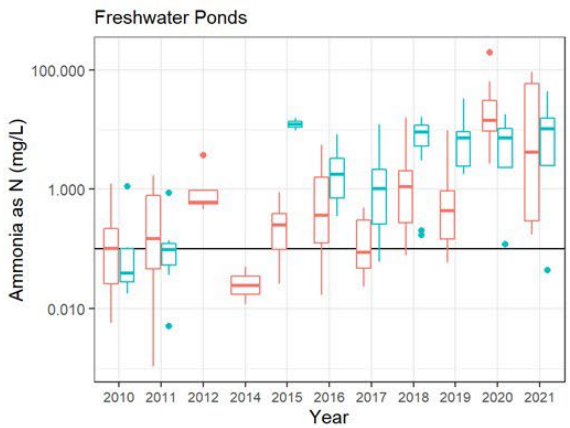
Site	Statistic	DO	FloatAlg%Cov	Chl-a	DIN	Nh4	TP
CAT	Median	7/11	0/11	6/11	8/11	9/11	6/11
ROK	Median	8/9	3/9	7/9	7/9	7/9	7/9











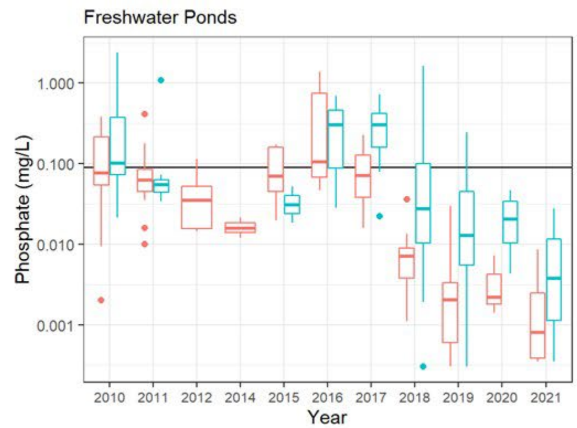


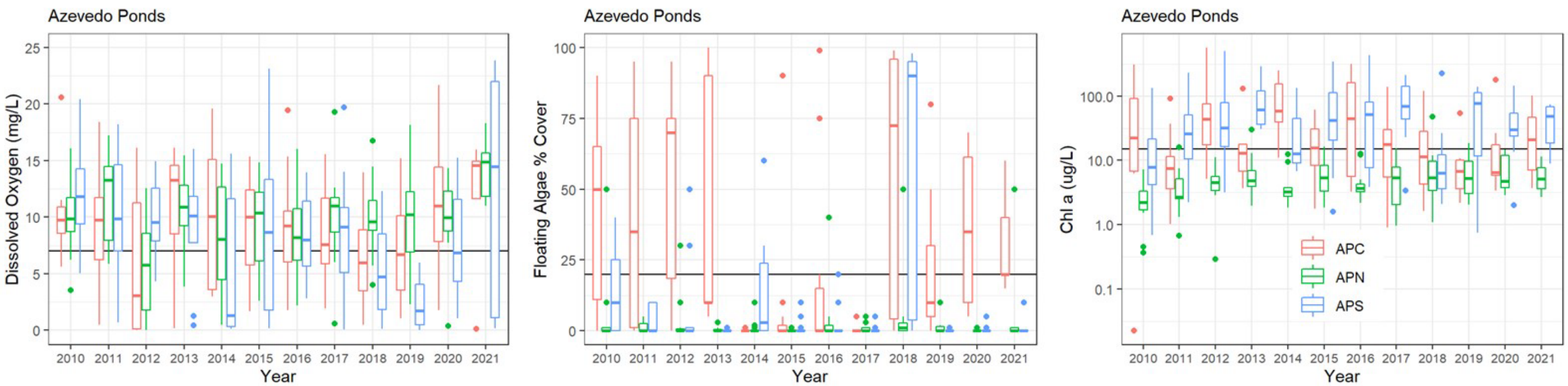
Figure A2-2. Top panel: number of years in which provisional biostimulatory targets under consideration were not met in Freshwater Pond sites. Middle and bottom panels of box-and-whiskers plots give statistical distribution of data by year by analyte.

Azevedo Ponds*
- Azevedo Pond Central (APC)
- Azevedo Pond North (APN)
- Azevedo Pond South (APS)

Number of Years over 12-year Record in Which <u>Target is Not Met</u>

Site	Statistic	DO	FloatAlg%Cov	Chl-a	DIN	Nh4	TP
APC	Eval. Statistic	12/12	10/12	11/12	4/12	10/12	5/12
APN	Eval. Statistic	10/12	5/12	0/12	5/12	11/12	11/12
APS	Eval. Statistic	12/12	4/12	12/12	7/12	8/12	9/12

Site	Statistic	DO	FloatAlg%Cov	Chl-a	DIN	Nh4	TP
APC	Median	3/12	5/12	6/12	0/12	3/12	0/12
APN	Median	1/12	0/12	0/12	0/12	3/12	0/12
APS	Median	4/12	1/12	9/12	0/12	0/12	1/12



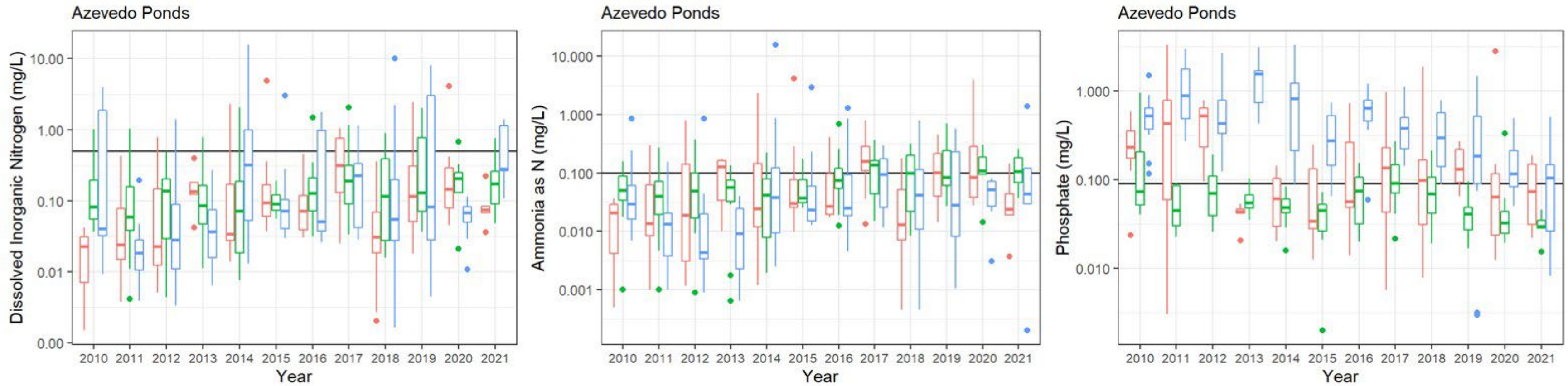


Figure A2-3. Top panel: number of years in which provisional biostimulatory targets under consideration were not met in Azevedo Pond volunteer monitoring sites. Middle and bottom panels of box-and-whiskers plots give statistical distribution of data by year by analyte.

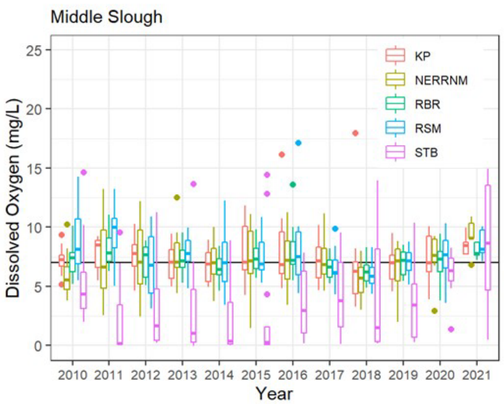
Middle Slough

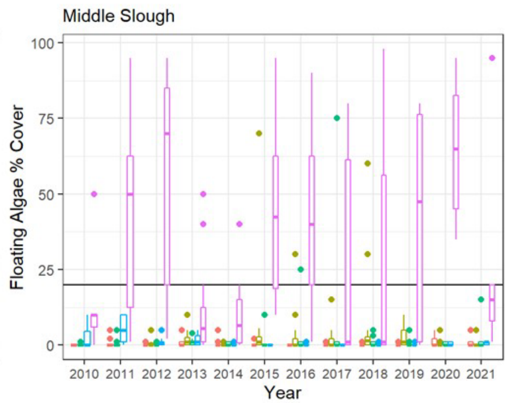
- Kirby Park (KP)
- Reserve North Marsh (NERRNM)*
- Reserve Bridge (RBR)*
- Reserve South Marsh (RSM)*
- Strawberry Road (STB)*

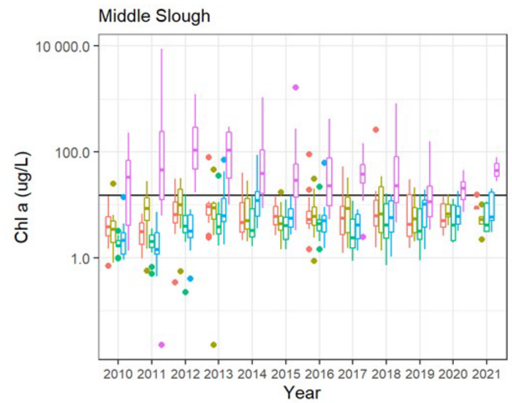
Number of Years over 12-year Record in Which Target is Not Met

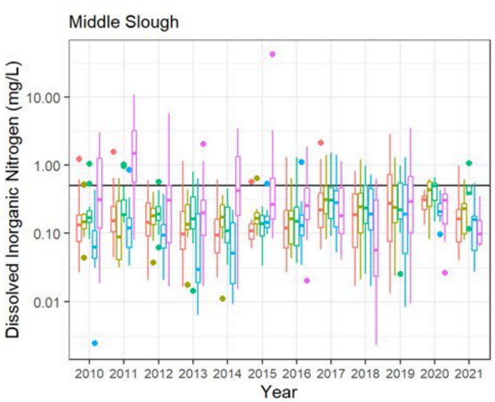
Site	Statistic	DO	FloatAlg%Cov	Chl-a	DIN	Nh4	TP
NERRNM	Eval. Statistic	11/12	3/12	7/12	5/12	9/12	12/12
RBR	Eval. Statistic	11/12	2/12	1/12	7/12	10/12	12/12
RSM	Eval. Statistic	11/12	0/12	5/12	3/12	7/12	12/12
STB	Eval. Statistic	12/12	12/12	12/12	9/12	12/12	6/12
KP	Eval. Statistic	11/12	0/12	5/12	6/12	9/12	11/12

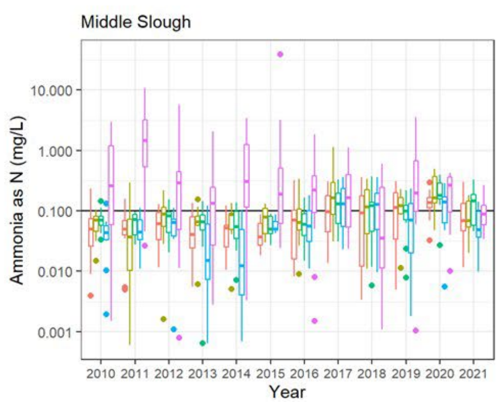
Site	Statistic	DO	FloatAlg%Cov	Chl-a	DIN	Nh4	TP
NERRNM	Median	4/12	0/12	0/12	0/12	4/12	5/12
RBR	Median	3/12	0/12	0/12	1/12	4/12	7/12
RSM	Median	5/12	0/12	0/12	0/12	3/12	1/12
STB	Median	11/12	6/12	11/12	1/12	10/12	0/12
KP	Median	3/12	0/12	0/12	0/12	2/12	2/12











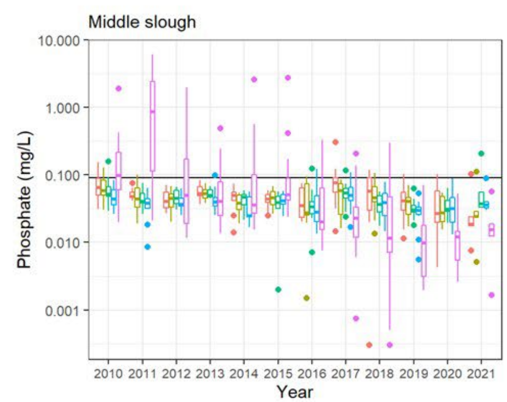


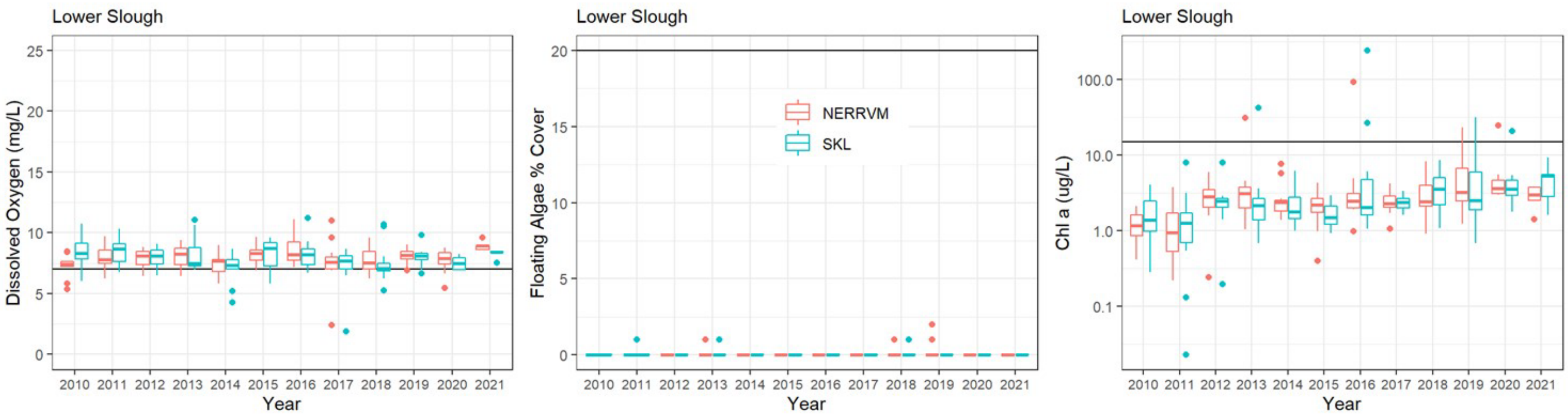
Figure A2-4. Top panel: number of years in which provisional biostimulatory targets under consideration were not met in Middle Slough volunteer monitoring sites. Middle and bottom panels of box-and-whiskers plots give statistical distribution of data by year by analyte.

Lower Slough
- Vierra Mouth (NERRVM)
- Skippers Landing (SKL)

Number of Years over 12-year Record in Which Target is Not Met

Site	Statistic	DO	FloatAlg%Cov	Chl-a	DIN	Nh4	TP
NERRVM	Eval. Statistic	8/12	0/12	0/12	6/12	6/12	12/12
SKL	Eval. Statistic	7/12	0/12	1/12	12/12	11/12	12/12

Site	Statistic	DO	FloatAlg%Cov	Chl-a	DIN	Nh4	TP
NERRVM	Median	0/12	0/12	0/12	0/12	3/12	12/12
SKL	Median	0/12	0/12	0/12	5/12	3/12	12/12



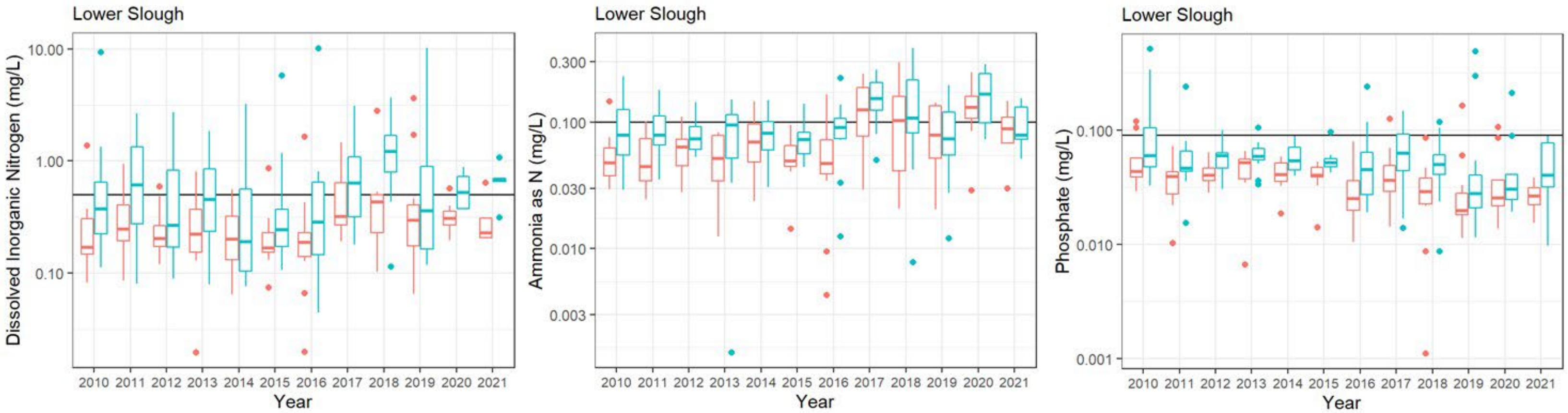


Figure A2-5. Top panel: number of years in which provisional biostimulatory targets under consideration were not met in Vierra Mouth volunteer monitoring sites. Middle and bottom panels of box-and-whiskers plots give statistical distribution of data by year by analyte.

Bennett Slough

- Bennett Slough East (BSE) & West (BSW)*
- Jetty Road (JR)
- Struve Pond (SP)*

Number of Years over 12-year Record in Which <u>Target is Not Met</u>

Site	Statistic	DO	FloatAlg%Cov	Chl-a	DIN	Nh4	TP
BSE	Eval. Statistic	11/12	7/12	12/12	3/12	7/12	9/12
JR	Eval. Statistic	12/12	1/12	0/12	12/12	12/12	12/12
SP	Eval. Statistic	8/8	5/8	5/8	7/8	4/8	8/8

Site	Statistic	DO	FloatAlg%Cov	Chl-a	DIN	Nh4	TP
BSE	Median	3/12	1/12	10/12	0/12	0/12	1/12
JR	Median	1/12	0/12	0/12	4/12	5/12	11/12
SP	Median	4/8	1/8	3/8	0/8	0/8	4/8

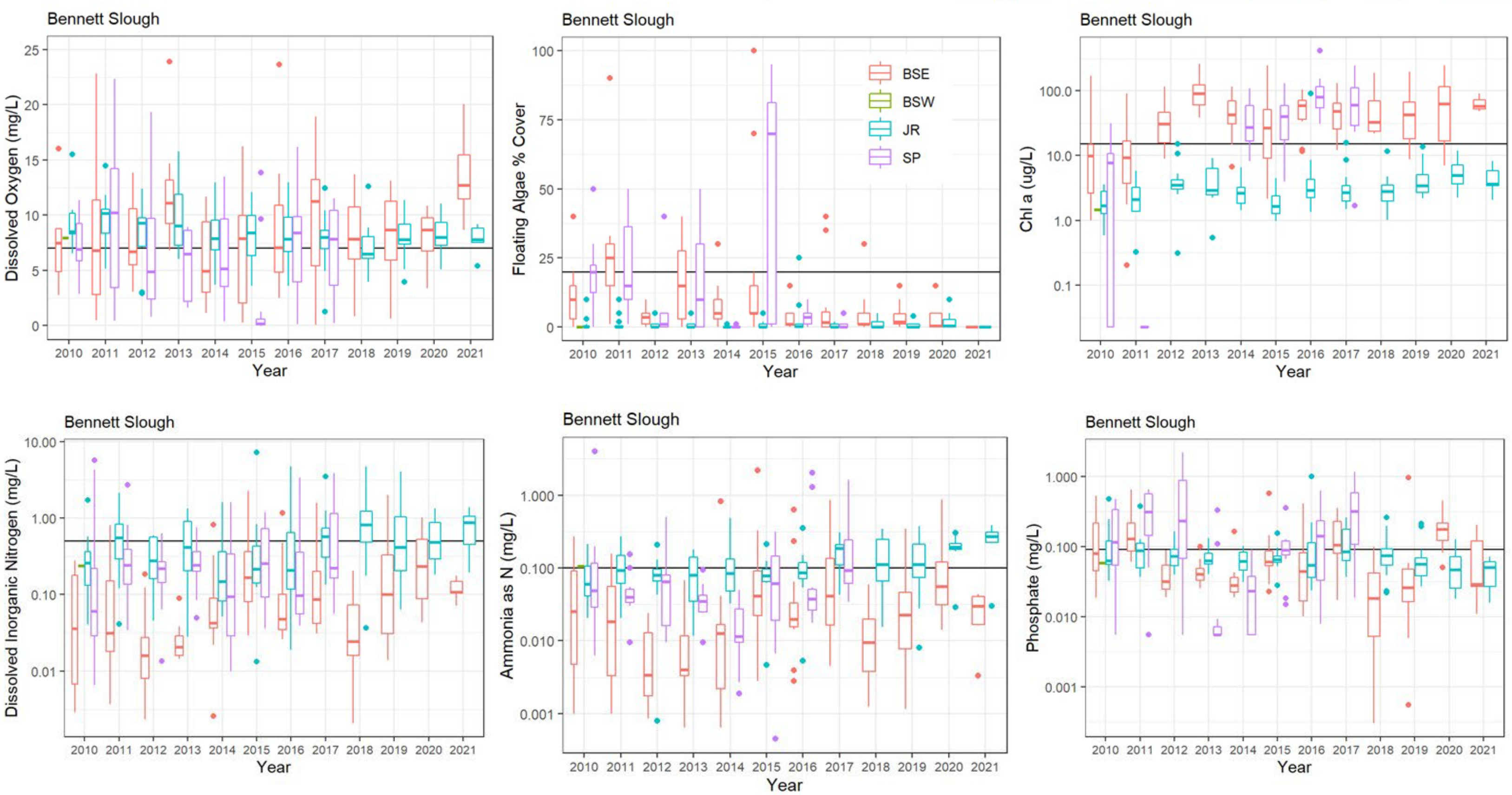


Figure A2-6. Top panel: number of years in which provisional biostimulatory targets under consideration were not met in Bennett Slough volunteer monitoring sites. Middle and bottom panels of box-and-whiskers plots give statistical distribution of data by year by analyte.

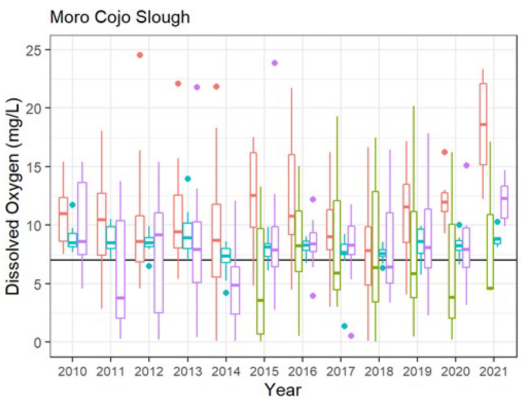
Moro Cojo Slough

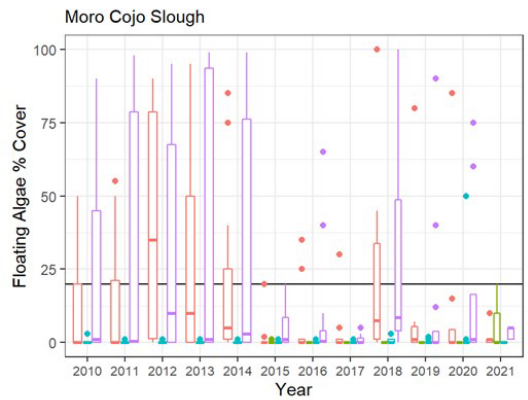
- Moss Landing R, North (MLN)
- Moss Landing R, South (MLS)
- Moro Cojo Slough (MC, MC2)*

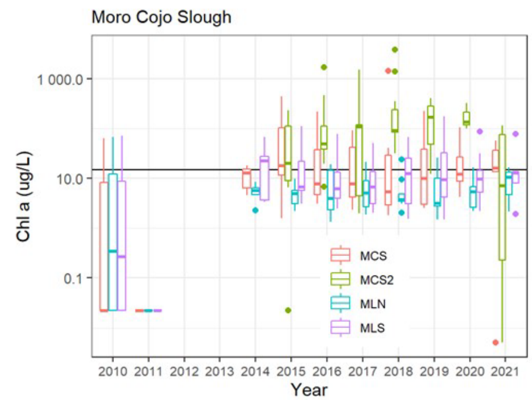
Number of Years over 12-year Record in Which Target is Not Met

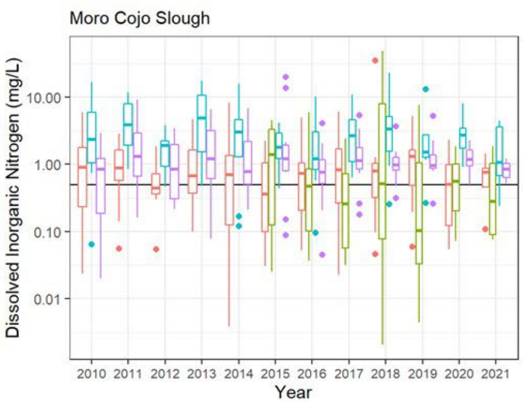
Site	Statistic	DO	FloatAlg%Cov	Chl-a	DIN	Nh4	TP
MCS	Eval. Statistic	7/12	10/12	9/12	12/12	12/12	12/12
MCS2	Eval. Statistic	7/7	0/11	7/12	7/12	7/12	6/12
MLN	Eval. Statistic	5/12	1/12	5/12	12/12	12/12	12/12
MLS	Eval. Statistic	11/12	9/12	9/12	12/12	12/12	12/12

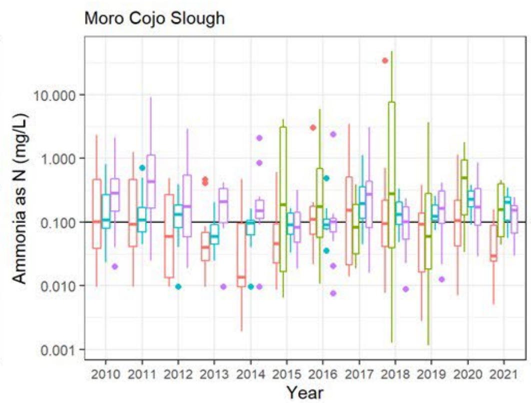
Site	Statistic	DO	FloatAlg%Cov	Chl-a	DIN	Nh4	TP
MCS	Median	0/12	1/12	2/12	10/12	4/12	12/12
MCS2	Median	5/7	0/7	7/7	3/7	5/7	3/7
MLN	Median	0/12	0/12	0/12	12/12	8/12	12/12
MLS	Median	3/12	0/12	0/12	12/12	11/12	12/12











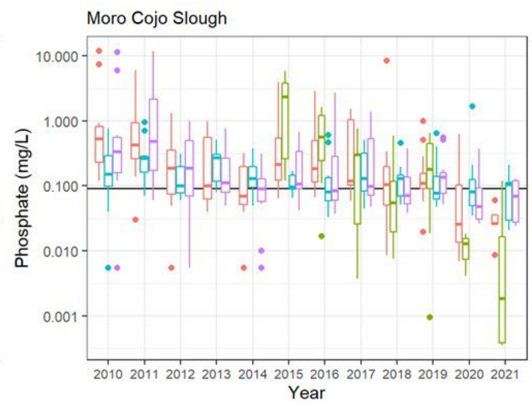


Figure A2-7. Top panel: number of years in which provisional biostimulatory targets under consideration were not met in Mojo Cojo Slough volunteer monitoring sites. Middle and bottom panels of box-and-whiskers plots give statistical distribution of data by year by analyte.

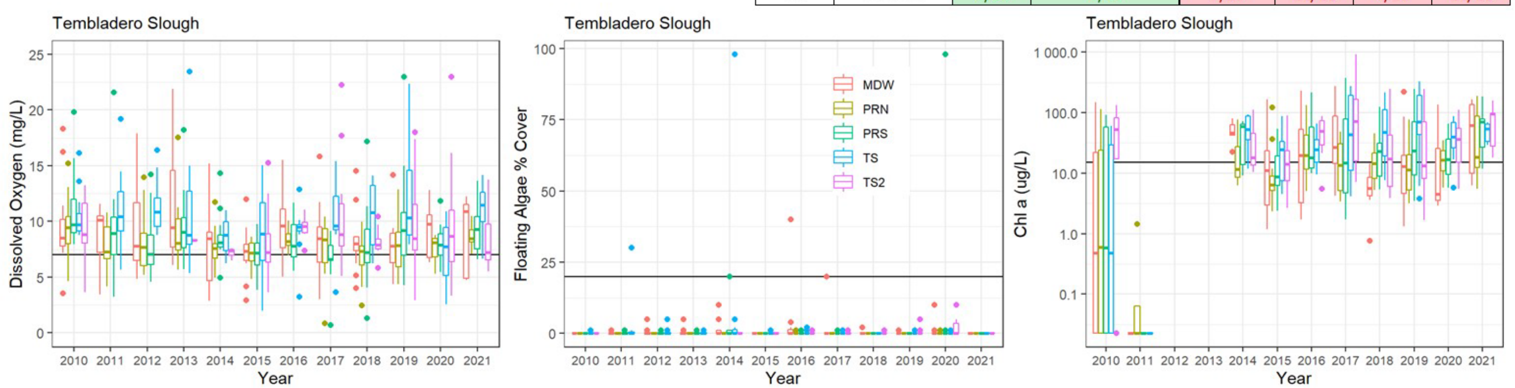
Tembladero Slough

- Monterey Dunes Way (MDW)*
- Potrero Rd North (PRN)
- Potrero Rd South (PRS)*
- Tembladero Slough (TS, TS2)*

Number of Years over 12-year Record in Which Target is Not Met

Site	Statistic	DO	FloatAlg%Cov	Chl-a	DIN	Nh4	TP
MDW	Eval. Statistic	10/12	1/12	8/12	12/12	11/12	12/12
PRN	Eval. Statistic	8/12	0/12	9/12	12/12	12/12	12/12
PRS	Eval. Statistic	9/12	1/12	9/12	12/12	12/12	12/12
TS	Eval. Statistic	4/12	2/12	9/12	12/12	12/12	12/12
TS2	Eval. Statistic	6/10	0/10	9/10	10/10	8/10	10/10

Site	Statistic	DO	FloatAlg%Cov	Chl-a	DIN	Nh4	TP
MDW	Median	0/12	0/12	3/12	12/12	4/12	12/12
PRN	Median	0/12	0/12	3/12	12/12	7/12	12/12
PRS	Median	1/12	0/12	5/12	12/12	4/12	12/12
TS	Median	0/12	0/12	7/12	12/12	7/12	12/12
TS2	Median	0/10	0/10	7/10	10/10	4/10	10/10



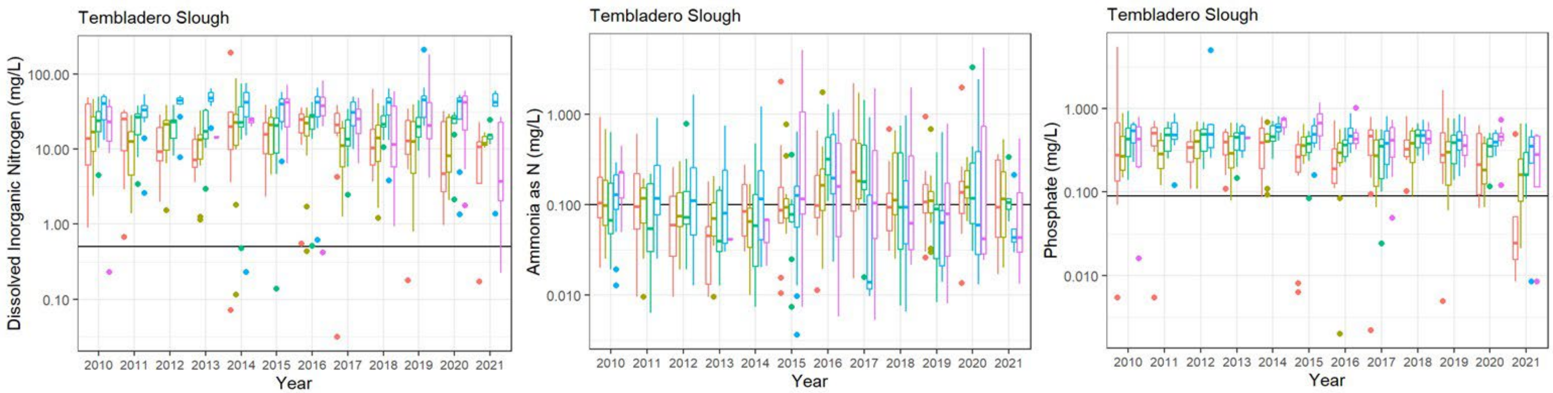


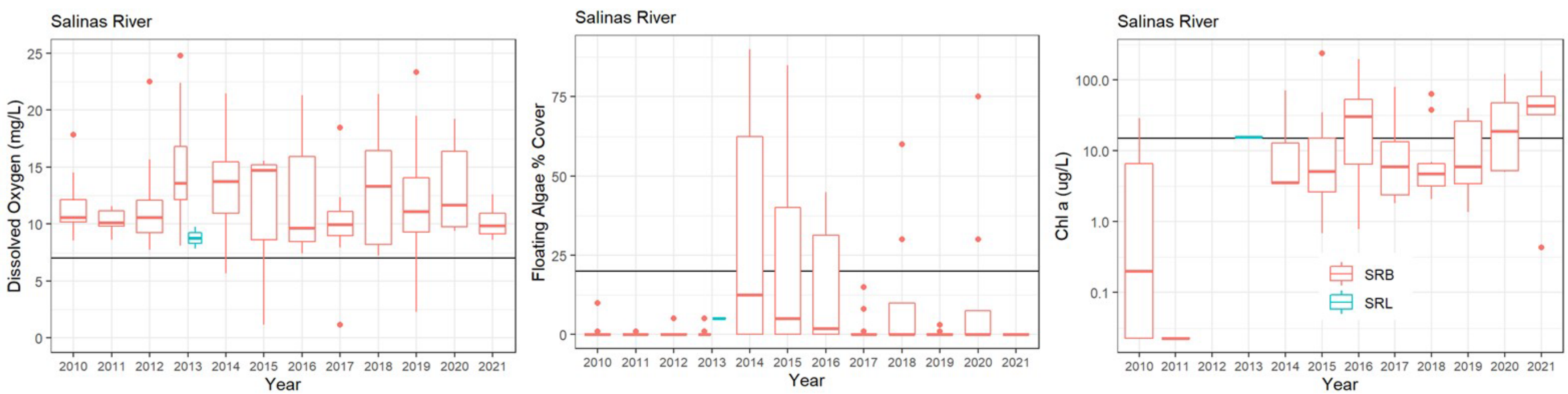
Figure A2-8. Top panel: number of years in which provisional biostimulatory targets under consideration were not met in Tembladero Slough volunteer monitoring sites. Middle and bottom panels of box-and-whiskers plots give statistical distribution of data by year by analyte.

Salinas River (SRB)

Number of Years over 12-year Record in Which <u>Target is Not Met</u>

Site	Statistic	DO	FloatAlg%Cov	Chl-a	DIN	Nh4	TP
SRB	Eval. Statistic	0/12	5/12	8/12	12/12	9/12	12/12

Site	Statistic	DO	FloatAlg%Cov	Chl-a	DIN	Nh4	TP
SRB	Median	0/12	0/12	3/12	12/12	2/12	12/12



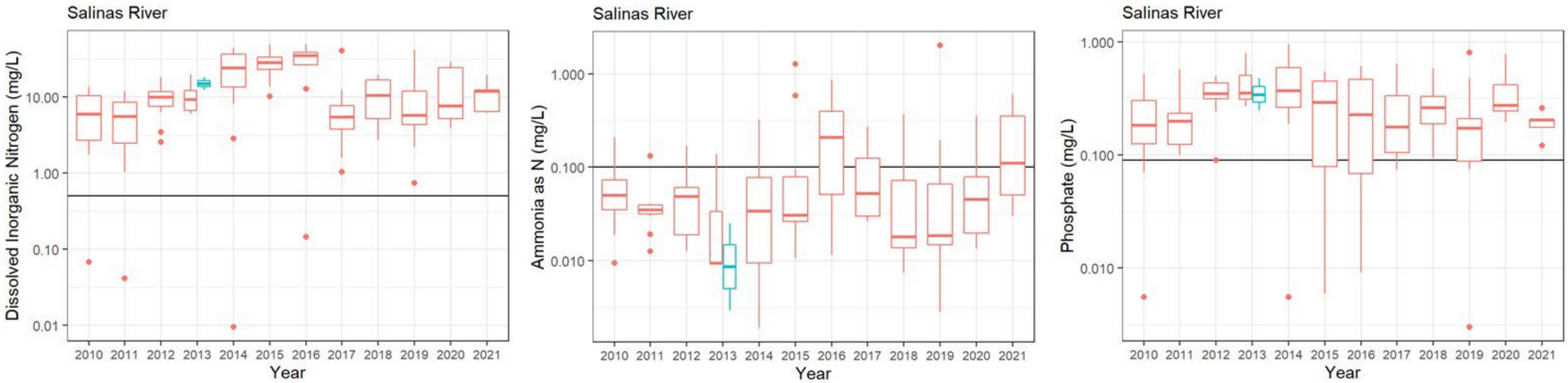


Figure A2-9. Top panel: number of years in which provisional biostimulatory targets under consideration were not met in Salinas River volunteer monitoring sites. Middle and bottom panels of box-and-whiskers plots give statistical distribution of data by year by analyte.