**Approach to Assessment, Prevention and Management of Biostimulatory Impacts to California Estuaries, Enclosed Bays, and Inland Waterbodies**

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# Executive Summary

Nitrogen (N) and phosphorus (P) overenrichment is a leading cause of eutrophication, defined as the accelerated accumulation of organic matter within an aquatic ecosystem. Other anthropogenic factors, such as hydromodification, altered water temperature, and light availability from developing landscapes can also cause or exacerbate cultural eutrophication. Scientific literature has demonstrated the shortcomings of using ambient nutrient concentration criteria alone to protect against eutrophication. The California State Water Resources Control and the Regional Water Quality Control Boards (Water Boards) have an alternative approach to regulating eutrophication through “biostimulatory” water quality objectives (WQO). As used here, “biostimulatory” refers to substances such as nutrients (i.e., nitrogen, phosphorus, and associated organic matter) or conditions, such as altered temperature, hydrology, etc. that can cause eutrophication. While all Regional Water Boards have a narrative biostimulatory objective in their Regional Basin Plans, no consistent guidance exists to interpret the narrative objective to assess eutrophication and beneficial use support in specific waterbodies, guide the prevention and management of eutrophication through permits or other Water Board actions, or to guide nutrient management actions across the state.

To address this issue, the State Water Board is proposing to adopt a statewide WQO for Biostimulatory Substances and a program to implement it, as an amendment to the Water Quality Control Plan for Inland Surface Water, Enclosed Bays and Estuaries of California. Because California is a large state and has a tremendous number and diversity of waterbodies, numeric biostimulatory guidance will be adopted in three phases. Phase I will establish the conceptual foundation and approach supporting the interpretation of the narrative biostimulatory objective, applicable to all enclosed bays, estuaries, and inland waterbodies, and establish numeric guidance for wadeable streams. Phase II and III will establish numeric guidance for lakes and estuaries, respectively. As a part of this amendment, the State Water Board intends to establish an Implementation Plan to protect biointegrity, which provides the conceptual basis and numeric goals, based on state-sponsored bioassessment protocols and indices for data interpretation, for the protection of aquatic life–related beneficial uses. These biointegrity goals can form the basis for biostimulatory numeric targets.

This document summarizes the scientific principles and assumption underpinning the development of an overarching biostimulatory policy applicable to all water body types in the State of California, and how it supports biointegrity. It has two intended uses. First, this document describes the general conceptual models, response and causal indicators of risk pathways by which nutrient pollution and eutrophication can impair beneficial uses. Second, it provides the organizing assumptions and scientific principles that can form that basis on assessment, prevention and management of biostimulatory impacts, consistent across all waterbody types (e.g., wadeable streams, lakes, and estuaries). The technical documents, specific to each waterbody class, which can form the basis for Water Board numeric guidance (e.g., wadeable streams, estuaries, lakes, etc.), will be published as appendices to this main document as they become available.

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# Introduction and Purpose

## 1.1 Introduction and Purpose of Document

Eutrophication, defined as the accelerated delivery, *in situ* production, and/or accumulation of organic matter within an aquatic ecosystem (Howarth 1988, Nixon 1995, Cloern 2001), can have far-reaching ecological impacts, from headwater streams, lakes, estuaries to the coastal ocean (Valiela et al. 1992). These impacts include hypoxia, fish-kills, lowered fishery production (Glasgow and Burkholder 2000), loss or degradation of seagrass and other aquatic beds ([Twilley 1985](#_46r0co2), [Burkholder et al. 1992](#_2lwamvv), [McGlathery 2001](#_111kx3o)), smothering of benthic macroinvertebrates, bivalves, and other organisms ([Rabalais and Harper 1992](#_3l18frh)), nuisance odors, impacts on aquatic life from increased frequency and extent of toxic harmful algal blooms, and poor water quality ([Bates et al. 1989](#_206ipza), [Bates et al. 1991](#_4k668n3), [Trainer et al. 2002](#_2zbgiuw)). There are also a range of impacts to human health (algal toxins), drinking water (algal toxins, odors and disinfection biproducts) and recreation (nuisance blooms, loss of clarity, aesthetic impairments; Nixon 1995, Paerl et al. 2011). These impacts have significant economic and social costs ([Turner et al. 1998](#_1egqt2p)). According to the U.S. Environmental Protection Agency (USEPA), eutrophication is one of the top three leading causes of impairments of the nation’s waters (US EPA 2001). Scientifically-based state water quality objectives and tools that relate these objectives to management controls are needed to protect against adverse effects from eutrophication.

California has significant nutrient pollution and eutrophication issues. Over 1,600 waterbodies have 303(d) listings for nutrient related impairments, including 10,827 miles of streams and rivers, and 484,936 acres of lakes, reservoirs, estuaries and bay, and associated wetland habitat (Table 1.1). In Southern California estuaries, macroalgal blooms and hypoxia have been observed as the majority of monitored segments (McLaughlin et al. 2014). Recreational uses have been affected by harmful algal blooms (HABs) in inland waterbodies such as Copco Reservoir, Pinto Lake, Clear Lake, Lake Elsinore (https://mywaterquality.ca.gov/habs/where/freshwater\_events.html). Fish consumption has been affected by algal toxins in the Klamath River and by episodic shellfish poisoning on the Coast. Aquatic life use impacts include episodic fish kills on the coast and the Salton Sea (Kaiser 1999), sea otter deaths in Monterey Bay (Miller et al. 2010) and sea lion deaths in Central and Southern California coast (Bargu 2011). Drinking water uses have also been affected by HABs which create odors and toxins (e.g., Lake Elsinore and Clear Lake). Ground water sources of drinking water in agricultural areas are threatened by high nitrate in the Tulare Basin and the Salinas Valley.

Table 1.1. Acres of Water Bodies Impaired by Region by Waterbody Type (California Integrated Report 2014/2016). www.waterboards.ca.gov/water\_issues/programs/tmdl/integrated2014\_2016.shtml

|  |  |  |  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- | --- | --- | --- |
| WB Type | R1 | R2 | R3 | R4 | R5 | R6 | R7 | R8 | R9 |
| Estuaries | 199 | 28 | 2497 | 408 |  |  |  | 653 | 2162 |
| Bays&Harbor |  | 8545 |  |  |  |  |  | 767 | 12 |
| Lakes&Reservoirs |  |  | 2801 | 700 | 40070 | 115335 |  | 5835 |  |
| Wetland/Tidal |  | 66339 |  | 30 |  |  |  |  | 5507 |
| Saline Lake |  |  |  |  |  |  | 233045 |  |  |
| Total Acres | **199** | **74913** | **5298** | **1138** | **40070** | **115335** | **233045** | **7256** | **7681** |

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Figure 1.1. High algal abundance in Loma Alta Slough, Oceanside (photo from RWQCB Phosphorus TMDL), Pinto Lake, Watsonville (photo from CA waterboards), Eel River (photo by Keith Bouma-Gregson), and Klamath Dam (photo source unknown).

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 riparian and channel physical habitat, water temperature, and light availability, grazing pressure, etc. (Paerl et al. 2011). Though in a risk prevent framework, scientific literature has demonstrated the shortcomings of using ambient nutrient concentrations alone to protect against eutrophication, e.g., in streams (Welch et al. 1989, Fevold 1998, Chetelat et al. 1999, Heiskary and Markus 2001, Dodds et al. 2002) and estuaries (Cloern 2001, Dettman et al. 2001, Kennison et al. 2003). In some cases, surface water nutrient concentrations alone are generally not effective for assessing eutrophication and the subsequent impact on beneficial use because ambient concentrations are not temporally and spatially representative and do not reflect the biological processing that has already occurred. In addition, biological response to nutrients (e.g., algal productivity) depends on a variety of mitigating factors such as basin morphology and substrate characteristics, tidal energy, stratification, temperature, light availability, biological community structure, and seed populations. Thus, high concentrations are not entirely predictive of eutrophication, and low concentrations do not necessarily indicate absence of eutrophication.

The California State Water Resources Control Board (State Water Board) has an alternative approach to regulating eutrophication impacts of nutrient pollution through an existing “biostimulatory” water quality narrative objective (WQO). As used here, “biostimulatory” refers to substances such as nutrients (i.e., nitrogen and phosphorus) or conditions, such as altered temperature, hydrology, etc. that can cause eutrophication (Figure 1.1). 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” (Central Coast Water Board Basin Plan 1989); similar narrative language is used throughout California Water Board Basin Plans. Fewer Regional Water Boards have narrative translators (Table 1.2). While narrative biostimulatory objective theoretically covers a wide range of environmental drivers, no consistent guidance exists to interpret this narrative objective to prevent eutrophication in specific waterbodies or to guide nutrient management actions across the state.

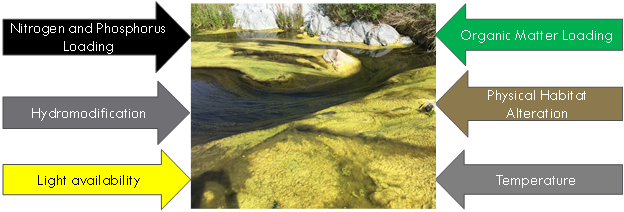


Figure 1.2. Conceptual representation of biostimulatory conditions and substances that result in eutrophication. Photo from Santa Margarita River in the San Diego Region.

Table 1.2. Biostimulatory objective language, fom each Regional Water Board Basin Plans

|  |
| --- |
| **North Coast.** 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. |
| **San Francisco.** 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. Changes in chlorophyll a and associated phytoplankton communities follow complex dynamics that are sometimes associated with a discharge of biostimulatory substances. Irregular and extreme levels of chlorophyll a or phytoplankton blooms may indicate exceedance of this objective and require investigation. |
| **Central Coast.** 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. |
| **Los Angeles.** Biostimulatory substances include excess nutrients (nitrogen and phosphorus) and other compounds that stimulate aquatic growth. In addition to being aesthetically unpleasant (causing taste, odor, or color problems), this excessive growth can also cause other water quality problems. Waters shall not contain biostimulatory substances in concentrations that promote aquatic growth to the extent that such growth causes nuisance or adversely affects beneficial uses. |
| **Central Valley**. Water shall not contain biostimulatory substances which promote aquatic growths in concentrations that cause nuisance or adversely affect beneficial uses. |
| **Lahontan.** Waters shall not contain biostimulatory substances in concentrations that promote aquatic growths to the extent that such growths cause nuisance or adversely affect the water for beneficial uses. |
| **Colorado**. 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. Nitrate and phosphate limitations will be placed on industrial discharges to New and Alamo Rivers and irrigation basins on a case-by-case basis, taking into consideration the beneficial uses of these streams. |
| **Santa Ana.** Excessive growth of algae and/or other aquatic plants can degrade water quality. Algal blooms sometimes occur naturally, but they are often the result of excess nutrients (i.e., nitrogen and phosphorous) from waste discharges or nonpoint sources. These blooms can lead to problems with tastes, odors, color, and increased turbidity and can depress the dissolved oxygen content of the water, leading to fish kills. Floating algal scum and algal mats are also an aesthetically unpleasant nuisance. Waste discharges shall not contribute to excessive algal growth in inland surface receiving waters. |
| **San Diego**. Excessive growth of algae and/or other aquatic plants can degrade water quality. Algal blooms sometimes occur naturally; however, they are often the result of waste discharges or nonpoint source pollutants. Algal blooms depress the dissolved oxygen content of water and can result in fish kills. Algal blooms can also lead to problems with taste, odors, color, and increased turbidity. Floating algal scum and algal mats are also an aesthetically unpleasant nuisance. This general condition is known as eutrophication. Concentrations of nitrogen and phosphorus, by themselves or in combination with other nutrients, shall be maintained at levels below  those which stimulate algae and emergent plant growth. Threshold total phosphorus (P) concentrations shall not exceed 0.05 milligrams per liter (mg/L) in any stream at the point where it enters any standing body of water, nor 0.025 mg/L in any standing body of water. A desired goal to prevent plant nuisance in streams and other flowing waters appears to be 0.1 mg/L total P. These values are not to be exceeded more than 10% of the time unless studies of the specific water body in question clearly show that water quality objective changes are permissible, and changes are approved by the Regional Board. Analogous threshold values have not been set for nitrogencompounds; however, natural ratios of nitrogen to phosphorus are to be determined by surveillance and monitoring and upheld. If data are lacking, a ratio of N:P = 10:1 , on a weight-to-weight basis shall be used. |

To address this issue, the State Water Board is proposing to adopt a statewide water quality objective for Biostimulatory Substances and a program to implement it as an amendment to the Water Quality Control Plan for Inland Surface Water, Enclosed Bays and Estuaries of California (ISWEBE Plan). Staff drafted a set of guiding principles for development of biostimulatory objectives (Table 1.3., SWRCB 2014) that were consistent with the 2010 workplan for development of biological objectives ([www.waterboards.ca.gov/plans\_policies/docs/biological\_objective/draft\_bio\_objs\_workplan.pdf](http://www.waterboards.ca.gov/plans_policies/docs/biological_objective/draft_bio_objs_workplan.pdf)). Because California is a large state and has a tremendous number and diversity of waterbodies, staff propose that numeric biostimulatory guidance will be adopted in three phases. Phase I will establish the consistent conceptual foundation and approach supporting the interpretation of the narrative biostimulatory objective, applicable to all inland, enclosed bays, and estuaries waterbodies, and establish numeric guidance for wadeable streams. Phase II and III will establish numeric guidance for lakes and estuaries, respectively. As a part of this amendment, the State Water Board intends to establish an Implementation Plan to protect biointegrity, which provides the conceptual basis for the protection of aquatic life–related beneficial uses and numeric guidance specifically for wadeable streams.

Table 1.3. The State Water Board established five guiding principles which frame the regulatory approach for the Biostimulatory-Biointegrity Project and provide important context for the science required to support policy options under consideration (State Water Board Biointegrity Work Plan 2010; State Water Board Nutrient Control Plan 2014; State Water Board Focus Group Outreach Document 2016; www.waterboards.ca.gov/water\_issues/programs).

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| **The State should develop objectives that address nutrient pollution and biostimulatory conditions (Figure 1.1).** Environmental variables such as hydrology, etc. can modify ecosystem response to nutrients. Anthropogenic activities that alter these environmental variables can be biostimulatory, even under low-nutrient conditions. Therefore, the Biostimulatory Substances Amendment should address both nutrient pollution and biostimulatory conditions. |
| **The State should develop narrative nutrient objectives with numeric guidance.** The Biostimulatory Substances Amendment could include: a statewide numeric objective or a statewide narrative objective (with a numeric translator), and various regulatory control options for point and nonpoint sources including a watershed management approach. The numeric objective or numeric guidance is herein referred as numeric guidance for simplicity. |
| **Numeric guidance should have a strong linkage to beneficial use**. Eutrophication results in adverse ecological responses in a water body. These ecological responses are directly linked to beneficial uses. The State Water Board is considering the option that the Biostimulatory Substances Amendment may consist of a set of numeric endpoints for these ecological response indicators and numeric targets for nutrients. |
| **The State should have numeric guidance for all water body types.** The State Water Board intends to develop numeric guidance that translates the narrative objective for all water body types. |
| **There should be statewide consistency with eco–regional flexibility**. Statewide consistency is important for equity among stakeholders. However, the State has many different ecosystems, each of which has varying biological characteristics. Therefore, a defensible statewide program must accommodate the unique qualities of each ecoregion. |

The purpose of this document is to provide the scientific approach underpinning the establishment of a narrative biostimulatory objective applicable to all water body types in the State of California. It has two intended uses. First, this document is intended to provide water quality managers and stakeholders with general conceptual models, response and causal indicators of risk pathways by which nutrient pollution and eutrophication can impair beneficial uses. Second, it provides the assumptions and scientific context for technical documents that summarize the scientific basis for numeric guidance for specific waterbody types (e.g., wadeable streams, lakes, estuaries). These waterbody–specific technical documents will be published as appendices to this main document, as they become available.

It is important to note that, in addition to nutrient enrichment and eutrophication, California’s complete suite of nutrient-related policies also addresses toxicity of certain forms of nitrogen, including toxicity of ammonia nitrogen to aquatic life and human health risks associated with elevated concentrations of nitrate and nitrite nitrogen in drinking water. The policy related to ammonia toxicity is not being updated at this time and nitrogen toxicity is not addressed further in this document. It is noted that the toxic levels for drinking water for all forms of inorganic nitrogen are generally greater than concentrations that are considered stimulatory or even saturating for plant and algal growth, and thus do not of themselves provide protection against adverse effects of nutrient enrichment.

**1.2 Document Organization**

The document is organized into 3 sections:

Section 1 provides an introduction and purpose of the document.

Section 2 provides a general conceptual model eutrophication, the linkage to biostimulatory substances and conditions, waterbody typology and examples of waterbody–specific conceptual models of eutrophication, and a statement of the problem with eutrophication across waterbody types in California.

Section 3 summarizes key tenets of the California’s approach to assess eutrophication and approaches establish the linkage to biostimulatory substances and conditions.

Appendix I includes key terms and definitions used in this document.

# 2. General Conceptual Model of Risk of Beneficial Use Impairment from Eutrophication and Linkage to Biostimulatory Substances and Conditions

Nutrient pollution and biostimulatory conditions can impair waterbodies in several ways that can ultimately result in failure to adequately support human and aquatic life beneficial uses. This can cause unaesthetic conditions, reduced recreational opportunities, or impaired quality of drinking water supply. The connections between these biostimulatory substances, ecological response and their ultimate impacts on uses are complex and can follow a variety of causal pathways, moderated by various environmental co-factors. An understanding of these pathways is important for several reasons, including (1) identification of the most sensitive pathways that are likely to result in the most stringent limits on nutrients, (2) specification of important co-factors that determine site-specific assimilative capacity for nutrients, and (3) elucidation of effective measures of effect or response indicators that can be used to measure and evaluate the connection between biostimulatory conditions and beneficial uses.

The biostimulatory assessment framework draws on concepts and terminology developed in EPA’s Ecological Risk Assessment (USEPA 1998a), Stressor Identification (USEPA 2000c), Biocriteria causal assessment (USEPA 1990, Cormier et al. 2000) and Nutrient Criteria guidance ([www.epa.gov/nutrient-policy-data/criteria-development-guidance](http://www.epa.gov/nutrient-policy-data/criteria-development-guidance)). Important factors in the success of this approach are identifying (1) the pathways by which stressors cause adverse effects and (2) informative and representative assessment endpoints that can be used to evaluate the status of beneficial uses. A key step in understanding the relationship between stressors and the management objective of protecting beneficial uses, along with the selection of appropriate indicators, is the development and evaluation of a conceptual model – a key component underpinning technical approaches to derive numeric nutrient criteria (USEPA 2001).

A conceptual model is a graphical and narrative description of the potential stressor sources within a system and the pathways by which they are likely to impact beneficial uses (Suter 1999). Conceptual models consist of two general components (USEPA 2001): (1) a description of the hypothesized pathways between human activities (sources of stressors, such as loading of nutrients from agricultural activities, wastewater discharges, or municipal stormwater), stressors (in this case, nutrients), assessment endpoints and beneficial uses; and (2) a diagram that illustrates the relationships between human activities, stressors, and direct and indirect effects on assessment endpoints. The pathways or connections between sources, stressors, and effects within the conceptual model are a series of hypotheses about risk to supporting the beneficial uses.

Conceptual models can be more informative if they capture key co-factors, ecological responses and beneficial use linkage that are specific to waterbody type. For this reason, a simple typology of aquatic habitats is provided for the purposes of organizing conceptual models and, ultimately, providing the roadmap for numeric guidance by waterbody type (SWRCB 2014).

## 2.1 Typology of Surface Waters Useful for Organizing Conceptual Models

For organizing conceptual models of risk to beneficial uses from nutrient overenrichment and eutrophication, California surface waters can be divided into five major aquatic habitat types:

* Estuaries and enclosed bays
* Lakes
* Wadeable rivers and streams
* Non-wadeable rivers and streams
* Depressional wetlands

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Figure 2.1. Different water body typologies throughout California: Merced River in Yosemite National Park (photo by Pavel Špindler), Tomales Bay in Marin County (photo by Oleg Alexandrov), the American River in Sacramento, Waddell Creek in Big Basin State Park, Lake Tahoe spanning the California-Nevada border , and the Klamath River mouth in Del Norte County. (Photo source: Wikimedia Commons)

We note that wetlands[[1]](#footnote-2) and deepwater habitat[[2]](#footnote-3) are components of each the first four categories of aquatic habitats. Depressional wetlands are a class in of themselves, but all other categories of waterbodies have wetlands as a habitat type. Below are definitions of these classes.

**Estuaries and Enclosed Bays**– **Estuaries** are the subtidal and intertidal aquatic habitats that are usually semi-enclosed by land but have surface water or groundwater exchange with the ocean, with ocean-derived water at least occasionally diluted by freshwater runoff from the land. The upstream and landward limit is where ocean-derived salts measure less than 0.5 ppt during the period of average annual low flow and water level is not influenced by the tides. The seaward limit can be (1) an imaginary line closing the mouth of a river, bay, or sound; and (2) the seaward limit of wetland emergents, shrubs, or trees when not included in (1). By the SWRCB definition (Porter-Cologne Act 1969,) “Enclosed bays” means indentations along the coast which enclose an area of oceanic water within distinct headlands or harbor works. Included are all bays where the narrowest distance between the headlands or outermost harbor works is less than 75% of the greatest dimension of the enclosed portion of the bay. They are perennially open to tidal exchange with a large ocean inlet and, as consequence, are well-flushed, often deep and subject to potentially high energy input from tides and currents.

**Lakes** – Lakes are the deepwater (greater than 2 m) and fringing wetland aquatic habitats that can be found situated in a topographic depression or dammed river channel. They are typically greater than 8 ha in size and are characterized by having an area of open water with fringing bedrock shoreline, riparian trees, shrubs, persistent emergents, emergent mosses, or lichens. Ocean-derived salinities are always less than 0.5 ppt, though some lakes can be alkaline with salinities >0.5 ppt due to underlying geology.

**Rivers and Streams** – Rivers and streams are the freshwater wetlands and deepwater habitats contained within a channel with a linear path of flow and ocean derived salinities of <0.5 ppt. Rivers and streams are further classified into wadeable and non-wadeable streams. This classification into wadeable and non-wadeable is useful because it generally points to the types of response indicators that are applicable in each type of riverine environments. We note that some response indicators typical of non-wadeable streams can be of use in wadeable rivers, and vise versa. Thus, this is meant to serve as a general guide, rather than a hard and fast rule.

**Wadeable streams, creeks and small rivers** are thus called because they are shallow enough to be sampled using methods that involve wading into the water. They typically include waters classified as 1st through 4th order in the Strahler Stream Order classification system (based on the number of tributaries upstream), but some large rivers greater than the 5th Strahler Stream order can be wadeable, particularly if they have intermittent flow.

**Non-wadeable streams and rivers** are thus called because they are typically deeper than what can be sampled by wading. They typically include waters classified as 5th order or larger in the Strahler Stream Order classification system (based on the number of tributaries upstream).

**Depressional Wetlands** – Depressional wetlands are non-tidal, shallow water aquatic habitats typically less than two meters in depth found in topographic low or dammed channel, with ocean-derived salinities less than 0.5 ppt. Depressional wetlands can dominated by open water, riparian trees, shrubs, persistent emergent vegetation, emergent mosses, or lichens.

## 2.2 Beneficial Uses

State policy for water quality control in California is directed toward achieving the highest water quality consistent with maximum benefit to the people of the state. Aquatic ecosystems and groundwater aquifers provide many different benefits to the people of the state. Beneficial uses define the resources, services, and qualities of the state’s aquatic systems that guide protection of water quality; they also serve as a basis for establishing water quality objectives. The list of designated uses provides a starting point in understanding the relationships between nutrients and use impairment. Table 2.1 provides some examples of designated beneficial uses in California estuaries, enclosed bays, and inland waters. It should be noted that waterbodies typically are assigned multiple beneficial uses.

Table 2.1 Designated Beneficial Uses in California Surface Waters

|  |  |  |
| --- | --- | --- |
| Agricultural supply  Areas of special biological significance  Aquaculture  Cold Freshwater Habitat  Commercial Fisheries  Freshwater Replenishment  Fish Migration  Groundwater Recharge | Hydropower Generation  Municipal and Domestic Supply  Navigation  Industrial Process Supply  Shellfish Harvesting  Fish Spawning  Warm Freshwater Habitat  Cultural Use | Preservation of Rare and Endangered Species  Water Contact Recreation  Non-contact Water Recreation  Estuarine Habitat  Warm Water Habitat  Marine Habitat  Wildlife Habitat  Tribal Use |

While all designated uses must be considered, some are more likely to be impaired by biostimulatory impacts than others, and water quality objectives must protect the most sensitive use of a waterbody. For example, uses for industrial service supply, hydropower generation, and industrial process supply are not likely to be the most sensitive use impacted by biostimulatory substances in a waterbody. Areas of Special Biological Significance and Preservation of Rare and Endangered Species would appear to require site-specific management plans. For this reason, the remainder of this discussion focuses on some of the other designated uses that are both commonly considered to be sensitive to eutrophication. These include but are not limited to: cold and warm freshwater habitat (COLD and WARM), estuarine and marine habitat (EST and MAR), fish Migration and spawning (MIGR and SPWN), municipal and domestic supply (MUN), water contact recreation (REC-1), non contact water recreation (REC-2), commercial, aquaculture and shellfisheries (COMM, AQUA, and SHELL).

## 2.3 Conceptual Models of Eutrophication Risk Pathways and Crosswalks to Response Indicators

A simple generic conceptual model of biostimulatory drivers, eutrophication response and waterbody ecological response to eutrophication can be described (Figure 2.1). Here, biostimulatory drivers consist of nutrients (including nitrogen, phosphorus and associated organic matter) and conditions (e.g. hydromodification, physical habitat alteration, light regime, etc.) that can accelerate eutrophication.

Most waterbodies, in their natural and minimally disturbed state, feature relatively low biomass, but high diversity of a mosaic of primary producers (Paul et al. in prep), including benthic diatoms, soft-bodied algae (e.g., macroalgae), submerged and emergent aquatic vegetation, and cyanobacteria, with a high percentage of pollution intolerant or rare taxa. In shallow waterbody types, good water clarity typically provides strong light penetration to the bottom, enhancing benthic over planktonic primary producers. In deep waters, planktonic algae thrive, again with high diversity but low overall biomass, except which is driven by season and interannual cycles in natural sources of nutrients (e.g., upwelling, seasonal runoff). In these waters, dissolved organic nutrients dominate the nutrient pool, which are conserved and recycled through the activity of heterotrophic microbes (bacteria, archaea and fungi; Findlay 2010). The diversity of habitats and food sources, with low ranges of natural stressors (nutrient forms, DO, pH, cyanotoxins), provide the basis for a complex matrix of community of primary producers, primary consumers (herbivores) and predators.

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| --- | --- |
| Photo by Carson Jeffres/UC Davis |  |
| Image result for lake mary | Image result for seagrass in tomales bay |

Figure 2.2. Examples of aquatic habitat types with mosaics of different primary producers including from top left clockwise (a) Chinook salmon admidst submerged aquatic vegetation in Big Springs Creek. Photo by Carson Jeffres/UC Davis, 2012 (b) periphyton in Antelope Creek (photo credit casalmon.org) (c) seagrass and macroalgae in Tomales Bay (National Park Service Photo) and (d) Emerald Lake showing benthic algae and phytoplankton (by Martha Sutula).

With watershed development, a series of factors change the range of natural variation in nutrients, light, temperature, and substrate (Hynes 1970, Cushing and Allan 2001, Delong and Brusven 1992, Miltner and Rankin 1998). Nutrient inputs are increased, changing the concentrations and ratios of nitrogen, phosphorus and silica, as well as micronutrients and trace elements and increasing the percentage of bioavailable nitrate, ammonium and phosphate relative to dissolved organic forms. Loss of floodplain habitat and riparian corridors increases light availability and increases temperature, a condition that is heightened as streams are armored to optimize the channel for flood control. Watershed development also reduced to changes in the flow and sedimentation, including increased reduced or increased discharge, altered retention time, decreased scour, and changes in substrate (including increased embeddedness of cobbles and pebbles with fine grained sediments). In some landscapes, turbidity increases, decreasing light penetration to the bottom. Invasive organisms can proliferate, in some cases reducing pressure from grazers that can exert a top down control on primary producer biomass. Physical habitat is degraded, by downcutting (streams), dredging, diking and filling (lakes, estuarine and enclosed bays), which modifies hydrology, mixing, residence time and stratification.

These biostimulatory substances (nutrient pollution) and biostimulatory conditions (alterations in environmental factors, e.g., hydromodification, altered light and temperature regime, changes to physical habitat) can result in changes to the ecosystem (eutrophication response) via (Figure 2.1):

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

This cascade of effects has a direct effect on the ecosystem functions, services and beneficial uses that wadeable streams provide (Table 2.1), including:

* Reduced habitat for and direct impacts to aquatic life (including WARM, COLD, WILD)
* Reduced protection of biodiversity including rare, threatened and endangered species and migratory and spawning habitat (RARE, SPWN, MIGR)
* Toxic contamination of and declines in productivity of commercial and recreational fisheries (SHELL, COMM, AQUA).
* Poor visual aesthetics and increased odors (REC2)
* Maintenance of good water quality (MUN, REC1, COMM, AQUA, SHELL)

In general, the major impairment pathways by which biostimulatory substances can impact uses constitute a suite of risk hypotheses. Different simple conceptual models of ecological response to eutrophication can be described, depending on whether there is a need to emphasize stressor (e.g., nutrient, hydromodification, altered light regime) or pathway of impairment, or linkage to beneficial use. Given the complexity of the conceptual models, there are many individual pathways or risk hypotheses to consider. The conceptual models may be reduced to a table showing risk pathways that are most likely to impair the use, as shown in Table 2.2 (top row). The conceptual models and sensitive risk pathways, described in more details in Section 2.3.1–2.3.3, form the basis for selection of appropriate response indicators and their corresponding assessment endpoints. They also form the basis for key assumptions in the linkage of the response endpoints to biostimulatory targets (e.g., nutrient inputs).

A generic set of common response indicators, linked to pathways identified in Table 2.2, are summarized in Table 2.3. We note that the list in Table 2.3 is not comprehensive; other indicators of eutrophication (e.g. respiration rate, denitrification rates, etc.), used in peer-reviewed special studies or research, can also be the basis for diagnosis of eutrophication.

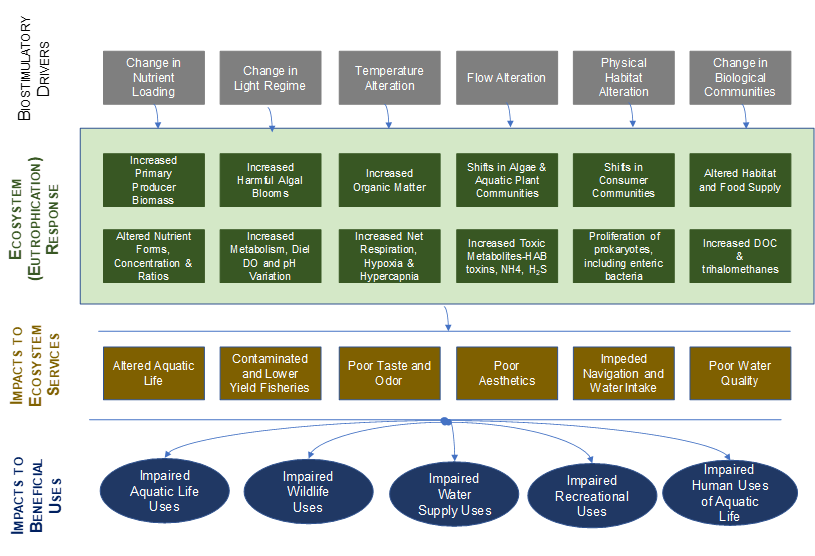


Figure 2.3. Example of simplified conceptual model for pathways in which nutrients and other biostimulatory conditions result in adverse ecosystem responses, which impact ecosystem services and impair beneficial uses in water bodies*.*

Table 2.2. Most Important Risk Pathways Associated with Impairment of Sensitive Uses by Nutrient Pollution and Eutrophication. Aquatic life-related uses, ALU, are grouped to include: EST, MAR, COLD, WARM, MIGR, RARE, and SPWN beneficial uses. Birds, amphibian and terrestrial wildlife are represented under WILD, MIGR, and RARE. Poor water quality is linked to human or aquatic/wildlife uses.

|  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- |
| **Use** | **Altered Aquatic Life** | **Contaminated and Lower Yield Fisheries** | **Poor Taste and Odor** | **Poor Aesthetics** | **Impeded Navigation and Water Intake** | **Poor Water Quality** |
| ALU | **X** |  |  |  |  | **X** |
| MIGR/WILD/RARE | **X** | **X** |  |  |  | **X** |
| COMM/AQUA/SHELL | **X** | **X** | **X** |  |  | **X** |
| TRIB/CUL | **X** | **X** | **X** | **X** | **X** | **X** |
| MUN |  |  | **X** |  | **X** | **X** |
| NAV/IND |  |  |  |  | **X** |  |
| REC-1 |  |  | **X** | **X** |  | **X** |
| REC-2 | **X** | **X** | **X** | **X** |  | **X** |

Note: This table attempts to highlight the *major* stressor-response factors associated with a specific beneficial use. Additional stressor-response relationships may also affect use support but are judged to be less likely as a primary cause of impairment of that use.

Table 2.3. Linkage of generic indicator groups to pathways of increased risk of beneficial use impairment. Precise metric used to measure response indicator may vary by waterbody type.

|  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- |
| **Response Indicator** | **Altered Aquatic Life** | **Contaminated or Low Yield Fisheries** | **Poor Taste and Odor** | **Poor Aesthetics** | **Impeded Navigation and Water Intake** | **Poor Water Quality** |
| DO | **X** | **X** | **X** |  |  | **X** |
| pH, carbonate saturation state | **X** | **X** |  |  |  | **X** |
| Secchi Depth, Turbidity, TSS, light attenuation | **X** |  |  | **X** |  | **X** |
| Benthic and/or Planktonic Algal Biomass (e,g, chl-a) | **X** | **X** | **X** | **X** | **X** | **X** |
| Benthic or floating algal percent cover | **X** | **X** | **X** | **X** | **X** | **X** |
| Benthic or planktonic ash free dry mass, particulate organic C, N and/or P | **X** | **X** | **X** | **X** | **X** | **X** |
| Planktonic or benthic algal community composition | **X** | **X** | **X** |  |  |  |
| Planktonic or benthic macroinvertebrate community composition | **X** | **X** |  |  |  |  |
| Aquatic macrophytes Diversity, biomass, shoot height, density and percent cover, epiphyte load | **X** | **X** |  | **X** | **X** |  |
| Harmful algal species abundance and toxin concentrations | **X** | **X** | **X** | **X** |  | **X** |
| Toxic nutrients or redox products (e.g. nitrate, phosphate, ammonia or sulfide) | **X** | **X** | **X** |  |  | **X** |
| Heterotrophic bacteria biomass or abundance | **X** |  | **X** |  |  | **X** |
| Increased DOC and trihalomethanes |  |  |  |  |  | **X** |

### 2.3.1 Changes in Aquatic Primary Producer (APP) Community Structure

As a waterbody becomes increasing eutrophic, predictable changes occur with respect the types and relative abundance of the primary producer communities, as depicted in Figure 2.2. Waterbodies in a “minimally disturbed” condition are typically dominated by a high diversity of primary producers that are tolerant of low nutrient conditions, including benthic and planktonic microalgae, aquatic vascular vegetation (e.g., seagrasses), and soft-bodied algae (e.g., macroalgae). Microalgae are complex communities comprised of multiple taxonomic groups including unicellular benthic and pelagic diatoms (phylum Bacillariophyta), filamentous cyanobacteria (Cyanophyta), chlorophytes (Chlorophyta), dinoflagellates (Dinophyta), euglenoids (Euglenophyta), and cryptophytes (Cryptophyta) as well as other photosynthetic bacteria (MacIntyre 1996). With increased nutrient availability, altered hydrology, temperature, light availability, and/or physical habitat disturbance (e.g., bottom substrate), eutrophication can begin to occur, resulting in marked changes to primary producer biomass and community structure. The biomass of nutrient tolerant epiphytic micro-, macroalgae as well as opportunistic macroalgae increases; in deep or turbid open water habitats, phytoplankton biomass increases. High biomass opportunistic species can outcompete both microalgae and meadow-forming benthic submerged aquatic vegetation. Eutrophication-tolerant benthic and planktonic harmful algal bloom species that can produce toxins harmful to aquatic life and human health can begin to proliferate (Fong et al. 1993 Valiela et al. 1997, Viaroli et al. 2008). In the extreme end of the eutrophication gradient, benthic and planktonic macroalgae, cyanobacteria, and picoplankton blooms dominate at extremely high biomass, causing dystrophy. These predictable changes along a gradient of increasing eutrophication provide the basis for selecting one or more primary producers as response indicators (Table 2.3). The precise indicators that will be relevant are dependent on the habitat type and waterbody class, as well as site–specific factors (e.g., light penetration, mixing) that modulate which primary producer dominates.

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Figure 2.4 Examples of four major primary producer groups found in tidal flats, shallow and deepwater habitat types in esuaries: macroalgae on tidal flats (top left) and floadting macroalgae in a closed lagoon (top right), seagrass (second panel, left), and Ruppia sp., a type of brackish water SAV (second panel, right), microphytobenthos (third panel left and right) and phytoplankton (bottom panel left and right). (Figure and caption from Sutula 2011).

Changes in algal community composition can provide a key indicator of impacts to aquatic life (Biggs and Smith 2002, Stevenson et al. 2006, Pan et al. 1996, 1999, Theroux et al. in prep). The State of California is developing an algal stream condition index (ASCI) that assesses overall health of the wadeable streams based on changes to algal community composition, relative to a population of reference streams (Theroux et al. in prep). This index applies standard mathematical formulas to data on the number and diversity of algal species (diatoms and soft-bodied algae) to find a score that rates the disturbance of the community. Currently, no such algal index exists for California lakes and estuaries but may be the focus of future state-sponsored research. A key component of such research is the development of metabarcoding approaches to taxonomic identification of algal species across all waterbody types, since the impediments exist to routine morphological taxonomy of algae (Theroux et al. 2018).

|  |  |
| --- | --- |
| 1. Wadeable habitats     Eutrophication Gradient | 1. Shallow aquatic beds     Eutrophication Gradient |
| 1. Deepwater and turbid habitats     Eutrophication Gradient | Figure 2.5. Conceptual model of relationship between biostimulatory conditions and relative dominance of primary producers in California waterbodies by major habitat type: shallow water wadeable habitats (e.g. streams, intertidal or shallow water flats (a), submerged aquatic vegetation (b), and deepwater or turbid habitats (c). \* Depends on water residence time; + Mediated by herbivory |

### 2.3.2 Changes in Physical Habitat, Water and Benthic Chemistry

As a waterbody becomes increasing eutrophic, legacy organic matter loading, organic matter deposited from upstream sources, as well as elevated live and dead aquatic primary producer (APP) biomass produced *in situ* from available nutrients provide an elevated supply of labile organic matter, setting off a cascade of altered biogeochemical cycling in the sediments and surface waters and impacts to physical habitats. These effects include, but are not limited to:

1) change in physical habitat through altered water clarity (Dennison et al., 1993), dampening of velocity, reducing reoxygenation at the surface and causing anoxic conditions at depth (Dodds and Biggs 2002) or changes in physical habitat from increased organic matter sedimentation or legacy organic matter that fundamentally alters benthic habitat for invertebrates and fish (Welch et al. 1989, Chessman et al. 1992, Hawkins et al 1982),

2) increased photosynthesis and respiration of live biomass and increased respiration of dead organic matter in the sediments and surface waters causes increased extent, frequency and duration of low DO and pH and/or carbonate saturation state waters, as well as large diurnal swings in DO and pH (Gray et al. 2002, Cloern 2001, Meyer-Reil and Koster 2000, Harper 1992; Mallin et al. 2006, Dodds 2007),

3) increased concentrations of water column sediment pore water ammonium, sulfide, increasing the potential for toxicity to benthic organisms (Figure 2.6, D'Avanzo and Kremer 1994, Nixon 1995, Diaz 2001, Howarth et al. 2002).

4) The efficiency of nitrogen and carbon cycling decreases, fueling increased organic matter accumulation and retention of nitrogen within the waterbody, both in the water column as well as in the sediments (Pearson and Rosenberg 1978, Sutula et al. 2006, Middelburg and Levin 2009).

5) Increased exposure to benthic and planktonic harmful algal bloom toxins, which poses a significant risk to humans and their pets and livestock, aquatic life and wildlife (Anderson-Abbs et al. 2006). Toxins can be freely dissolved (released from cells), in a particulate form (sometimes forming surface scum), or shellfish or fish tissue, or aerosolized with spray from boats, etc. These toxins form multiple exposure pathways with which to impact human, aquatic and wildlife related uses. These studies have shown toxins to have far reaching effects downstream or far afield of their point of origin (Miller et al., 2010; Kudela, 2011).

6) Increased dissolved organic carbon (DOC) and exposure totrihalomethanes (THMs), which are byproducts of drinking water treatment that result from the chlorination or bromination of certain DOC compounds. The DOC content of natural waters can be increased by algal production; however, in most cases, the total DOC supply is dominated by loading of organic compounds from the watershed. THMs include several known and suspected carcinogens, creating concern for drinking water safety. Algal metabolites and decomposition products present in raw water are candidates for THM production (USEPA 2000b; Graham et al., 1998; Plummer and Edzwald 2001). Higher levels of DOC also increase the amount and costs of disinfectants required to achieve disinfection goals.



Figure 2.6. An illustration of the Pearson-Rosenberg (1978) conceptual model depicting changes in macrobenthic community structure with increasing eutrophication and organic matter accumulation in the sediment. For discussion purposes, the model has been subdivided to highlight four primary condition categories associated with such increases: A – Non-eutrophic, B – Intermediate Eutrophication; C –Severe Eutrophication; and D - Anoxic bottom water and azoic sediments. (Figure and caption from Sutula 2011)

Table 2.4. Examples of indicators of altered water column and sediment chemistry applicable to assessment of eutrophication.

|  |  |
| --- | --- |
| Indicator Group | Indicator or Metric |
| Wate Column or Sediment Nutrients | Total Nitrogen and Phosphorus |
| Dissolved organic and Inorganic Nitrogen (Ammonium, nitrate+nitrite) |
| Urea |
| Dissolved organic phosphorus and phosphate |
| Water Clarity | Secchi Depth |
| Kd (Light extinction) |
| Turbidity |
| Dissolved Oxygen, pH and carbonate saturation state | Dissolved Oxygen concentration, pH, carbonate saturation state |
| DO or pH diel variability |
| Water column biological or chemical oxygen demand |
| Sediment oxygen demand |
| Benthic or Pelagic Primary production, respiration and metabolism | Production: respiration ratio |
| Net primary production |
| TCO2 flux |
| Organic Matter Accumulation and Sediment Redox Status | Sediment or benthic %OC, %N, and %P |
| Sediment or benthic C:N: P ratio |
| Sediment or benthic TOC:TS and degree of pyritization |
| Nitrogen Cycling | Denitrification rate |
| Harmful Algal Bloom Toxins | Benthic or planktonic toxin concentrations |
| Tissue toxin concentrations |
| Solid phase absorption and tracking (SPATT) for dissolved toxins |
| Dissolved organic carbon | Dissolved organic carbon |
| Trihalomethanes |

### 2.3.3 Altered Community Composition of Secondary and Tertiary Consumers

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

Because invertebrates that live in or on sediments are exposed to environmental stressors on an ongoing basis, the benthic community composition often provides a good indicator of impacts to aquatic life. The State of California has developed benthic macroinvertebrate indices of condition that are indicative of aquatic life related beneficial use support. This includes the California Stream Condition Index (CSCI; Mazor et al. 2016) and, for bays and estuaries, the benthic response index (BRI; Ranashinghe et al. 2012) or ambient macroinvertebrate response index (AMBI; Gillett et al. 2015) for salinities of 18 ppt or greater. These indices provide a simple means for communicating complex ecological data to environmental managers. While secondary and tertiary consumers are closely linked to ecosystem services and beneficial uses (Figure 2.1), use of these organisms as diagnostic indicators for eutrophication is caveated because organism and population measures of health are impacted by a variety of different stressors in a complex environment which is not easy to model.

### 2.3.4 Impacts to Food Water Resources and Recreational Uses

Eutrophication is responsible for direct impacts to human water resources and recreational uses. Several distinct pathways exist through which these impacts can occur (Nixon 1995, Paerl et al. 2011). First, human uses are impacted by poor aesthetics that result from visual scums, high biomass blooms, and odor that can result from decaying algal or bacterial biomass. Second, recreational and commercial uses occur from the reduced abundance and biodiversity of aquatic and terrestrial wildlife (e.g., salmonids, crabs, bivalves, et al. sportsfish) and from harmful algal blooms. Third, primary contact recreation is directly impacted through the exposure of harmful algal blooms toxins from swimming, fishing, boating (including via aerosolized pathways). Fourth, reduced quality of drinking water sources occurs from high HAB toxins, DOC, trihalomethanes, which poses significant costs to water purveyors to remove to assure safe drinking water for the public. Finally, eutrophication can impede navigation and cause problems for municipal or industrial water intakes because of high biomass algal blooms and aquatic vegetation.

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Figure 2.7. Impacts of eutrophication on beneficial uses of streams, rivers and estuaries: contamination of drinking water, toxin-producing blooms that are unsafe for humans and have caused dog and sea otter deaths, non-toxic nuisance blooms that can lead to hypoxia and fish kills, as well as toxic and non-toxic blooms that form algae scums and release unpleasant odors. (Photo sources: top left from epa.gov, top middle from dogtrekker.com, top right SCCWRP, middle left from USC Annenberg Center for Health Journalism, middle middle: Harry Morse, bottom left SCCWRP, bottom middle:USFWS , bottom right: Michiganfarmer.com).

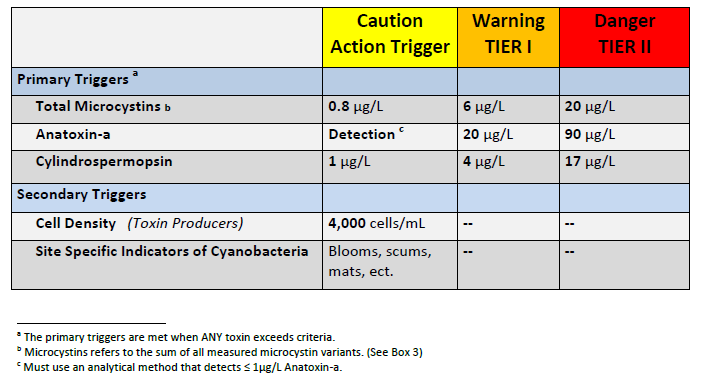
## 2.4 Evidence of Eutrophication Impacts to California Water bodies

Eutrophication symptoms are ubiquitous across the California landscape, Regional Boards and waterbody types, from ridgeline to reefs, providing a strong justification for coordinated state policy to prevent and mitigate eutrophication, particularly as climate change impacts on temperature and hydrology are likely to continue to exacerbate these conditions in the future (Paerl et al. 2011).

### 2.4.1 Toxic Harmful Algal Bloom.

HABs can have negative impacts on the environment and raise serious concerns for drinking water sources, recreational use, pets, wildlife, and livestock. In recent years, harmful algae blooms from cyanobacteria (CyanoHABs), and associated cyanotoxins, have gained national attention due to increases in the frequency and severity of blooms, and their impacts on drinking water sources, on bivalves and recreational fisheries, causing concerns of bioaccumulation to aquatic and human food chains. There are several well-documented problem areas in California that have been monitored through either assessment studies or water quality monitoring programs. Some of the areas with recurrent toxic cyanobacteria blooms include the Klamath River watershed (including Copco and Iron Gate Reservoirs), Clear Lake, Pinto Lake, lower Sacramento and San Joaquin Rivers and Delta, Lake Elsinore and several East San Francisco Bay Area lakes. The Klamath River and Pinto Lake have been placed on the State’s Clean Water Act Section 303(d) list due to impairment caused by cyanotoxins. In the Klamath River Watershed, high cell densities and toxins (microcystin) of Microcystis aeruginosa have been observed since 2004 (Kann, 2004; Jacoby and Kann 2007; Fetcho 2007; Moisander et al. 2009). Microcystis aeruginosa cells and microcystin have been documented in mussels (bivalves) and fish tissue collected from the river (Kann 2008). Concern with impact to recreational uses has caused the State to agree on voluntary listing guidance for cyanoHABs (Table 2.5, CCHAB 2016) and to undertake an annual pre-4th of July and Labor Day recreational lakes assessment to assess safe-to-swim (<https://mywaterquality.ca.gov/habs/data_viewer/>).

Table 2.5. CCHAB trigger levels for cyanotoxin impacts to human health (from MyWaterQualityPortal.ca.gov).



CyanoHABs are not just associated with lakes or other lentic waterbodies (Magrann et al. 2015). More recently, cyanobacteria and cyanotoxin data have been collected opportunistically through several programs. These data indicate that cyanobacteria are prevalent throughout California in all types of waterbodies sampled (lakes, rivers, streams, wetlands, estuaries and coastal). Recent statewide assessment surveys of wadeable streams found that benthic cyanobacteria and related cyanotoxins are widely present, suggesting that these streams can be a significant cyanotoxin source to receiving waters (Fetscher et al. 2015). In statewide studies conducted from 2007 through 2013, samples were collected from more than 1,200 wadeable stream reaches. Analysis revealed a high occurrence of potentially toxic benthic cyanobacteria taxa, and detection of microcystins in one-third of reaches and 34% of stream kilometers. Detected toxins included lyngbyatoxin, saxitoxins, anatoxin-a, and microcystins (Fetscher et al. 2015). Additionally, the State Water Quality Control Board’s Surface Water Ambient Monitoring Program (SWAMP) has measured cyanotoxins in sediment at the bottom of major watersheds at most sampling sites.

Freshwater HABs and associated toxins can impact estuarine and marine uses. For example, the mortality of over 30 endangered California Sea Otters (*Enhydra lutris*) in Elkhorn Slough and Monterey Bay was determined to be due to microcystin intoxication, with ingestion of contaminated marine bivalves as the primary mechanism (Miller et al. 2010). Pinto Lake, a eutrophic lake that experiences frequent cyanobacteria blooms and drains to Elkhorn Slough and Monterey Bay via the Pajaro River, was identified as the primary source of the toxin (Figure 2.8; Miller et al. 2010; Kudela 2011). Microcystin-laden water from the Pajaro River, and other tributaries to the Bay, flow to the coast where the toxin is biomagnified by bivalves, and ultimately consumed by otters (Miller et al. 2010). Microcystins have been shown to bioaccumulate in commercially and recreationally-harvested invertebrates such as Pacific oysters (*Crassostrea gigas*) and mussels (*Mytilus edulis*) (Miller et al. 2010). Microcystins were shown to be present and persistent in most of the coastal watersheds that flow to the Monterey Bay National Marine Sanctuary over a 3-year time-series survey (Gibble and Kudela 2014); nutrient loading was determined to be a significant predictor of microcystin concentrations in the watersheds (Gibble and Kudela 2014). These studies have shown cyanotoxins to have far reaching effects downstream of their origin and have promoted cyanotoxins from predominantly a freshwater issue to a land-sea interface problem.



Figure 2.8 Cyanobacterial bloom in Pinto Lake, California. Photo credit: Kudela Lab, UCSC

Anthropogenic nutrient pollution can also exacerbate marine HABs including frequent blooms of *Pseudo-nitzschia* (PN). Previously considered rare in phytoplankton communities, toxic and/or extremely high-biomass toxigenic harmful algal diatom genus *Pseudo-nitzschia* (PN) *spp.* blooms are now recognized as high impact events reported to frequently affect aquaculture operations, recreational zones, marine mammal and bird populations, and even open ocean ecosystems (Anderson et al. 2012). Domoic acid (DA) poisoning in marine mammal/bird populations and the threat of Amnesic Shellfish Poisoning (ASP) in humans is now considered to be the leading HAB and conservation issue for much of the U.S. West Coast (Kudela et al. 2008; Lewitus et al. 2012; Trainer et al. 2012) since these events have the potential to be devastating to aquatic life via bioaccumulation in the food web (Fritz et al. 1992; Lefebvre et al. 1999; Scholin et al. 2000). The summer of 2015 saw a record PN bloom, that stretched from Alaska to the Central Coast of California and lasted several months in duration, causing the closure of economically important fisheries such as razor clams, Dungeness crab, sardines, anchovies, which cost hundreds of millions of dollars; it also resulted in record levels of marine birds and mammal stranding along the coast (McCabe et al. in 2016). Several studies from the Monterey Bay and Southern California Bight (SCB) suggest that the the toxigenic diatom genus, PN, is partially associated with periods of increased river runoff (Anderson et al. 2008a; Bates et al. 1998b; Dortch et al. 1997; Fisher et al.2003; Pan 2001; Trainer et al. 2000a; Van Dolah et al. 2003; Wells et al. 2005. Laboratory manipulations of toxigenic PN species in culture have shown that production of its deadly neurotoxin, DA, is in turn often a function of phosphate limitation at different phases of PN growth (Bates et al. 1991; Fehling et al. 2004b; Pan et al. 1996c; Pan et al. 1996d). At the same time, these changes in nutrient stoichiometry are likely coupled to increased loading of urea and other terrestrial nutrients, which further enhance toxin production by at least some species of PN (Howard et al. 2007; Kudela et al. 2008b).

**Non-Toxic Blooms.** Non-toxic harmful algal blooms can be problematic for beneficial uses (Figure 2.9). Some of these species are non-native, invasive species. For example, the freshwater golden haptophyte alga, Prymnesium parvum, which has caused fish kills recently in Southern California, resulting in the impairment of beneficial uses of recreational lakes. In estuarine and marine waters, marine invasive such as Calpuera, Sargassum, and Undaria are aggressive invaders and displacing native kelp and seagrass habitats (Kaplan et al. 2016). ln Southern California wadeable streams, 20-30% of stream miles are impacted by elevated macroalgal blooms (Mazor et al. 2018) and eutrophication was identified a top causal stressor in degraded wadeable stream benthic macroinvertebrate and algal communities in that region. Similarly, McLaughlin et al. (2014) found that half of Southern California Bight estuaries had macroalgal biomass exceeding thresholds known to impact benthic macroinvertebrates (Sutula et al. 2014, Green et al. 2013) and seagrass (Bittick et al. 2018) and one third had chronic blooms >2 months. Similar high biomass, chronic blooms have been observed in the estuaries of other regions including Elkhorn Slough (Hughes et al. 2011), Humboldt and Bodega Bays (Sutula et al. 2014) and Morro Bay and Tomales Bay (Bittick et al. 2018). These algal blooms pose a threat to native organisms and habitats and can impede recreational (REC1 and REC2) uses such as swimming and standup paddleboarding.

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Figure 2.9. Non-toxic algal blooms: Sargassum off Catalina Island (top left, photo by Tom Boyd), herbicide being sprayed on Eichhornia crassipes (top right, photo by California State Parks Division of Boating and Waterways), algal scum in Santa Margarita River (photo by SCCWRP), news article about fish death from a golden algae bloom.

**Diel DO and pH Swings, Hypoxia and Acidification.** Dissolved oxygen is necessary to sustaining the life of all aquatic organisms that depend on aerobic respiration. Eutrophication produces excess organic matter that fuels the development of low surface water DO concentrations (hypoxia) as that organic matter is respired (Diaz 2001). When the supply of oxygen from the surface waters is reduced or the consumption of oxygen exceeds the resupply (via decomposition of excessive amounts of organic matter), oxygen concentrations can decline below the limit for survival and reproduction of benthic (bottom-dwelling) or pelagic (water column dwelling) organisms (Stanley and Nixon 1992, Borsuk et al. 2001, Diaz 2001). Hypoxia has several adverse effects on aquatic organisms, including: lowered growth rates, altered behavior, reduced reproductive success, and diminished survival (Diaz and Rosenberg 1995; Breitburg et al. 1997, 2009; Vaquer-Sunyer and Duarte 2008). Changes in the survival and reproduction of benthic and pelagic organisms can result in habitat and biological diversity losses, foul odors and taste, and altered food webs (USEPA 2007). Consequently, management of hypoxia in aquatic habitats has become a global issue (Smith et al. 1987, Karlson et al. 2002, OSPAR 2003, Diaz and Rosenberg 2008), yet in California DO is not typically monitored using moored sensors to produce a continuous data stream that allows assessment of extent of hypoxia. Anecdotal evidence shows that hypoxia is likely extensive. For example, in hypoxic events in SCB estuaries is extensive, with at least 53% of segments had at least one event 30 longer than 24 hours (McLaughlin et al. 2014). Estuarine habitats with investments in long term monitoring show chronic and recurrent hypoxia and even anoxia, e.g., in Tijuana River estuary and Los Penasquitos Lagoon (McLaughlin et al. 2013), San Elijo Lagoon (Sutula at al. 2017), Malibu Lagoon (Sutula et al. 2006), Newport Bay (Sutula et al. 2006), Pajaro Slough, Elkhorn Slough (Jeppesen et al. 2016), and Suisun Marsh (Baily et al. 2014).

In California, continuous diel measurements of DO are less common in inland aquatic habitats, particularly in wadeable streams, though high diel variability in pH and DO as well as chronic low DO has been documented in river TMDLs (Klamath River, North Coast Water Board 2010; Santa Margarita River, Sutula et al. 2018); discrete DO samples are more prevalent, leading the Central Valley Water Board to utilize discrete DO measures to estimate the maximum variability in DO associated with stream algal blooms (Worcester et al. 2010), as the basis for their Board’s interpretation of biostimulatory objectives. Extent of hypoxia in California lakes is not well understood, because of lack of routine monitoring.

Acidification is a dominant driver of long-term changes in pH in the coastal oceans and is largely attributable to anthropogenic CO2 emissions, raising concern for the future of calcifying organisms, many of which are present in estuarine habitats (Feely et al. 2008, Bednarsek et al. 2014, 2017); however, changes in pH in freshwater and estuarine ecosystems can result from a multitude of drivers, including local pollution sources (nutrients and carbon) and altered watershed processes (Duarte et al. 2013). However, the instrumentation required to discern eutrophication-driven acidification is not routinely used in estuarine and inland water quality monitoring, despite anecdotal evidence that acidification may be strongly linked to eutrophication and therefore of concern (Trowbridge et al. 2017; Wasson, personal communication).

**Aquatic Vegetation**. Native submerged and floating vegetation can be beneficial components of aquatic ecosystems; however, several non-native species have been found to adversely ecosystem services and associated beneficial uses at the high densities at which they typically occur, e.g., in the SFB Delta (Boyer and Sutula 2017) and in other estuaries (e.g., Malibu Lagoon, Buena Vista Lagoon, Sutula et al. 2006. McLaughlin et al. 2012). Adverse effects include: (1) changes to water chemistry, including diurnal swings in pH and dissolved oxygen, (2) changes to physical properties of water, including flow and turbidity, (3) outcompeting of native SAV, phytoplankton, and other benthic primary producers, (3) changes to the food web, (4) impedance of navigation and obstruction of water conveyance, and (5) poor aesthetics. As an example of this, two species in the Delta, the invasive species, *Egeria densa* (Brazilian waterweed, a submersed species) and *Eichhornia crassipes* (water hyacinth, a floating species), are widely recognized as problematic and appear to be increasing in abundance despite control efforts. Existing scientific literature has documented several environmental and management-related factors that have control over the growth of invasive aquatic plants, including biostimulatory drivers (Boyer and Sutula 2017). Climate change and anthropogenic activity associated with land use changes have the potential to further increase the prevalence of invasive macrophytes, and thus this remains an area of increasing concern for biostimulatory impacts.

## 2.5 Biostimulatory Substances and Conditions

Biostimulatory drivers of eutrophication consist of nutrient pollution (including forms of nitrogen and phosphorus and associated organic matter) and other conditions that can cause or significantly contribute to eutrophication. Like nutrient pollution, these factors, such as hydromodification, physical habitat alteration, altered water temperature, and light availability, and grazing pressure are associated with conversion of watersheds and coastal zones to developed land uses (Figure 2.10). Climatic change, specifically global warming, increased water column vertical stratification, and heightened frequency and intensity of drought and storm events can exacerbate these biostimulatory conditions, thus modulating eutrophication (Paerl et al. 2011).

Harmful algal blooms (HABs) in freshwater systems (rivers and lakes), particularly cyanobacteria, tend to become limited by P sooner than by N, and thus are frequently linked with excessive P loading (Likens 1972, Schindler 1977, Edmondson and Lehman 1981, Elmgren and Larsson 2001, Paerl 2008, Schindler et al. 2008). In contrast with freshwater systems, estuarine and marine systems tend to be more sensitive to N loading (Ryther and Dunstan 1971, Nixon 1986, Suikkanen et al. 2007, Paerl 2008, Conley et al. 2009, Ahn et al. 2011). Traditionally, phosphorus (P) input reductions have been prescribed to control freshwater algal blooms, because P limitation is widespread and some CyanoHABs can fix atmospheric nitrogen N2 to satisfy their N requirements. However, eutrophic systems are increasingly plagued with non N2 fixing CyanoHABs that are N and P co-limited or even N limited. Moreover, both non-point and point source nutrient contributions, such as agriculture and wastewater effluent, tend to increase N and P concentrations simultaneously (Paerl and Paul 2012, Paerl et al. 2014b). At low and intermediate nutrient loadings, reduction in only N or P may be sufficient to control blooms. But with elevated loadings of both N and P, reduction of only one type of nutrient can lead to an imbalance in the N:P ratio of the water column potentially leading to a worsening of HAB problems (Smith 1983; Paerl and Huisman 2008; Pearl et al. 2011, 2014b). Therefore, N and P input constraints are likely needed for long-term eutrophication control in such systems (Paerl et al. 2011).

Sources of nutrient pollution released to freshwater and coastal areas are diverse, and include agriculture, animal husbandry, aquaculture, septic tanks, urban wastewater, urban stormwater runoff, industry, and fossil fuel combustion. Nutrients enter aquatic ecosystems via the air, surface water, or groundwater (Figure 2.11). Among regions and from watershed to watershed, there are signiﬁcant variations in the relative importance of nutrient sources and pathways that contribute to eutrophication of local and coastal waterbodies

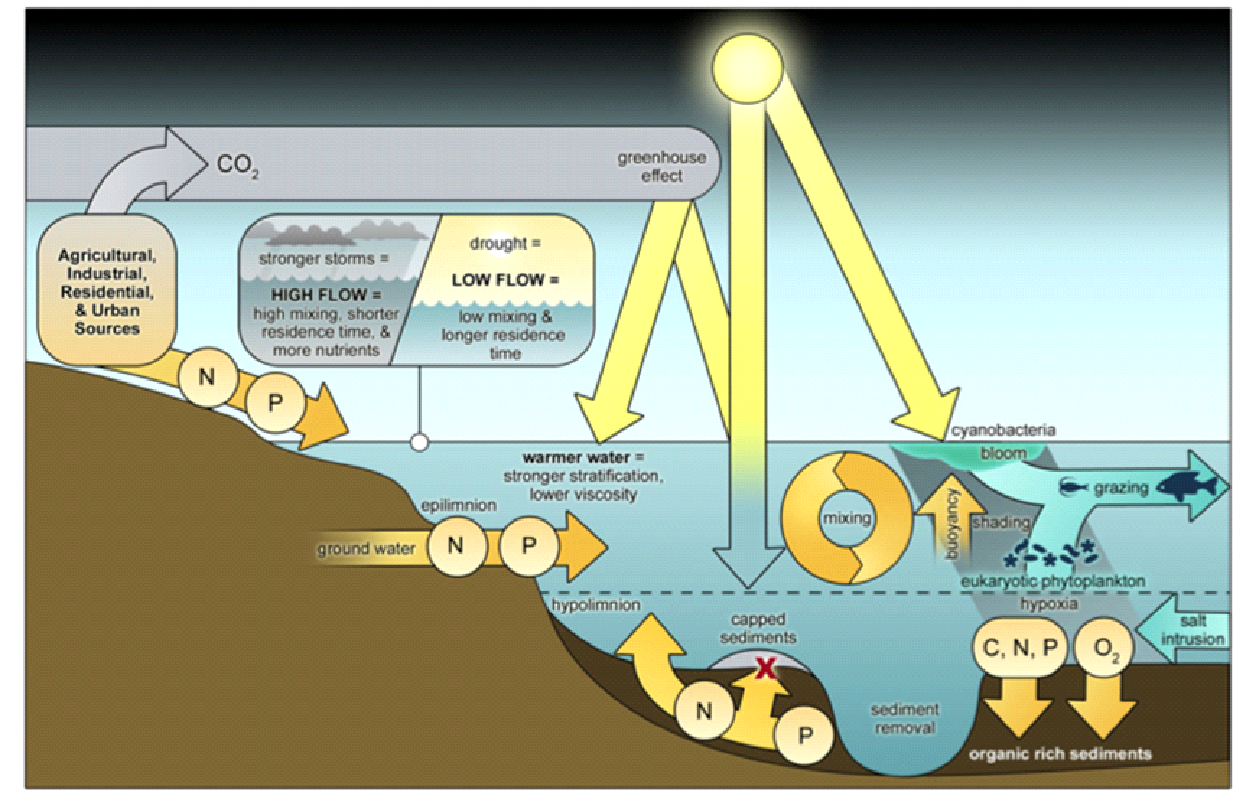


Figure 2.10. Conceptual model of some of the co-factors affecting algal blooms including warmer water, drought and decreased flow, decreased mixing, increased residence time, and increased N and P inputs from agricultural, industrial and urban sources. From Paerl et al. 2011.

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|  |  | Image result for la freeways during a rainstorm |
|  |  | Kennedy Jenks Wastewater Treatment Plant |

Figure 2.11. Source of nutrients to rivers, streams, and estuaries, clockwise from top left: discharge from an ag tile drainage in Central Coast, stormwater runoff and other runoff in storm drains and transporation sources (top middle and right), wastewater treatment plant (bottom right), agriculture/animal husbandry (bottom middle), and fire plume causing atmospheric deposition (Photo sources: Wikimedia commons, City of Riverside, epa.gov, sewerhistory.org).

At a statewide scale, the USGS SPARROW models for California (Saleh and Domagalski 2015, Domagalski and Saleh 2015) provides loading estimates for both nitrogen and phosphorus from mostly HUC6 watersheds (Fig 2.12). Manure and synthetic fertilizers associated with agriculture contributes 51.7% of the TN and 38.4% of TP on a mass basis to surface waters. Most of agricultural loadings are from three basins: the Sacramento River, San Joaquin River and Tulare-Buena Vista Basins. These three basins account for 79% of agricultural phosphorus loads statewide? There is also significant agriculture and loading? in the Salinas Valley and the Imperial Valley. Within other regions, ag-dominated land uses within individual watersheds can provide significant sources of nutrient to fuel eutrophication (e.g., Santa Margarita River in San Diego County, Sutula et al. 2014). Nutrient pollution in these areas not only lead to surface water eutrophication symptoms, they also lead to groundwater contamination, resulting in impairment or restrictions on drinking water use (Harter et al. 2012). For example, the Tulare Basin is an agricultural area that receives roughly 176 Gg of fertilizer and 127 Gg of manure for land application each year. As it is a closed basin, the nitrogen that is not used for plant uptake ends up in groundwater. As a result, one outcome is that safe drinking water sources are being provided to the populations living in these basins, at significant economic cost. The prevention of future nitrate contamination has become a focus of the State Board’s regulatory programs for irrigated lands and for dairies to regulate nitrate loadings.

Other major sources of nutrients in CA are the publicly owned treatment works (POTWs) that treat the municipal sewage (Salah and Domagalski 2015). The SPARROW models estimate that 15.9% of nitrogen and 23% of phosphorus in surface waters is from wastewater. Sewage loadings across the region are a function of population density. The largest loads are from Southern California, followed by San Francisco Bay Area and the greater Sacramento area. For perspective, roughly 48 Gg N/yr are discharged by ocean dischargers in Southern California (Howard et al. 2014). This does not include inland POTW discharges in this area which contribute loads via river runoff, e.g., Los Angeles County’s inland plants, City of Los Angeles’s Tillman and Glendale plants or the City of Burbank’s plant which together contribute another 6.1 Gg N/yr. San Francisco Bay POTWs discharge around 20.6 Gg N/yr (Viers et al. 2012). Sacramento Regional POTW discharges around 4.7 Gg N/yr (Howard et al. 2014). POTW discharges in the rest of the Central Valley discharge 0.34 Gg N/yr (EPA Permits Database). These high nutrient loadings to these surface waters have the potential to impact beneficial uses. Smaller wastewater discharges may also cause problems in rivers where there is insufficient dilution accommodate additional nutrient loadings.

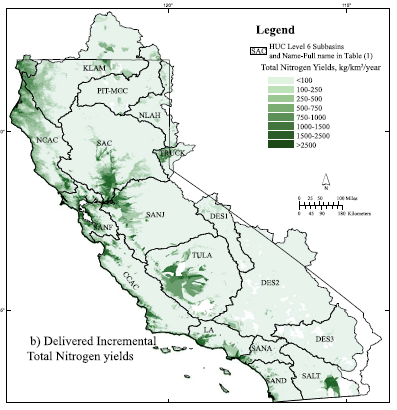
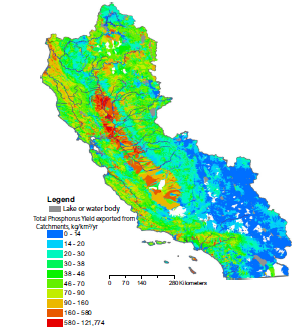
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Figure 2.12. Results from USGS SPARROW model for California. Left panel is from Saleh and Domagalski (2015), while right panel is from Domagalski and Saleh (2015). Yield is the amount of N or P per unit area of land per year. (Kg/m2/yr).

### 2.5.2 Irradiance, Water Clarity, and Temperature

Solar radiation is of fundamental importance of all aquatic ecosystems. Irradiance is central to metabolism and photosynthesis, and thus plays an important role in the growth, diversity and density of algae and aquatic plants, as algal and aquatic plants species have an optimal range of irradiance. In addition, the absorption of thermal energy and the dissipation of heat have profound effects on thermal structure, water mass stratification, and circulation patterns. Changes in radiant energy, e.g, from physical habitat alteration (e.g., engineered channels, removal of riparian habitat, hydromodification), natural events (fires and floods), and climate change can impact the composition and relative abundance of primary producers, which compete for light and space. As an example of this, submerged macrophytes occurs in many shallow water bodies with increasing eutrophication, due to reduced water clarity from algal blooms (REF). Cyanobacteria such as Microcystis, can grow very close to the surface by tolerating irradiance levels that are inhibitory to other members of the phytoplankton community, increasing their cell densities past the point where they would ordinarily become light-limited by self-shading (Carey et al. 2012).

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Figure 2.13. Concrete channels, floods, and fires that result in the loss of riparian habitats lead to increased temperature in rivers and streams. Top left: Los Angeles River (photo by California State Parks), top right: South Yuba River (photo from Wikimedia commons), bottom left: tributary to the Los Angeles River near Van Nuys (photo from Wikimedia commons), bottom right: (photo from SCCWRP).

As an example of this, submerged macrophytes occurs in many shallow water bodies with increasing eutrophication, due to reduced water clarity from algal blooms (Boyer and Sutula 2015).

Temperature is one of the most important factors in controlling the growth rate and community composition of primary producers and prokaryotes responsible for the respiration of dead and decaying organic matter (Robarts and Zohary 1987, Butterwick et al. 2005, Reynolds 2006, Paerl and Huisman 2008). Temperature will have profound influences in restructuring primary producer communities. For example, the difference in the optimum growth temperatures of the various phytoplankton taxa is hypothesized to become increasingly important in determining phytoplankton community composition as global temperatures continue to increase above 20°C (Lehman et al. 2005, Paerl and Huisman 2008). Reynolds (2006) suggests that in freshwater ecosystems with a mixed phytoplankton assemblage, all else being equal, cyanobacteria will be able to grow faster and outcompete other phytoplankton taxa as the temperature increases. With continued climate change and global warming, there’s an increased risk that cyanoHABs will become increasingly competitive vis-à-vis diatoms which often dominate community composition in temperate regions. Temperature controls the growth rate of all prokaryotes responsible for the respiration of dead and decaying organic matter (Paerl et al. 2011), thus exerting a major control on oxygen demand and hypoxia as well as other key rates of nutrient transformations in the water column and in the sediments (e.g., nitrification, denitrification, sulfate reduction, etc.).

### 2.5.3 Hydromodification

USEPA (1993) defines hydromodification as the “alteration of the hydrologic characteristics of surface waters, which in turn could cause degradation of water resources.” Hydromodification can occur through urbanization (water withdrawals, and inter-basin transfers, channelization and channel modification, construction of dams and impoundments, streambank and shoreline erosion, as well as hydrograph modification) and climate change. Net impervious surface, increases with urbanization, leading to accelerating runoff velocities and increases in base hydrological flows. Changes in climate encompass both short-term changes, such as seasonal or annual variability that can lead to conditions such as drought, and long-term changes that are being triggered by global climate change. Climate change in California will impact hydrological patterns. Preliminary studies of downscaled models indicate an increase in overall temperature in California, a shift from less snow to more rain, and an increased frequency of extreme drought and storms. Under these conditions, the magnitude and duration of hydrological flows will accelerate runoff velocities and alter base flows. Water resource management has had a tremendous impact on natural hydrological budget of all of California (Reisner 1986), through dams, flood control of streams and rivers, water withdrawals and interbasin transfers). These changes have fundamentally altered basic physical and hydrodynamic forcing of aquatic habitats, which constitute major controls on eutrophication.

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Figure 2.14. Examples of hydromodification. (Photo source: top left Wikimedia Commons, top right esassoc.com, bottom left SCCWRP, bottom right SCCWRP).

Taken together, these alterations can result in a myriad of impacts that lead to physical habitat alteration, modified hydrographs and hydroperiods, stratification, and altered hydraulic residence times. The ecological impacts are strongly linked to eutrophication, via pathways of altered channel substrate (including hardscaping), increased sedimentation, higher water temperature (see previous section), longer residence time, reduced surface water reaeration. The outcomes are several-fold—including but not limited to higher organic accumulation, lower dissolved oxygen, and a fundamental shift in natural biological communities, including a loss of fish and other aquatic populations. Drought-prone California is increasingly focused on development of sustainable water supplies that promote conservation, water-use efficiency, conjunctive use, wastewater recycling, groundwater remediation, and desalination. These actions may result in changes in conditions that can either exacerbate or mitigation eutrophication.

### 2.5.4 Other Biostimulatory Factors

Additional to the above-mentioned factors, several others may influence eutrophication, including grazing by higher trophic levels Jassby 2008, Lucas et al. 2016) and exposure to toxic compounds such as herbicides and pesticides. For example, substantial variability exists in sensitivity to herbicides among cyanobacteria compared with other phytoplankton such as green algae and diatoms (Peterson et al. 1997, Lurling and Roessink 2006), potentially leading to dominance by HAB species. Herbicides or pesticides may depress growth or consumer grazing of algae or aquatic plants on site, but blooms may form downstream once the inhibitory effects of the chemicals have dissipated.

### 2.5.5 Origins, Variability and Natural Background in Biostimulatory Substances and Drivers

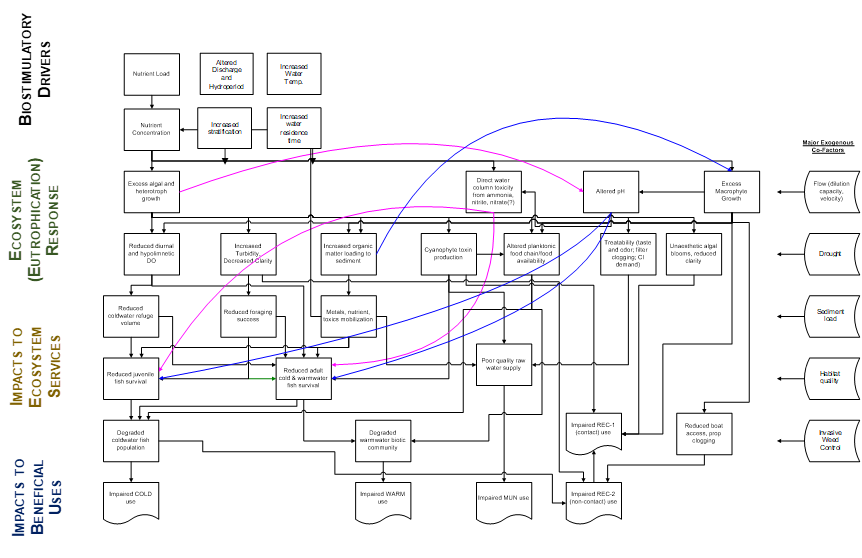
Biostimulatory drivers can be local, regional or global in origin, with high temporal and spatial variability, and some component of natural background forcing that can contribute to eutrophication symptoms.Biostimulatory drivers can be *in situ* (e.g., sediment benthic flux of nutrients, legacy organic matter), upstream sources or within the hydrological basin or watershed (e.g., nutrient inputs, organic matter loading scoured from upstream algal blooms, regional (e.g., atmospheric deposition), or global (elevated CO2, temperature from global). Biostimulatory drivers can be temporally and spatially variable, driven by factors such as complexity of land use within a watershed, climate forcing, and watershed and coastal hydrodynamics. Natural background conditions (e.g., geology, headwater springs) can produce locally-elevated nutrient concentrations or biostimulatory conditions that should be considered in the synthesis of biostimulatory drivers within a watershed. Climate events such as drought, fires, flood events, etc. can dramatically change biostimulatory conditions within a watershed for extended periods. Such natural background and event-driven biostimulatory conditions should be considered when developing establishing compliance for biostimulatory targets.

## 2.6 Development of Watershed–Specific Conceptual Models of Biostimulatory Impacts

Ultimately, an effort to develop watershed or waterbody nutrient management and restoration strategies should begin with development of a detailed conceptual model, (e.g., Figure 2.5) followed by identification of the most significant risk hypotheses, and analysis of the actions necessary to minimize the risk and protect uses. To develop an approach that is applicable across many sites it is necessary to prioritize the list to identify, in generic form, those risk hypotheses that are the most important causes of use impairment and can stand in as surrogates for other, less significant or less common risk hypotheses.

The detailed conceptual model features the major environmental co-factors that influence how nutrients et al. biostimulatory drivers are processed within a water body, and / or have a direct impact on the endpoints (Figure 2.5, on the right side). These co-factors are important to consider because they affect the way in which biostimulatory drivers are processed in a waterbody and thus the associated risk of use impairment and are an integral part of the decision-making process to maintain or restore water body integrity. The co-factors operate in different ways in different types of waterbodies. For example, light availability is an important cofactor for both streams and lakes; however, in headwater streams light availability is primarily controlled by riparian vegetation, while in lakes it is often most strongly controlled by turbidity and lake management practices. They can include natural gradients (geology) and anthropogenic factors, at local (e.g., water conservation actions) and regional -global scales (climate change).

Detailed literature reviews including conceptual models and crosswalks to response indicators are available for California wadeable streams (Sutula et al. 2018) and estuaries (Sutula 2011), including San Francisco Bay (Senn et al. 2013) and the Delta (Berg and Sutula 2015, Boyer and Sutula 2015, Dahm et al. 2016). Additional linkages may be significant in individual waterbodies; however, most of the major linkage connections are captured in these figures. Other examples of eutrophication conceptual models are available through EPA’s CADDIS (Norton et al., 2009) website as “interactive conceptual diagrams” ([www.epa.gov/caddis/cd\_icds\_intro.html](http://www.epa.gov/caddis/cd_icds_intro.html)).



*Figure 2.15. Example of detailed conceptual model of biostimulatory drivers, eutrophication responses, and impacts on uses in a lake. Modified from Tetra Tech (2006).*

# Organizing Assumptions and Scientific Principles for Assessment, Prevention and Management of Biostimulatory Impacts

The California State Water Board has initiated a project to develop a consistent interpretation to narrative biostimulatory objectives found in all Regional Water Board basin plans and to develop numeric guidance for wadeable streams as Phase I, with other waterbody types in subsequent phases (SWRCB 2014). That current project is preceded by nearly two decades or more of technical and policy discussions on approaches for developing nutrient objectives (nutrient numeric endpoint or NNE; Tetra Tech 2006; SWRCB 2014). During the same time period, Water Board staff has funded or sponsored science to develop the bases for numeric objectives (Tetra Tech 2006) in wadeable streams (Mazor et al., in prep, Theroux et al. in prep, Paul et al. in prep), estuaries and enclosed bays (Sutula 2011, Sutula et al. 2012, Green et al. 2013, Sutula et al. 2014, McLaughlin et al. 2014), and has recently funded work in lakes. These scientific approaches are supported by more than a decade of experience in addressed eutrophication through TMDL.

These organizing assumptions and scientific principles are consistent across all waterbody types and form the common basis for scientific approaches to assess, prevent, and manage eutrophication (Table 3.1).

Table 3.1. Summary of Organizing Assumptions and Scientific Principles Supporting the Assessment, Prevention and Management of Biostimulatory Impacts. These assumptions and principles are explained in more detail below. Bolded terms are defined in Appendix 1.

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| 1. ***“*Biostimulatory drivers*”*** are defined as **substances** such as nutrients (i.e., nitrogen and phosphorus and associated organic matter) or **condition**s, such as altered physical habitat, temperature, hydrology, etc. that can cause eutrophication. |
| 2. Assessment of biostimulatory impacts is based on the diagnosis of eutrophication and its consequences (e.g., poor odors and taste, cyanotoxins); inclusion of causal nutrients or other biostimulatory drivers are part of a comprehensive causal assessment and risk prevention approach. |
| 3. Biostimulatory impacts to beneficial uses will be assessed through an assessment framework developed for each waterbody type, with indicators that represent lines of evidence. |
| 4. Assessment of biostimulatory impacts should consider evidence for impacts to both human and wildlife (aquatic and terrestrial) related beneficial uses. |
| 5. Statewide indicesof ***biological integrity*** can be used as assessment endpoints from which to derive ranges of biostimulatory targets that are protective of aquatic life related beneficial uses. |
| 6. To address total “biostimulatory” potential, thresholds should be based on total nutrients (as opposed to dissolved inorganic form) and for both nitrogen and phosphorus, as opposed to just controlling what is considered the limiting nutrient on-site (either nitrogen or phosphorus). |
| 7. Eutrophication symptoms may be caused by biostimulatory drivers far-field from the waterbody, and thus assessment of biostimulatory impactsshould take a watershed-wide approach. |
| 8. Biostimulatory conditions can be a focal point of development of watershed-specific numeric targets and adaptive management strategies. |
| 9. Implementation options to address biostimulatory conditions and substances should recognize the complexity of these drivers and how they can vary spatially and temporally from watershed to watershed and among certain waterbodies. |
| 10. Generic conceptual models of biostimulatory impacts to waterbodies, presented here, should be refined to illustrate key hypotheses for how biostimulatory substances and conditions are linked to eutrophication symptoms and their relationship to designated waterbody uses. |

***1. “Biostimulatory” is defined as to substances such as nutrients (i.e. nitrogen and phosphorus and associated organic matter) or conditions, such as altered physical habitat, temperature, hydrology, etc. that can* cause eutrophication.** Eutrophication is defined as the accelerated accumulation of organic matter from in situ process or external loading (Nixon 1995) and is driven by a suite of natural and anthropogenic factors such as substances (e.g., nutrients --nitrogen, phosphorus, silica and associated organic matter) and conditions (e.g., hydromodification, temperature and light regime changes, physical habitat modification, etc.). While Regional Board Basin Plan narrative objectives specify substances, a suite of recent TMDLS has interpreted the objective more broadly to include conditions (Klamath River nutrient TMDL, Loma Alta Slough TMDL, etc.).

***2. Assessment of biostimulatory impacts is based on the diagnosis of eutrophication or other adverse effects and its consequences; inclusion of causal biostimulatory drivers is part of a comprehensive causal assessment and risk prevention approach.*** Eutrophication (response) indicators and causal biostimulatory indicators should be used to assess biostimulatory impacts; Table 2.3; Cloern et al. 2001, Paerl et al. 2011). The rationale relates to causal pathways; ecological response to biostimulatory substances and conditions causes adverse effects that impair uses. Highlighting response indicators reduces the likelihood of false negatives and false positives in diagnosing impacts to beneficial uses. In some cases, nutrients themselves can directly impair beneficial uses through toxicity (e.g., ammonia) or by exceeding the range of organismal tolerances, but evidence of biostimulatory impacts and information on causal pathways are strengthened by the inclusion of one or more response parameters, including eutrophication, biological integrity, or human use measures (e.g., cyanotoxins). These concepts are consistent with recent EPA guidance on combined use of response and causal drivers to establish nutrient criteria (EPA 2013). When possible, the use of multiple indicators in a “weight of evidence” approach provides a more robust means to assess ecological condition, determine causal pathways of impairment, and to inform management options to address the impairment (Karr and Chu 1999).

***3. Biostimulatory impacts to beneficial uses can be assessed through a framework developed for each waterbody type, with eutrophication indicators that represent lines of evidence.*** Water Board staff guidance on biostimulatory impact assessment can be supported by scientific evaluations of available eutrophication indicators. These evaluations should yield an understaning of the confidence in that indicator to represents a primary or supporting line of evidence. A primary line of evidence designates a higher level of confidence that this indicator can be used to assess eutrophicatoin, based on a wealth of experience and knowledge about how this indicator captures and represents ecological response. Other indicators that do not meet that level of confidence may be used as supporting lines of evidence. Four evaluation criteria were chosen by regulatory and stakeholder advisory groups (Sutula 2011) to evaluate eutrophication indicators for eutrophication assessment, in estuaries and enclosed bays (e.g., Sutula 2011, Sutula and Senn 2016) and wadeable streams (Fetscher et al. 2014, Sutula et al. 2018): 1) response indicator has documented linkages to beneficial uses; 2) the indicator has a well-vetted and cost-effective means of measurement; 3) science exists to model the relationship between the response indicator and biostimulatory substances (e.g., nutrients) or biostimulatory conditions; and 4) the indicator has a proven track record of acceptable signal to noise ratio for eutrophication assessment. Taken together, these evaluation criteria can be used to judge whether an indicator qualifies as a primary or supporting line of evidence.

***4. Assessment of biostimulatory impacts should consider evidence for impacts to both human and wildlife (aquatic and terrestrial) related uses.*** Eutrophication has the potential to impact human uses through direct impacts to human health (e.g., HAB toxins via REC1 and MUN) and other services (e.g., aquatic macrophytes that choke intake on industrial or municipal water intake valves, impacts to aesthetic recreational experiences). Other aquatic and terrestrial wildlife related beneficial uses can be impacted through a variety of impairment pathways. While statewide numeric guidance for assessment of biostimulatory impacts may be focused on a selected suite of standardized monitoring indicators and protocols that are broadly representative of aquatic life or biological integrity (e.g., benthic macroinvertebrates and algae), assessment should consider any supporting lines of evidence that link to all uses designated for that waterbody.

***5. Statewide bioassessment indices, can be used as assessment endpoints from which to derive ranges of nutrients and biostimulatory conditions that are protective of aquatic life related beneficial uses****.* Statewide bioassessment indices such as CSCI, ASCI and AMBI are state accepted measures of aquatic life related beneficial uses (e.g., WARM, COLD, EST, MAR). Protection goals for these indices (e.g. Mazor et al. 2016) can serve as “assessment endpoints” from which to derive the ranges of nutrients and biostimulatory conditions (flow, temperature, etc.) that are protective of aquatic life (e.g., have a low risk of impacting uses). In this fashion, policies and approaches that support biointegrity and biostimulatory can be synergistic, recognizing that biostimulatory drivers are a major category of causal pathways by which biointegrity may become impaired. Bioassessment indices may be used for assessment of eutrophication as a line of evidence of impaired biointegrity; however, because these indices are constructed to be responsive to generalized stressor gradients, they are not diagnostic of eutrophication. Evidence for indicators such as those presented in Table 2.3 should be included in the assessment framework to confirm the biostimulatory impacts as a causal pathway.

***6. To address total “biostimulatory” potential, numeric targets should be based on total nutrients (as opposed to dissolved inorganic form) and for both nitrogen and phosphorus, as opposed to just controlling what is considered the limiting nutrient on-site (either nitrogen or phosphorus).*** Even though anthropogenic nutrient pollution typically results in elevated dissolved inorganic nitrogen and phosphorus (e.g., nitrate, ammonium, phosphate), numerous studies support the use of total nitrogen (TN) and total phosphorus (TP) as nutrient targets (Paerl et al. 2011), because high dissolved organic N and P released in point and nonpoint sources, algal and plant biomass etc. can be subsequently remineralized to dissolved inorganic nutrients and therefore made bioavailable to further fuel eutrophication, albeit over longer timescales (Fan et al. 2017). Establishment of nutrient targets should include both N and P limits, because aquatic primary producer responses to single nutrient reduction strategies (N or P) may exacerbate eutrophication problems and impacts on uses (Paerl et al. 2011). Organic matter may be part of the TN and TP targets in surface waters, or separate targets can be established to represent organic matter in benthos (benthic chlorophyll-a or organic carbon) or sediment (sediment percent organic carbon).

***8. Biostimulatory conditions can be a focal point of development of watershed- or waterbody specific numeric targets and adaptive management strategies.*** Eutrophication symptoms can be caused by factors other than or in addition to nutrients and organic matter loading. In some cases, the level of nutrients may not cause eutrophication but modification of waterbody condition, such as loss of riparian habitat, results in the development of eutrophic conditions.Mitigation of biostimulatory conditions through environmental flow management, low impact development, and watershed or waterbody restoration may be more cost effective than a sole focus on nutrient management alone. In some cases, it may be possible to derive regional, watershed, or waterbody specific flow or temperature targets that are protective against eutrophication impacts. In other cases, derivation of numeric targets may not be possible, but proactive implementation of restoration strategies (e.g., floodplain restoration, etc.) that improve physical habitat, restore natural hydroperiods, and reduce water temperatures or light availability may be effective in mitigating eutrophication.

***7. Eutrophication symptoms may be caused by biostimulatory conditions far-field from the waterbody, and thus biostimulatory assessment should take a watershed-wide approach.***Eutrophication symptoms such as low dissolved oxygen, elevated HAB toxic concentrations, or poor benthic habitat quality can be the result of organic matter or toxin loading from upstream or distal sources. Such symptoms may be present without evidence of high onsite primary production (e.g., elevated chlorophyll-a). Examples of this is the detection of high concentrations of freshwater cyanoHAB toxins in estuaries with low cyanoHAB cell densities (Peacock et al. 2018) and the impact of upstream benthic algal blooms on low dissolved oxygen in lower Santa Margarita River main stem (Sutula and Shultz 2018). There can be both spatial and temporal lags in symptoms linked to eutrophication drivers. For example, nutrient laden sediments from surface waters in the upper watershed can cause “internal sources” of nutrients via benthic flux in lakes and estuaries. Contaminanted groundwater sources, particularly from agriculture (Harter et al. 2012), can be a source of eutrophication in surfaces waters (McLaughlin et al. 2013).

***9. Management of biostimulatory impacts to waterbodies should begin with conceptual model development that illustrates key hypothesis for how biostimulatory substances and conditions are linked to eutrophication symptoms and their relationship to designated waterbody uses.*** Ultimately, an effort to develop watershed or waterbody nutrient management and restoration strategies should begin with development of a detailed conceptual model, followed by identification of the most significant risk hypotheses, and analysis of the actions necessary to minimize the risk and protect uses.

***10. Management approaches to biostimulatory conditions and substances should recognize the complexity of these drivers and how they can vary spatially, temporally, from watershed to watershed and among waterbodies.*** Biostimulatory drivers can be local, regional or global in origin, with high temporal and spatial variability, and some component of natural background forcing that can contribute to eutrophication symptoms. Biostimulatory drivers can be temporally and spatially variable, driven by factors such as complexity of land use within a watershed, climate forcing, and watershed and coastal hydrodynamics; seasonal or spatially-varying targets can help to address this complexity. Natural background conditions (e.g., geology, headwater springs) can produce locally-elevated nutrient concentrations or biostimulatory conditions that should be considered in the synthesis of biostimulatory drivers within a watershed. Climate events such as drought, fires, flood events, etc. can dramatically change biostimulatory conditions within a watershed for extended periods. This variability and the potential for natural background and event-driven biostimulatory conditions should be considered when establishing biostimulatory targets.

California wide, annual estimates of natural, point and nonpoint sources of total nitrogen and total phosphorus are available at a watershed scale through the USGS SPARROW portal. This tool is an important starting point to prioritize nutrient management within a region and can guide data collection and assessment for sources within a specific watershed. Dynamic watershed loading and receiving water models are important tools to synthesize understanding of a fuller suite of biostimulatory drivers and eutrophication responses specific to each watershed or waterbody and to support conversation about potential implementation strategies.

# Summary

This document summarizes the scientific foundation underpinning the consistent interpretation of a narrative biostimulatory objective applicable to all water body types in the State of California. The review provides a synthesis of information with general conceptual models of risk pathways by which nutrient pollution and eutrophication can impair beneficial uses, specific to waterbody class. It also provides several fundamental principles for assessment, prevention and management of biostimulatory impacts applicable to all waterbodies. These conceptual models, indicators, and guiding principles provide the initial basis for Water Board guidance to support the consistent application of a narrative biostimulatory objective across waterbody types, in advance of numeric guidance developed for each waterbody class. These waterbody class–specific technical documents will be published as appendices to this main document, as they become available.

# 5. References

Ahn, C.-Y., Oh, H.-M., Park, Y.-S., 2011. Evaluation of Environmental Factors on Cyanobacterial Bloom in Eutrophic Reservoir Using Artificial Neural Networks1. Journal of Phycology 47, 495–504. <https://doi.org/10.1111/j.1529-8817.2011.00990.x>

Anderson, C.R., Siegel, D.A., Brzezinski, M.A., Guillocheau, N., 2008. Controls on temporal patterns in phytoplankton community structure in the Santa Barbara Channel, California. Journal of Geophysical Research: Oceans 113. <https://doi.org/10.1029/2007JC004321>

Anderson, D.M., Cembella, A.D., Hallegraeff, G.M., 2012. Progress in Understanding Harmful Algal Blooms: Paradigm Shifts and New Technologies for Research, Monitoring, and Management. Annual Review of Marine Science 4, 143–176. <https://doi.org/10.1146/annurev-marine-120308-081121>

Anderson-Abbs, B., Howard, M., Taberski, K., Worcester, K., 2016. California Freshwater Harmful Algal Blooms Assessment and Support Strategy.

Baily, H., Curran, C., Poucher, S., Sutula, M.A., 2014. Science Supporting Dissolved Oxygen Objectives for Suisun Marsh (Technical Report No. 830). Southern California Coastal Water Research Project, Costa Mesa, CA.

Bargu, S., Smith, E., Ozhan, K., 2011. Toxic Diatom Pseudo-nitzschia and Its Primary Consumers (Vectors), in: Seckbach, J., Kociolek, P. (Eds.), The Diatom World, Cellular Origin, Life in Extreme Habitats and Astrobiology. Springer Netherlands, Dordrecht, pp. 491–512. <https://doi.org/10.1007/978-94-007-1327-7_22>

Bates, S.S., Bird, C.J., Freitas, A.S.W. de, Foxall, R., Gilgan, M., Hanic, L.A., Johnson, G.R., McCulloch, A.W., Odense, P., Pocklington, R., Quilliam, M.A., Sim, P.G., Smith, J.C., Rao, D.V.S., Todd, E.C.D., Walter, J.A., Wright, J.L.C., 1989. Pennate Diatom Nitzschia pungens as the Primary Source of Domoic Acid, a Toxin in Shellfish from Eastern Prince Edward Island, Canada. Can. J. Fish. Aquat. Sci. 46, 1203–1215. <https://doi.org/10.1139/f89-156>

Bates, S.S., Freitas, A.S.W. de, Milley, J.E., Pocklington, R., Quilliam, M.A., Smith, J.C., Worms, J., 1991. Controls on Domoic Acid Production by the Diatom Nitzschia pungens f. multiseries in Culture: Nutrients and Irradïance. Can. J. Fish. Aquat. Sci. 48, 1136–1144. <https://doi.org/10.1139/f91-137>

Bates, S.S., Garrison, D.L., Horner, R.A., 1998. Bloom Dynamics and Physiology of Domoic-Acid- Producing Pseudo-nitzschia Species. NATO ASI series G ecological sciences 40, 267–292.

Bednaršek, N., Klinger, T., Harvey, C.J., Weisberg, S., McCabe, R.M., Feely, R.A., Newton, J., Tolimieri, N., 2017. New ocean, new needs: Application of pteropod shell dissolution as a biological indicator for marine resource management. Ecological Indicators 76, 240–244. <https://doi.org/10.1016/j.ecolind.2017.01.025>

Bednaršek, N., Tarling, G.A., Bakker, D.C.E., Fielding, S., Feely, R.A., 2014. Dissolution Dominating Calcification Process in Polar Pteropods Close to the Point of Aragonite Undersaturation. PLOS ONE 9, e109183. <https://doi.org/10.1371/journal.pone.0109183>

Berg, M., Sutula, M., 2015. Factors Affecting Growth of Cyanobacteria with Special Emphasis on the Sacramento-San Joaquin Delta (Technical Report No. 869). Southern California Coastal Water Research Project, Costa Mesa, CA.

Biggs, B.J.F., Smith, R.A., 2002. Taxonomic richness of stream benthic algae: Effects of flood disturbance and nutrients. Limnology and Oceanography 47, 1175–1186. <https://doi.org/10.4319/lo.2002.47.4.1175>

Bittick, S.J., Sutula, M., Fong, P., 2018. A tale of two algal blooms: Negative and predictable effects of two common bloom-forming macroalgae on seagrass and epiphytes. Marine Environmental Research 140, 1–9. <https://doi.org/10.1016/j.marenvres.2018.05.018>

Borsuk, M.E., Stow, C.A., Luettich, R.A., Paerl, H.W., Pinckney, J.L., 2001. Modelling Oxygen Dynamics in an Intermittently Stratified Estuary: Estimation of Process Rates Using Field Data. Estuarine, Coastal and Shelf Science 52, 33–49. <https://doi.org/10.1006/ecss.2000.0726>

Boyer, K., Sutula, M., 2015. Factors Controlling Submersed and Floating Macrophytes in the Sacramento-San Joaquin Delta (Technical Report No. 870). Southern California Coastal Water Research Project, Costa Mesa, CA.

Breitburg, D.L., Craig, J.K., Fulford, R.S., Rose, K.A., Boynton, W.R., Brady, D.C., Ciotti, B.J., Diaz, R.J., Friedland, K.D., Hagy, J.D., Hart, D.R., Hines, A.H., Houde, E.D., Kolesar, S.E., Nixon, S.W., Rice, J.A., Secor, D.H., Targett, T.E., 2009. Nutrient enrichment and fisheries exploitation: interactive effects on estuarine living resources and their management. Hydrobiologia 629, 31–47. <https://doi.org/10.1007/s10750-009-9762-4>

Breitburg, D.L., Loher, T., Pacey, C.A., Gerstein, A., 1997. Varying Effects of Low Dissolved Oxygen on Trophic Interactions in an Estuarine Food Web. Ecological Monographs 67, 489–507. [https://doi.org/10.1890/0012-9615(1997)067[0489:VEOLDO]2.0.CO;2](https://doi.org/10.1890/0012-9615(1997)067%5b0489:VEOLDO%5d2.0.CO;2)

Burkholder, J.M., Mason, K.M., Glasgow, H.B., 1992. Water-column nitrate enrichment promotes decline of eelgrass Zostera marina: evidence from seasonal mesocosm experiments. Mar. Ecol. Prog. Ser. 16.

Butterwick, C., Heaney, S.I., Talling, J.F., 2005. Diversity in the influence of temperature on the growth rates of freshwater algae, and its ecological relevance. Freshwater Biology 50, 291–300. <https://doi.org/10.1111/j.1365-2427.2004.01317.x>

Carey, C.C., Ibelings, B.W., Hoffmann, E.P., Hamilton, D.P., Brookes, J.D., 2012. Eco-physiological adaptations that favour freshwater cyanobacteria in a changing climate. Water Research, Cyanobacteria: Impacts of climate change on occurrence, toxicity and water quality management 46, 1394–1407. <https://doi.org/10.1016/j.watres.2011.12.016>

CCHAB: Blue Green Algae Work Group of the State Water Resources Control Board (SWRCB), the California Department of Public Health (CDPH), and Office of Environmental Health and Hazard Assessment (OEHHA), 2016. Cyanobacteria in California Recreational Water Bodies: Providing Voluntary Guidance about Harmful Algal Blooms, Their Monitoring, and Public Notification (Draft Report).

Chessman, B.C., Hutton, P.E., Burch, J.M., 1992. Limiting nutrients for periphyton growth in sub-alpine, forest, agricultural and urban streams. Freshwater Biology 28, 349–361. <https://doi.org/10.1111/j.1365-2427.1992.tb00593.x>

Chételat, J., Pick, F.R., Morin, A., Hamilton, P.B., 1999. Periphyton biomass and community composition in rivers of different nutrient status. Canadian Journal of Fisheries and Aquatic Sciences 56, 560–569.

Cloern, J., 2001. Our evolving conceptual model of the coastal eutrophication problem. Marine Ecology Progress Series 210, 223–253. <https://doi.org/10.3354/meps210223>

Conley, D.J., Paerl, H.W., Howarth, R.W., Boesch, D.F., Seitzinger, S.P., Havens, K.E., Lancelot, C., Likens, G.E., 2009. ECOLOGY: Controlling Eutrophication: Nitrogen and Phosphorus. Science 323, 1014–1015. <https://doi.org/10.1126/science.1167755>

Cormier, S.M., Norton, S.B., Suter II, G.W., Reed-Judkins, D., 2000. Stressor Identification Guidance Document (No. EPA/822/B-00/025). U.S. Environmental Protection Agency, Office of Research and Development, Washington, D.C.

Cushing, C.E., Allan, J.D., 2001. Streams their ecology and life. San Diego Academic Press.

Dahm, C.N., Parker, A.E., Adelson, A.E., Christman, M.A., Bergamaschi, B.A., 2016. Nutrient Dynamics of the Delta: Effects on Primary Producers. San Francisco Estuary and Watershed Science 14.

D’Avanzo, C., Kremer, J.N., 1994. Diel Oxygen Dynamics and Anoxic Events in an Eutrophic Estuary of Waquoit Bay, Massachusetts. Estuaries 17, 131. <https://doi.org/10.2307/1352562>

Delong, M.D., Brusven, M.A., 1992. PATTERNS OF PERIPHYTON CHLOROPHYLL α IN AN AGRICULTURAL NONPOINT SOURCE IMPACTED STREAM1. JAWRA Journal of the American Water Resources Association 28, 731–741. <https://doi.org/10.1111/j.1752-1688.1992.tb01495.x>

Dennison, W.C., Orth, R.J., Moore, K.A., Stevenson, J.C., Carter, V., Kollar, S., Bergstrom, P.W., Batiuk, R.A., 1993. Assessing Water Quality with Submersed Aquatic Vegetation. BioScience 43, 86–94. <https://doi.org/10.2307/1311969>

Dettmann, E.H., 2001. Effect of water residence time on annual export and denitrification of nitrogen in estuaries: A model analysis. Estuaries 24, 481–490. <https://doi.org/10.2307/1353250>

Diaz, R.J., 2001. Overview of Hypoxia around the World. Journal of Environmental Quality 30, 275–281. <https://doi.org/10.2134/jeq2001.302275x>

Diaz, R.J., Rosenberg, R., 2008. Spreading Dead Zones and Consequences for Marine Ecosystems. Science 321, 926–929. <https://doi.org/10.1126/science.1156401>

Diaz, R.J., Rosenberg, R., 1995. MARINE BENTHIC HYPOXIA: A REVIEW OF ITS ECOLOGICAL EFFECTS AND THE BEHAVIOURAL RESPONSES OF BENTHIC MACROFAUNA. Oceanography and Marine Biology - Annual Review 33, 245–303.

Dodds, W.K., 2007. Trophic state, eutrophication and nutrient criteria in streams. Trends in Ecology & Evolution 22, 669–676. <https://doi.org/10.1016/j.tree.2007.07.010>

Dodds, W.K., Biggs, B.J.F., 2002. Water Velocity Attenuation by Stream Periphyton and Macrophytes in Relation to Growth Form and Architecture. Journal of the North American Benthological Society 21, 2–15. <https://doi.org/10.2307/1468295>

Domagalski, J., Saleh, D., 2015. Sources and Transport of Phosphorus to Rivers in California and Adjacent States, U.S., as Determined by SPARROW Modeling. JAWRA Journal of the American Water Resources Association 51, 1463–1486. <https://doi.org/10.1111/1752-1688.12326>

Dortch, Q., Robichaux, R., Pool, S., Milsted, D., Mire, G., Rabalais, N., Soniat, T., Fryxell, G., Turner, R., Parsons, M., 1997. Abundance and vertical flux of Pseudo-nitzschia in the northern Gulf of Mexico. Marine Ecology Progress Series 146, 249–264. <https://doi.org/10.3354/meps146249>

Duarte, R., Mainar, A., Sánchez-Chóliz, J., 2013. The role of consumption patterns, demand and technological factors on the recent evolution of CO2 emissions in a group of advanced economies. Ecological Economics 96, 1–13. <https://doi.org/10.1016/j.ecolecon.2013.09.007>

Edmondson, W.T., Lehman, J.T., 1981. The effect of changes in the nutrient income on the condition of Lake Washington1. Limnology and Oceanography 26, 1–29. <https://doi.org/10.4319/lo.1981.26.1.0001>

Elmgren, R., Larsson, U., 2001. Nitrogen and the Baltic Sea: Managing Nitrogen in Relation to Phosphorus [WWW Document]. The Scientific World Journal. <https://doi.org/10.1100/tsw.2001.291>

Fan, L., Brett, M.T., Jiang, W., Li, B., 2017. Dissolved organic nitrogen recalcitrance and bioavailable nitrogen quantification for effluents from advanced nitrogen removal wastewater treatment facilities. Environ. Pollut. 229, 255–263. <https://doi.org/10.1016/j.envpol.2017.05.093>

Federal Geographic Data Committee (FGDC), 2013. Classification of Wetlands and Deepwater Habitats of the United States (No. FGDC-STD-004-2013. Second Edition). Wetlands Subcommittee, Federal Geographic Data Committee and U.S. Fish and Wildlife Service, Washington, DC. <https://doi.org/10.1002/047147844X.sw2162>

Feely, R.A., Sabine, C.L., Hernandez-Ayon, J.M., Ianson, D., Hales, B., 2008. Evidence for Upwelling of Corrosive “Acidified” Water onto the Continental Shelf. Science 320, 1490–1492. <https://doi.org/10.1126/science.1155676>

Fehling, J., Davidson, K., Bolch, C.J., Bates, S.S., 2004. Growth and Domoic Acid Production by Pseudo-Nitzschia Seriata (bacillariophyceae) Under Phosphate and Silicate Limitation1. Journal of Phycology 40, 674–683. <https://doi.org/10.1111/j.1529-8817.2004.03213.x>

Fetcho, K., 2007. Klamath River Blue-Green Algae Summary Report. Yurok Tribe: Klamath, CA, USA.

Fetscher, A.E., Howard, M.D.A., Stancheva, R., Kudela, R.M., Stein, E.D., Sutula, M.A., Busse, L.B., Sheath, R.G., 2015. Wadeable streams as widespread sources of benthic cyanotoxins in California, USA. Harmful Algae 49, 105–116. <https://doi.org/10.1016/j.hal.2015.09.002>

Fetscher, A.E., Sutula, M.A., Sengupta, A., 2014. Linking Nutrients to Alterations in Aquatic Life in California Wadable Streams (Report for USEPA No. 0834). Southern California Coastal Water Research Project, Costa Mesa, CA.

Fevold, K.L., 1998. Sub-surface controls on the distribution of benthic algae in floodplain back channel habitats of the Queets Rive (Master’s thesis). University of Washington, Seattle.

Findlay, S., 2010. Stream microbial ecology. Journal of the North American Benthological Society 29, 170–181. <https://doi.org/10.1899/09-023.1>

Fisher, W.S., Malone, T.C., Giattina, J.D., 2003. A Pilot Project to Detect and Forecast Harmful Algal Blooms in the Northern Gulf of Mexico, in: Melzian, B.D., Engle, V., McAlister, M., Sandhu, S., Eads, L.K. (Eds.), Coastal Monitoring through Partnerships: Proceedings of the Fifth Symposium on the Environmental Monitoring and Assessment Program (EMAP) Pensacola Beach, FL, U.S.A., April 24–27, 2001. Springer Netherlands, Dordrecht, pp. 373–381. <https://doi.org/10.1007/978-94-017-0299-7_31>

Fong, P., Zedler, J.B., Donohoe, R.M., 1993. Nitrogen vs. phosphorus limitation of algal biomass in shallow coastal lagoons. Limnology and Oceanography 38, 906–923. <https://doi.org/10.4319/lo.1993.38.5.0906>

Fritz, L., Quilliam, M.A., Wright, J.L.C., Beale, A.M., Work, T.M., 1992. An Outbreak of Domoic Acid Poisoning Attributed to the Pennate Diatom Pseudonitzschia Australis1. Journal of Phycology 28, 439–442. <https://doi.org/10.1111/j.0022-3646.1992.00439.x>

Gibble, C.M., Kudela, R.M., 2014. Detection of persistent microcystin toxins at the land–sea interface in Monterey Bay, California. Harmful Algae 39, 146–153. <https://doi.org/10.1016/j.hal.2014.07.004>

Gillett, D.J., Weisberg, S.B., Grayson, T., Hamilton, A., Hansen, V., Leppo, E.W., Pelletier, M.C., Borja, A., Cadien, D., Dauer, D., Diaz, R., Dutch, M., Hyland, J.L., Kellogg, M., Larsen, P.F., Levinton, J.S., Llansó, R., Lovell, L.L., Montagna, P.A., Pasko, D., Phillips, C.A., Rakocinski, C., Ranasinghe, J.A., Sanger, D.M., Teixeira, H., Dolah, R.F.V., Velarde, R.G., Welch, K.I., 2015. Effect of ecological group classification schemes on performance of the AMBI benthic index in US coastal waters. Ecological Indicators 50, 99–107. <https://doi.org/10.1016/j.ecolind.2014.11.005>

Glasgow, H.B., Burkholder, J.M., 2000. Water Quality Trends and Management Implications from a Five-Year Study of a Eutrophic Estuary. Ecological Applications 10, 1024–1046. [https://doi.org/10.1890/1051-0761(2000)010[1024:WQTAMI]2.0.CO;2](https://doi.org/10.1890/1051-0761(2000)010%5b1024:WQTAMI%5d2.0.CO;2)

Graham, N.J.D., Wardlaw, V.E., Perry, R., Jiang, J.-Q., 1998. The significance of algae as trihalomethane precursors. Water Science and Technology, Reservoir Management and Water Supply — An Integrated System 37, 83–89. <https://doi.org/10.1016/S0273-1223(98)00013-4>

Gray, J.S., Wu, R.S., Or, Y.Y., 2002. Effects of hypoxia and organic enrichment on the coastal marine environment. Marine Ecology Progress Series 238, 249–279. <https://doi.org/10.3354/meps238249>

Green, L., Sutula, M., Fong, P., 2014. How much is too much? Identifying benchmarks of adverse effects of macroalgae on the macrofauna in intertidal flats. Ecological Applications 24, 300–314. <https://doi.org/10.1890/13-0524.1>

Harper, D.M., 1992. Eutrophication of freshwaters. Chapman & Hall, London.

Harter, T., Lund, J.R., Darby, J., Fogg, G.E., Howitt, R., Jessoe, K.K., Pettygrove, G.S., Quinn, J.F., Viers, J.H., Boyle, D.B., Canada, H.E., Dzurella, K.N., Fryjoff, A., Hollander, A.D., Honeycutt, K.L., Jenkins, M.W., Jensen, V.B., Kourakos, G., Liptzin, D., Mayzelle, M.M., McNally, A., Rosenstock, T.S., Lawrence, C., Dolan, D.V., 2012. Addressing Nitrate in California’s Drinking Water with a Focus on Tulare Lake Basin and Salinas Valley Groundwater. Report for the State Water Resources Control Board Report to the Legislature. Center for Watershed Sciences, University of California, Davis, Davis, CA.

Hawkins, C.P., Murphy, M.L., Anderson, N.H., 1982. Effects of Canopy, Substrate Composition, and Gradient on the Structure of Macroinvertebrate Communities in Cascade Range Streams of Oregon. Ecology 63, 1840–1856. <https://doi.org/10.2307/1940125>

Heiskary, S., Markus, H., 2001. Establishing Relationships Among Nutrient Concentrations, Phytoplankton Abundance, and Biochemical Oxygen Demand in Minnesota, USA, Rivers. Lake and Reservoir Management 17, 251–262. <https://doi.org/10.1080/07438140109354134>

Howard, M.D.A., Cochlan, W.P., Ladizinsky, N., Kudela, R.M., 2007. Nitrogenous preference of toxigenic Pseudo-nitzschia australis (Bacillariophyceae) from field and laboratory experiments. Harmful Algae 6, 206–217. <https://doi.org/10.1016/j.hal.2006.06.003>

Howard, M.D.A., Sutula, M., Caron, D.A., Chao, Y., Farrara, J.D., Frenzel, H., Jones, B., Robertson, G., McLaughlin, K., Sengupta, A., 2014. Anthropogenic nutrient sources rival natural sources on small scales in the coastal waters of the Southern California Bight. Limnology and Oceanography 59, 285–297. <https://doi.org/10.4319/lo.2014.59.1.0285>

Howarth, R.W., 1988. Nutrient Limitation of Net Primary Production in Marine Ecosystems. Annual Review of Ecology and Systematics 19, 89–110. <https://doi.org/10.1146/annurev.es.19.110188.000513>

Howarth, R.W., Sharpley, A., Walker, D., 2002. Sources of nutrient pollution to coastal waters in the United States: Implications for achieving coastal water quality goals. Estuaries 25, 656–676. <https://doi.org/10.1007/BF02804898>

Hughes, B., Haskins, J., Wasson, K., Watson, E., 2011. Identifying factors that influence expression of eutrophication in a central California estuary. Marine Ecology Progress Series 439, 31–43. <https://doi.org/10.3354/meps09295>

Hynes, H.B.N., 1970. Eutrophication: Causes, Consequences, Correctives. National Academy of Sciences, Washington, D.C. <https://doi.org/10.17226/20256>

Jacoby, J.M., Kann, J., 2007. The occurrence and response to toxic cyanobacteria in the Pacific Northwest, North America. Lake and Reservoir Management 23, 123–143. <https://doi.org/10.1080/07438140709353916>

Jassby, A., 2008. Phytoplankton in the Upper San Francisco Estuary: Recent Biomass Trends, Their Causes, and Their Trophic Significance. San Francisco Estuary and Watershed Science 6.

Jeppesen, R., Rodriguez, M., Rinde, J., Haskins, J., Hughes, B., Mehner, L., Wasson, K., 2018. Effects of Hypoxia on Fish Survival and Oyster Growth in a Highly Eutrophic Estuary. Estuaries and Coasts 41, 89–98. <https://doi.org/10.1007/s12237-016-0169-y>

Kaiser, J., 1999. Battle Over a Dying Sea. Science 284, 28–30. <https://doi.org/10.1126/science.284.5411.28>

Kann, J., 2008. Microcystin Bioaccumulation in Klamath River Fish and Freshwater Mussel Tissue: Preliminary 2007 Results (Technical Memorandum). Aquatic Ecosystem Sciences LLC.

Kann, J., 2004. Copco Lake Analysis (To: Kier and Associates) (Memo). Aquatic Ecosystem Sciences LLC, Ashland, OR.

Kaplanis, N.J., Harris, J., Smith, J., 2016. Distribution patterns of the non-native seaweeds Sargassum horneri (Turner) C. Agardh and Undaria pinnatifida (Harvey) Suringar on the San Diego and Pacific coast of North America. Aquatic Invasions 11, 111–124. <https://doi.org/10.3391/ai.2016.11.2.01>

Karlson, K., 2002. Temporal and spatial large-scale effects of eutrophication and oxygen deficiency on benthic fauna in Scandinavian and Baltic waters-a review. Oceanogr. Mar. Biol. Ann. Rev. 40, 427–489.

Karr, J.R., Chu, E.W., 1999. Restoring Life in Running Waters: Better Biological Monitoring, 2nd ed. edition. ed. Island Press, Washington, D.C.

Kennison, R., Kamer, K., Fong, P., 2003. Nutrient dynamics and macroalgal blooms: A Comparison of five southern California estuaries (No. 416). Southern California Coastal Water Research Project.

Kudela, R.M., 2011. Characterization and deployment of Solid Phase Adsorption Toxin Tracking (SPATT) resin for monitoring of microcystins in fresh and saltwater. Harmful Algae 11, 117–125. <https://doi.org/10.1016/j.hal.2011.08.006>

Kudela, R.M., Banas, N.S., Barth, J.A., Frame, E.R., Jay, D.A., Largier, J.L., Lessard, E.J., Peterson, T.D., Vander Woude, A.J., 2008a. New insights into the controls and mechanisms of plankton productivity in coastal upwelling waters of the northern California Current System. Oceanography 21, 46–59.

Kudela, R.M., Lane, J.Q., Cochlan, W.P., 2008b. The potential role of anthropogenically derived nitrogen in the growth of harmful algae in California, USA. Harmful Algae, HABs and Eutrophication 8, 103–110. <https://doi.org/10.1016/j.hal.2008.08.019>

Lefebvre, K.A., Powell, C.L., Busman, M., Doucette, G.J., Moeller, P.D.R., Silver, J.B., Miller, P.E., Hughes, M.P., Singaram, S., Silver, M.W., Tjeerdema, R.S., 1999. Detection of domoic acid in northern anchovies and california sea lions associated with an unusual mortality event. Natural Toxins 7, 85–92. [https://doi.org/10.1002/(SICI)1522-7189(199905/06)7:3<85::AID-NT39>3.0.CO;2-Q](https://doi.org/10.1002/(SICI)1522-7189(199905/06)7:3%3c85::AID-NT39%3e3.0.CO;2-Q)

Lewitus, A.J., Horner, R.A., Caron, D.A., Garcia-Mendoza, E., Hickey, B.M., Hunter, M., Huppert, D.D., Kudela, R.M., Langlois, G.W., Largier, J.L., Lessard, E.J., RaLonde, R., Jack Rensel, J.E., Strutton, P.G., Trainer, V.L., Tweddle, J.F., 2012. Harmful algal blooms along the North American west coast region: History, trends, causes, and impacts. Harmful Algae 19, 133–159. <https://doi.org/10.1016/j.hal.2012.06.009>

Likens, G.E., 1972. Eutrophication and aquatic ecosystems, in: Nutrients and Eutrophication: The Limiting-Nutrient Controversy, Limnology and Oceanography Special Symposium. pp. 3–13.

Lucas, L.V., Cloern, J.E., Thompson, J.K., Stacey, M.T., Koseff, J.R., 2016. Bivalve Grazing Can Shape Phytoplankton Communities. Front. Mar. Sci. 3. <https://doi.org/10.3389/fmars.2016.00014>

Lürling, M., Roessink, I., 2006. On the way to cyanobacterial blooms: Impact of the herbicide metribuzin on the competition between a green alga (Scenedesmus) and a cyanobacterium (Microcystis). Chemosphere 65, 618–626. <https://doi.org/10.1016/j.chemosphere.2006.01.073>

MacIntyre, H.L., Geider, R.J., Miller, D.C., 1996. Microphytobenthos: The Ecological Role of the “Secret Garden” of Unvegetated, Shallow-Water Marine Habitats. I. Distribution, Abundance and Primary Production. Estuaries 19, 186. <https://doi.org/10.2307/1352224>

Magrann, T., Meredith D. A., H., Martha, S., Danilo S., B., William K., H., Stephen G., D., 2015. Screening Assessment of Cyanobacteria and Cyanotoxins in Southern California Lentic Habitats. Environmental Management and Sustainable Development 4, 91. <https://doi.org/10.5296/emsd.v4i2.8036>

Mallin, M.A., Johnson, V.L., Ensign, S.H., MacPherson, T.A., 2006. Factors contributing to hypoxia in rivers, lakes, and streams. Limnology and Oceanography 51, 690–701. <https://doi.org/10.4319/lo.2006.51.1_part_2.0690>

Mazor, R.D., Beck, M., Brown, J., 2018. Report on the Stormwater Monitoring Coalition Regional Stream Survey (Technical Report No. 1029). Southern California Coastal Water Research Project, Costa Mesa, CA.

Mazor, R.D., Rehn, A.C., Ode, P.R., Engeln, M., Schiff, K.C., Stein, E.D., Gillett, D.J., Herbst, D.B., Hawkins, C.P., 2016. Bioassessment in complex environments: designing an index for consistent meaning in different settings. Freshwater Science 35, 249–271. <https://doi.org/10.1086/684130>

Mazor, R.D., Sutula M., Theroux S., Beck M., and Ode. In prep. Eutrophication indicator thresholds protective of biological integrity in California wadeable streams. For submission to Freshwater Science.

McCabe, R.M., Hickey, B.M., Kudela, R.M., Lefebvre, K.A., Adams, N.G., Bill, B.D., Gulland, F.M.D., Thomson, R.E., Cochlan, W.P., Trainer, V.L., 2016. An unprecedented coastwide toxic algal bloom linked to anomalous ocean conditions. Geophysical Research Letters 43, 10,366-10,376. <https://doi.org/10.1002/2016GL070023>

McGlathery, K.J., 2001. Macroalgal Blooms Contribute to the Decline of Seagrass in Nutrient-Enriched Coastal Waters. Journal of Phycology 37, 453–456. <https://doi.org/10.1046/j.1529-8817.2001.037004453.x>

McLaughlin, K., Sutula, M., Busse, L., Anderson, S., Crooks, J., Dagit, R., Gibson, D., Johnston, K., Stratton, L., 2014. A Regional Survey of the Extent and Magnitude of Eutrophication in Mediterranean Estuaries of Southern California, USA. Estuaries and Coasts 37, 259–278. <https://doi.org/10.1007/s12237-013-9670-8>

McLaughlin, K., Sutula, M.A., Cable, J., Fong, P., 2013. Eutrophication and Nutreint Cycling in Santa Margarita River Estuary: A Summary of Baseline Studies for Monitoring Order R9-2006-0076 (Technical Report No. 635). Southern California Coastal Water Research Project, Costa Mesa, CA.

McLaughlin, K., Sutula, M.A., Cable, J., Fong, P., 2010. Eutrophication and Nutrient Cycling in Buena Vista Lagoon: A Summary of Baseline Studies for Monitoring Order R9-2006-0076 (Technical Report No. 638). Southern California Coastal Water Research Project, Costa Mesa, CA.

Meyer-Reil, L.-A., Köster, M., 2000. Eutrophication of Marine Waters: Effects on Benthic Microbial Communities. Marine Pollution Bulletin, Seas at the Millennium: an Environmental Evaluation 41, 255–263. <https://doi.org/10.1016/S0025-326X(00)00114-4>

Middelburg, J.J., Levin, L.A., 2009. Coastal hypoxia and sediment biogeochemistry 21.

Miller, M.A., Kudela, R.M., Mekebri, A., Crane, D., Oates, S.C., Tinker, M.T., Staedler, M., Miller, W.A., Toy-Choutka, S., Dominik, C., Hardin, D., Langlois, G., Murray, M., Ward, K., Jessup, D.A., 2010. Evidence for a Novel Marine Harmful Algal Bloom: Cyanotoxin (Microcystin) Transfer from Land to Sea Otters. PLOS ONE 5, e12576. <https://doi.org/10.1371/journal.pone.0012576>

Miltner, R.J., Rankin, A.E.T., 1998. Primary nutrients and the biotic integrity of rivers and streams. Freshwater Biology 40, 145–158. <https://doi.org/10.1046/j.1365-2427.1998.00324.x>

Moisander, P.H., Ochiai, M., Lincoff, A., 2009. Nutrient limitation of Microcystis aeruginosa in northern California Klamath River reservoirs. Harmful Algae 8, 889–897. <https://doi.org/10.1016/j.hal.2009.04.005>

Nixon, S.W., 1995. Coastal marine eutrophication: A definition, social causes, and future concerns. Ophelia 41, 199–219. <https://doi.org/10.1080/00785236.1995.10422044>

Nixon, S.W., 1986. Nutrient dynamics and the productivity of marine coastal waters, in: Marine Environment and Pollution. The Alden Press, Oxford, UK, pp. 97–115.

North Coast Regional Water Quality Control Board, 2010. Klamath river total maximum daily loads (TMDLs) addressing temperature, dissolved oxygen, nutrient, and microcystin impairments in California the proposed site specific dissolved oxygen objectives for the Klamath River in California, and the Klamath River and Lost River implementation plans (Final Staff Report). Santa Rosa, CA.

Norton, S.B., Cormier, S.M., Suter, G.W., Schofield, K., Yuan, L., Shaw-Allen, P., Ziegler, C.R., 2009. CADDIS: The Causal Analysis/Diagnosis Decision Information System, in: Marcomini, A., Suter II, G.W., Critto, A. (Eds.), Decision Support Systems for Risk-Based Management of Contaminated Sites. Springer US, Boston, MA, pp. 1–24. <https://doi.org/10.1007/978-0-387-09722-0_17>

OSPAR Commission, 2003. OSPAR integrated report 2003 on the eutrophication status of the OSPAR maritime area based upon the first application of the comprehensive procedure. OSPAR Commission.

Paul M.J., Jessup B., Brown L., Carter C, Cantonati M., Charles D.F., Gerritsen J., Herbst D., Howard J., Isham B., Lowe R., Mazor R., Mendez P., O’Dowd A., Olson J., Pan Y., Rehn A., Spaulding S., Sutula M., Stancheva Hristova R., and Theroux S. In Prep. Development of benthic macroinvertebrate and algal biological condition model for California streams. For submission for Freshwater Science.

Paerl, H.W., Hall, N.S., Calandrino, E.S., 2011. Controlling harmful cyanobacterial blooms in a world experiencing anthropogenic and climatic-induced change. Science of The Total Environment 409, 1739–1745. <https://doi.org/10.1016/j.scitotenv.2011.02.001>

Paerl, H.W., Hall, N.S., Peierls, B.L., Rossignol, K.L., 2014. Evolving Paradigms and Challenges in Estuarine and Coastal Eutrophication Dynamics in a Culturally and Climatically Stressed World. Estuaries and Coasts 37, 243–258. <https://doi.org/10.1007/s12237-014-9773-x>

Paerl, H.W., Huisman, J., 2008. Blooms like It Hot. Science 320, 57–58.

Paerl, H.W., Paul, V.J., 2012. Climate change: Links to global expansion of harmful cyanobacteria. Water Research 46, 1349–1363. <https://doi.org/10.1016/j.watres.2011.08.002>

Pan, Y., Parsons, M.L., Busman, M., Moeller, P.D.R., Dortch, Q., Powell, C.L., Doucette, G.J., 2001. Pseudo-nitzschia sp. cf. pseudodelicatissima ‹ a confirmed producer of domoic acid from the northern Gulf of Mexico. Marine Ecology Progress Series 220, 83–92. <https://doi.org/10.3354/meps220083>

Pan, Youlian, Rao, D.V.S., Mann, K.H., 1996. Changes in Domoic Acid Production and Cellular Chemical Composition of the Toxigenic Diatom Pseudo-Nitzschia Multiseries Under Phosphate Limitation1. Journal of Phycology 32, 371–381. <https://doi.org/10.1111/j.0022-3646.1996.00371.x>

Pan, Y., Stevenson, R.J., Hill, B.H., Herlihy, A.T., Collins, G.B., 1996. Using Diatoms as Indicators of Ecological Conditions in Lotic Systems: A Regional Assessment. Journal of the North American Benthological Society 15, 481–495. <https://doi.org/10.2307/1467800>

Pan, Y., Stevenson, R.J., Vaithiyanathan, P., Slate, J., Richardson, C.J., 2000. Changes in algal assemblages along observed and experimental phosphorus gradients in a subtropical wetland, U.S.A. Freshwater Biology 44, 339–353. <https://doi.org/10.1046/j.1365-2427.2000.00556.x>

Pan, Y., Subba Rao, D., Mann, K., Brown, R., Pocklington, R., 1996. Effects of silicate limitation on production of domoic acid, a neurotoxin, by the diatom Pseudo-nitzschia multiseries. I. Batch culture studies. Marine Ecology Progress Series 131, 225–233. <https://doi.org/10.3354/meps131225>

Peacock, M.B., Gibble, C.M., Senn, D.B., Cloern, J.E., Kudela, R.M., 2018. Blurred lines: Multiple freshwater and marine algal toxins at the land-sea interface of San Francisco Bay, California. Harmful Algae 73, 138–147. <https://doi.org/10.1016/j.hal.2018.02.005>

Pearson, T.H., Rosenberg, R., 1978. Macrobenthic Succession in Relation to Organic Enrichment and Pollution of the Marine Environment. Oceanography and Marine Biology - Annual Review 16, 229–311.

Peterson, H.G., Boutin, C., Freemark, K.E., Martin, P.A., 1997. Toxicity of hexazinone and diquat to green algae, diatoms, cyanobacteria and duckweed. Aquatic Toxicology 39, 111–134. <https://doi.org/10.1016/S0166-445X(97)00022-2>

Plummer, J.D., Edzwald, J.K., 2001. Effect of Ozone on Algae as Precursors for Trihalomethane and Haloacetic Acid Production. Environmental Science & Technology 35, 3661–3668. <https://doi.org/10.1021/es0106570>

Rabalais, N.N., Harper Jr., D.E., 1992. Studies of benthic biota in areas affected by moderate and severe hypoxia (Proceedings). Workshop on Nutrient Enhanced Coastal Ocean Program Office, Louisiana.

Ranasinghe, J.A., Welch, K.I., Slattery, P.N., Montagne, D.E., Huff, D.D., Ii, H.L., Hyland, J.L., Thompson, B., Weisberg, S.B., Oakden, J.M., Cadien, D.B., Velarde, R.G., 2012. Habitat-related benthic macrofaunal assemblages of bays and estuaries of the western United States. Integrated Environmental Assessment and Management 8, 638–648. <https://doi.org/10.1002/ieam.62>

Reisner, M., 1993. Cadillac Desert: The American West and Its Disappearing Water. Penguin.

Reynolds, C.S., 2006. The Ecology of Phytoplankton. Cambridge University Press.

Robarts, R.D., Zohary, T., 1987. Temperature effects on photosynthetic capacity, respiration, and growth rates of bloom‐forming cyanobacteria. New Zealand Journal of Marine and Freshwater Research 21, 391–399. <https://doi.org/10.1080/00288330.1987.9516235>

Ryther, J.H., Dunstan, W.M., 1971. Nitrogen, Phosphorus, and Eutrophication in the Coastal Marine Environment. Science 171, 1008–1013.

Saleh, D., Domagalski, J., 2015. SPARROW Modeling of Nitrogen Sources and Transport in Rivers and Streams of California and Adjacent States, U.S. JAWRA Journal of the American Water Resources Association 51, 1487–1507. <https://doi.org/10.1111/1752-1688.12325>

Schindler, D.W., 1977. Evolution of Phosphorus Limitation in Lakes. Science 195, 260–262.

Schindler, D.W., Hecky, R.E., Findlay, D.L., Stainton, M.P., Parker, B.R., Paterson, M.J., Beaty, K.G., Lyng, M., Kasian, S.E.M., 2008. Eutrophication of lakes cannot be controlled by reducing nitrogen input: Results of a 37-year whole-ecosystem experiment. PNAS 105, 11254–11258. <https://doi.org/10.1073/pnas.0805108105>

Scholin, C.A., Gulland, F., Doucette, G.J., Benson, S., Busman, M., Chavez, F.P., Cordaro, J., DeLong, R., Vogelaere, A.D., Harvey, J., Haulena, M., Lefebvre, K., Lipscomb, T., Loscutoff, S., Lowenstine, L.J., Iii, R.M., Miller, P.E., McLellan, W.A., Moeller, P.D.R., Powell, C.L., Rowles, T., Silvagni, P., Silver, M., Spraker, T., Trainer, V., Dolah, F.M.V., 2000. Mortality of sea lions along the central California coast linked to a toxic diatom bloom. Nature 403, 80–84. <https://doi.org/10.1038/47481>

Senn, D., 2013.

Smith, A.J., 1983. Modes of cyanobacterial carbon metabolism. Annales de l’Institut Pasteur / Microbiologie 134, 93–113. <https://doi.org/10.1016/S0769-2609(83)80099-4>

Smith, R.A., Alexander, R.B., Wolman, M.G., 1987. Water-Quality Trends in the Nation’s Rivers. Science 235, 1607–1615. <https://doi.org/10.1126/science.235.4796.1607>

Smith, V.H., 1983. Low Nitrogen to Phosphorus Ratios Favor Dominance by Blue-Green Algae in Lake Phytoplankton. Science 221, 669–671. <https://doi.org/10.1126/science.221.4611.669>

Stanley, D.W., Nixon, S.W., 1992. Stratification and bottom-water hypoxia in the Pamlico River estuary. Estuaries 15, 270–281. <https://doi.org/10.2307/1352775>

State Water Resources Control Board, Regional Water Quality Control Board, 1989. Water Quality Control Plan - Central Coast Basin.

State Water Resources Control Board, 2010. Biointegrity Workplan.

State Water Resources Control Board (SWRCB), 2014. Nutrient Control Plan.

State Water Resources Control Board, 2016. Focus group outreach document.

Stevenson, R.J., Rier, S.T., Riseng, C.M., Schultz, R.E., Wiley, M.J., 2006. Comparing Effects of Nutrients on Algal Biomass in Streams in Two Regions with Different Disturbance Regimes and with Applications for Developing Nutrient Criteria. Hydrobiologia 561, 149–165. <https://doi.org/10.1007/s10750-005-1611-5>

Suikkanen, S., Laamanen, M., Huttunen, M., 2007. Long-term changes in summer phytoplankton communities of the open northern Baltic Sea. Estuarine, Coastal and Shelf Science 71, 580–592. <https://doi.org/10.1016/j.ecss.2006.09.004>

Suter II, G.W., 1999. Developing Conceptual Models for Complex Ecological Risk Assessments. Human and Ecological Risk Assessment: An International Journal 5, 375–396. <https://doi.org/10.1080/10807039991289491>

Sutula, M., 2011. Review of Indicators for Development of Nutrient Numeric Endpoints in California Estuaries (Technical Report No. 646). Southern California Coastal Water Research Project, Costa Mesa, CA.

Sutula, M., Green, L., Cicchetti, G., Detenbeck, N., Fong, P., 2014. Thresholds of Adverse Effects of Macroalgal Abundance and Sediment Organic Matter on Benthic Habitat Quality in Estuarine Intertidal Flats. Estuaries and Coasts 37, 1532–1548. <https://doi.org/10.1007/s12237-014-9796-3>

Sutula, M., Kamer, K., Cable, J., 2004. Sediments as a non-point source of nutrients to Malibu Lagoon, California (USA) (Technical Report No. 441). Southern California Coastal Water Research Project, Costa Mesa, CA.

Sutula, M., Kamer, K., Cable, J., Collis, H., Berelson, W., Mendez, J., 2006. Sediments as an internal source of nutrients to Upper Newport Bay, California (Technical Report No. 482). Southern California Coastal Water Research Project, Costa Mesa, CA.

Sutula, M.A., Bailey, H., Poucher, S., 2012. Science Supporting Dissolved Oxygen Objectives in California Estuaries (Technical Report No. 684). Southern California Coastal Water Research Project, Costa Mesa, CA.

Sutula, M.A., Butcher, J., Boschen, C., Molina, M., 2016a. Application of watershed loading and estuary water quality models to inform nutrient management in the Santa Margarita River Watershed (933). Southern California Coastal Water Research Project, Costa Mesa, CA.

Sutula, M.A., Gillett, D.J., Jones, A., 2016b. Status of Eutrophication in San Elijo Lagoon and its Relevance for Restoration (Technical Report No. 938). Southern California Coastal Water Research Project, Costa Mesa, CA.

Sutula, M.A., Mazor, R.D., Theroux, S., In Prep. Scientific Bases for Assessment, Prevention, and Management of Biostimulatory Impacts in california Wadeable Streams (Technical Report No. 1048). Southern California Coastal Water Research Project, Costa Mesa, CA.

Sutula, M.A., Senn, D., 2017. Scientific Basis to Assess the Effects of Nutrients on San Francisco Bay Beneficial Uses (Technical Report No. 864). Southern California Coastal Water Research Project, Costa Mesa, CA.

Sutula, M.A., Shultz, D., 2018. Status of eutrophication in the Upper (2016-2017) and Lower Main Stem (2015-2016) of the Santa Margarita River, San Diego County, California. Southern California Coastal Water Research Project, Costa Mesa, CA.

Tetra Tech, 2006. Technical approach to develop nutrients numeric endpoints for California (No. Contract No. 68-C-02-108-To-111). U.S. EPA Region IX.

Theroux, S., 2018. Workplan for SWAMP’s Algae Program. State Water Resources Control Board Surface Water Ambient Monitoring Program.

Theroux, S., Mazor, R.D., Ode, P., Sutula, M.A., Stein, E.D., n.d. A statewide algal bioassessment index for statewide wadable streams. In prep for submission to Ecological Indicators. In prep for submission to Ecological Indicators.

Trainer, V.L., Adams, N.G., Bill, B.D., Stehr, C.M., Wekell, J.C., Moeller, P., Busman, M., Woodruff, D., 2000. Domoic acid production near California coastal upwelling zones, June 1998. Limnology and Oceanography 45, 1818–1833. <https://doi.org/10.4319/lo.2000.45.8.1818>

Trainer, V.L., Bates, S.S., Lundholm, N., Thessen, A.E., Cochlan, W.P., Adams, N.G., Trick, C.G., 2012. Pseudo-nitzschia physiological ecology, phylogeny, toxicity, monitoring and impacts on ecosystem health. Harmful Algae, Harmful Algae--The requirement for species-specific information 14, 271–300. <https://doi.org/10.1016/j.hal.2011.10.025>

Trainer, V.L., Hickey, B.M., Horner, R.A., 2002. Biological and physical dynamics of domoic acid production off the Washington coast. Limnology and Oceanography 47, 1438–1446. <https://doi.org/10.4319/lo.2002.47.5.1438>

Trowbridge, P., Shimabuki, I., Bresnahan, P., Wheeler, S., Knight, E., Nielsen, K., Largier, J., Sutula, M., Valiela, L., Nutters, H., 2016. Summary of Workshop on Monitoring for Acidi ication Threats in West Coast Estuaries: A San Francisco Bay Case Study. San Francisco Estuary Institute, Richmond, CA.

Turner, R.K., Lorenzoni, I., Beaumont, N., Bateman, I.J., Langford, I.H., McDonald, A.L., 1998. Coastal Management for Sustainable Development: Analysing Environmental and Socio-Economic Changes on the UK Coast. The Geographical Journal 164, 269–281. <https://doi.org/10.2307/3060616>

Twilley, R., Kemp, W., Staver, K., Stevenson, J., Boynton, W., 1985. Nutrient enrichment of estuarine submersed vascular plant communities. 1. Algal growth and effects on production of plants and associated communities. Marine Ecology Progress Series 23, 179–191. <https://doi.org/10.3354/meps023179>

USEPA, 2010. Using Stressor-response Relationships to Derive Numeric Nutrient Criteria (No. EPA-820-S-10-001). U.S. Environmental Protection Agency, Office of Water, Office of Science and Technology, Washington, DC.

USEPA, 2007. Technical Approach To Develop Nutrient Numeric Endpoints For California Estuaries.

USEPA, 2001. Nutrient Criteria Technical Guidance Manual: Estuarine and Coastal Marine Waters (No. EPA-822-B-01-003). Washington, D.C.

USEPA, 1998. Guidelines for Ecological Risk Assessment (No. EPA/630/R-95/002F). U.S. Environmental Protection Agency.

USEPA, 1993. Guidance Specifying Management Measures For Sources of Nonpoint Pollution in Coastal Waters (No. 840- B-92– 002). U.S. Environmental Protection Agency, Office of Water, Washington, D.C.

USEPA, 1990. Biological Criteria National Program Guidance for Surface Waters (No. EPA-440/5-90-004). United States Environmental Protection Agency, Office of Water, Washington, D.C.

Valiela, I., Foreman, K., LaMontagne, M., Hersh, D., Costa, J., Peckol, P., DeMeo-Andreson, B., D’Avanzo, C., Babione, M., Sham, C.-H., Brawley, J., Lajtha, K., 1992. Couplings of Watersheds and Coastal Waters: Sources and Consequences of Nutrient Enrichment in Waquoit Bay, Massachusetts. Estuaries 15, 443. <https://doi.org/10.2307/1352389>

Valiela, I., McClelland, J., Hauxwell, J., Behr, P.J., Hersh, D., Foreman, K., 1997. Macroalgal blooms in shallow estuaries: Controls and ecophysiological and ecosystem consequences. Limnology and Oceanography 42, 1105–1118. <https://doi.org/10.4319/lo.1997.42.5_part_2.1105>

Van Dolah, F.M., Doucette, G.J., Gulland, F.M.D., Rowles, T., Bossart, G., 2003. 10 Impacts of algal toxins on marine mammals., in: Toxicology of Marine Mammals Chapter 10. CRC Press.

Vaquer-Sunyer, R., Duarte, C.M., 2008. Thresholds of hypoxia for marine biodiversity. PNAS 105, 15452–15457. <https://doi.org/10.1073/pnas.0803833105>

Viaroli, P., Bartoli, M., Giordani, G., Naldi, M., Orfanidis, S., Zaldivar, J.M., 2008. Community shifts, alternative stable states, biogeochemical controls and feedbacks in eutrophic coastal lagoons: a brief overview. Aquatic Conservation: Marine and Freshwater Ecosystems 18, S105–S117. <https://doi.org/10.1002/aqc.956>

Viers, J.H., Liptzin, D., Rosenstock, T.S., Jensen, V.B., Hollander, A.D., King, A.M., Kourakos, G., Lopez, E.M., Dzurella, K.N., Canada, H., Laybourne, S., McKenney, C., Darby, J., Quinn, J.F., Harter, T., 2012. Nitrogen Sources and Loading to Groundwater (Report for the State Water Resources Control Board Report to the Legislature No. Technical Report 2), Addressing Nitrate in California’s Drinking Water with a Focus on Tulare Lake Basin and Salinas Valley Groundwater. Center for Watershed Sciences, University of California, Davis.

Welch, E.B., Horner, R.R., Patmont, C.R., 1989. Prediction of nuisance periphytic biomass: A management approach. Water Research 23, 401–405. <https://doi.org/10.1016/0043-1354(89)90130-9>

Wells, M.L., Trick, C.G., Cochlan, W.P., Hughes, M.P., Trainer, V.L., 2005. Domoic acid: The synergy of iron, copper, and the toxicity of diatoms. Limnology and Oceanography 50, 1908–1917. <https://doi.org/10.4319/lo.2005.50.6.1908>

Worcester, K.R., Paradies, D.M., Adams, M., 2010. Interpreting Narrative Objectives for Biostimulatory Substances for California Central Coast Waters (Technical Report). Central Coast Ambient Monitoring Program, California Central Coast Water Board.

# Appendix I. Important Definitions

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

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

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

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

1. **Designated uses** of the water body (e.g., recreation, water supply, aquatic life, agriculture; Table 1.1),
2. **Water quality criteria** to protect designated uses (numeric pollutant concentrations and narrative requirements), and
3. **Antidegradation policy** to maintain and protect existing uses and high-quality waters.

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

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

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

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

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

* Biological criteria: a description of the desired biological condition of the aquatic community, for example, based on the numbers and kinds of organisms expected to be present in a water body.
* Nutrient criteria: a means to protect against nutrient over-enrichment and cultural eutrophication.
* Sediment criteria: a description of conditions that will avoid adverse effects of contaminated and uncontaminated sediments.

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

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

Biostimulatory Target:

1. Wetlands typically occur in topographic settings where surface and/or groundwater collects, making the area wet for extended periods of time. While definitions vary, the National Wetland Inventory emphasizes three important attributes of wetlands: (1) a period of flooding or soil saturation; (2) presence of plants adapted to grow in water or in a soil or substrate that is occasionally oxygen deficient due to saturation (hydrophytes); and (3) hydric or saturated soils for periods long enough during the growing season to produce oxygen-deficient conditions in the upper part of the soil, which commonly includes the major part of the root zone of plants (FGDC 2013). [↑](#footnote-ref-2)
2. For estuaries and enclosed bays, “deepwater” refers to the elevation of extreme low water of a spring tide (MLLW). For rivers and lakes, deepwater lies at a depth of 2.5 m, roughly the limit at which emergent plants grow (FGDC 2013). [↑](#footnote-ref-3)