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Status, Trends, and Drivers of Harmful Algal Blooms Along the Freshwater-to-Marine Gradient in the San Francisco Bay–Delta System

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ABSTRACT

Harmful algal blooms (HABs) are on the rise worldwide. Known drivers for the proliferation and intensification of HAB events include increasing nutrient pollution, climate change, regulation and modification of hydrological flow, and the combined effect of climate drivers and nutrient pollution. The San Francisco Bay–Delta system has largely been immune to severe or acute HAB events, but there is both a potential and realized threat which has been underestimated and under-reported, in part because of the lack of coordinated sampling and data reporting. There is also increasing evidence that HABs must be considered in the context of a freshwater-to-marine continuum, and that the physical and political boundaries separating components of the Bay–Delta system are porous barriers to HABs and their toxins. Much remains to be learned about the ecology and physiology of HAB organisms in this system, but five primary environmental drivers can be identified: temperature, salinity, irradiance, nutrients, and stratification/residence time. All these drivers are responding rapidly to climate change, but nutrients are the primary variable that is largely under human control. Plans for the development of a comprehensive monitoring, prediction, and mitigation strategy across the freshwater-to-marine continuum have been documented; effectively following through on these plans provides a roadmap toward identifying the drivers and threats—and reducing the potential consequences now and in the future. While HABs alone are not a sufficient motivator for potentially costly and extensive mitigation efforts, there is strong evidence that decreasing nutrient loads, maintaining hydrological connectivity while minimizing stagnant regions, and managing the biota to maintain biodiversity of the Bay–Delta system will result in multiple co-benefits, including reduction of the HAB threat potential.

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KEY WORDS

San Francisco Estuary, harmful algal blooms (HABs), algal toxins, nutrients, hydrology, climate change

INTRODUCTION AND BACKGROUND

This paper is a contribution to the State of the Bay–Delta Science assessment for 2022 and addresses the increasing interest in potential effects of harmful algal blooms in the system. Harmful algal blooms (HABs) are a catch-all term for phytoplankton and macroalgae that result in a recognized negative effect as perceived by society. This classically includes high-biomass events that may result in low oxygen, poor aesthetics, or disruption of food webs and altered carbon and nutrient cycling, as well as the presence of algal toxins with or without high biomass (Smayda 1997; Sunda et al. 2006; Kudela et al. 2015). This paper focuses primarily on production of algal toxins and disruption to food webs; a related analysis (Sutula et al. 2017) provides more direct linkages between nutrients, biomass, harmful algal events, and low oxygen. There is clear evidence for global expansion of HABs in freshwater, estuarine, and marine waters (Paerl et al. 2018; Glibert et al. 2018), with corresponding effects and consequences for ecosystem health, food and water security, human health, and the recreational and aesthetic value of these waters. Known drivers for the proliferation and intensification of HAB events include increasing nutrient pollution (Anderson et al. 2008; Heisler et al. 2008; Paerl et al. 2018; Burford et al. 2020); climate change (Wells et al. 2015; Gobler 2020; Griffith and Gobler 2020), which includes rising temperatures, shifts in rainfall patterns and shifts in both species distribution and phenology, regulation, and modification of hydrological flow in freshwater systems (Paerl et al. 2018); and the combined effect of climate drivers and nutrient pollution (Burford et al. 2020; Glibert 2020; Paerl and Barnard 2020)—in addition to numerous and complex local processes related to species succession, habitat suitability, temperature, nutrients, physical structure, etc. (Paerl 1988; Anderson et al. 2019).

Estuaries can be particularly complex in terms of HABs, given the competing influence of both the watershed and rivers (source waters) and the ocean (receiving waters) with the estuarine system forming a dynamic mixing zone (cf. Peierls et al. 2012; Hall et al. 2013; Harding et al. 2016). Arguably, estuaries are where the greatest potential effects for HABs occur, given that estuarine systems are both highly populated by humans and highly productive; they provide numerous valuable ecosystem functions, and are sites of intensive aquaculture, subsistence, and commercial fisheries. Some well-studied examples include the Pearl River and Changjiang (Yangtze) River estuaries (China; Zhou et al. 2017), the Indian River Lagoon (Florida, USA; Phlips et al. 2011), Nauset (Cape Cod, MA, USA; Crespo et al. 2011), Chesapeake Bay (USA; Tango and Batluk 2013), the Neuse River (North Carolina, USA; Rothenberger et al. 2009), and the New South Wales estuaries (Australia; Ajani et al. 2013). Conspicuously absent from this list are several large and productive estuarine systems in the US, including the Columbia River estuary and the San Francisco Bay–Delta system, where until recently HAB problems were considered to be negligible or at least under-reported.

For description of specific geographic and ecological regions within the San Francisco Estuary (estuary; Figure 1), this chapter generally follows the convention of Sutula et al. (2017) and Lehman et al. (2021). The Delta extends from the Sacramento River (north) and the San Joaquin River (south); it extends northward to the head of tide at Freeport on the Sacramento River and Vernalis on the San Joaquin River. San Francisco Bay (SFB) consists of South Bay, Central Bay, and San Pablo Bay, with Suisun Bay acting as a transitional brackish system that connects the more marine SFB waters with the fresher Delta waters (note that Figure 4 divides SFB into Suisun, San Pablo, Central, South Central, and South Bay). The entire system is referred to as the San Francisco Estuary (estuary) and is the largest Pacific estuary in the Americas. The region combining Suisun with the Delta is often referred to as the upper San Francisco Estuary (upper estuary) to separate it from the more marine SFB.

As described in Sutula et al. 2017, nutrient over-enrichment has led to ecosystem impairments in the majority of the world's estuaries (Nixon 1995; Boesch et al. 2001; Diaz and Rosenberg 2008; Bricker et al. 2008), and this impairment

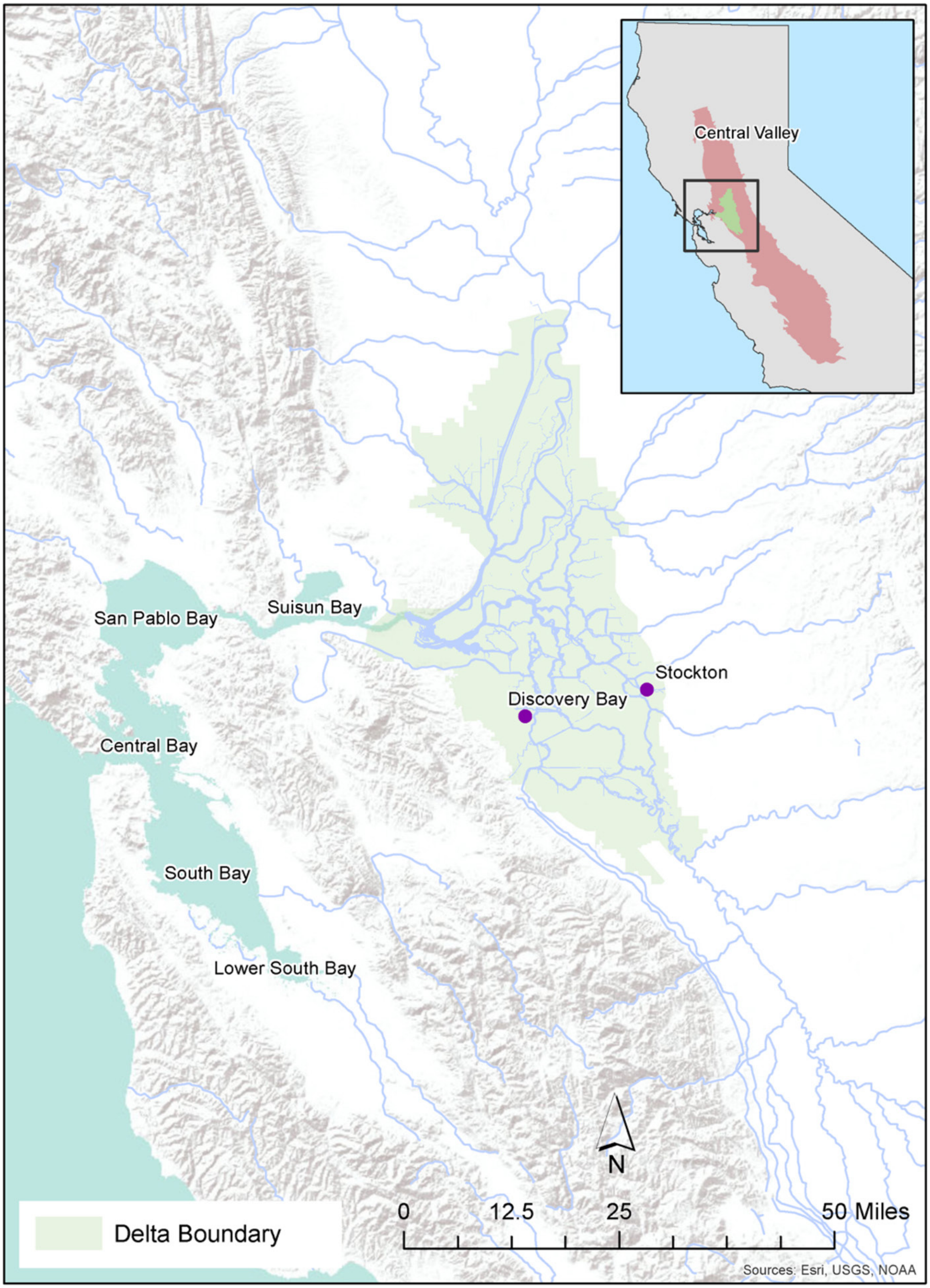

Figure 1 Map of the study region showing San Francisco Bay, the Sacramento–San Joaquin Delta (*green shading*), and the sampling sites used in Figure 5B.

often includes the presence or expansion of HAB organisms that respond directly to nutrient enrichment (Heisler et al. 2008). SFB has largely been resistant to proliferation of HABs, in part because of vigorous tidal mixing and flushing, which leads to both light limitation and rapid flushing that precludes biomass accumulation (Cloern 1982, as summarized in Sutula et al. 2017). Within the marine to brackish waters of SFB, several well-known HAB organisms have been identified, including the diatom genus *Pseudo-nitzschia*, which produces the neurotoxin domoic acid (DA), and the dinoflagellate genus Alexandrium, which produces paralytic shellfish toxins. These two organisms and toxins are generally considered to be the dominant HAB issues on the US West Coast (Lewitus et al. 2012) and have been present at background concentrations for decades in SFB, with occasional blooms that are greatly muted relative to comparable coastal waters in California (Sutula et al. 2017) or the Chesapeake Bay as a comparative mid-latitude estuarine system (Wolny et al. 2020).

Within the riverine end of the system, the upper estuary has also exhibited some signs of resistance to freshwater HABs, with absence of the toxic cyanobacterial genus *Microcystis* before ~1982 (Lehman and Smith, 1991) and absence of large *Microcystis* blooms before ~1990 (Lehman et al. 2005). Most HABs are attributable to cyanobacteria (cyanoHABs) in freshwater (Paerl et al. 2018; Huisman et al. 2018), and severe events are typically caused by the cyanobacterial genus *Microcystis*, a non-nitrogen-fixing cyanobacterium that produces microcystin, a potent hepatotoxin (Vesterkvist et al. 2012).

HARMFUL ORGANISMS AND TOXINS IN THE BAY-DELTA

Harmful Algal Bloom Organisms in Fresh, Brackish, and Marine Waters

In freshwater, *Microcystis* is the most commonly reported cyanoHAB in the region, and microcystins are generally considered to be the most common cyanoHAB toxin globally (Vesterkvist et al. 2012; Preece et al. 2017). Several other cyanoHAB genera are also present in the upper estuary, however (Table 1), including *Anabaenopsis*, *Aphanizomenon*, *Dolichospermum*, *Lyngbya*, *Phormidium*, *Planktolyngbya*, *Planktothrix*, and *Oscillatoria* (Cloern and Dufford 2005; Lehman et al. 2010; Kurobe et al. 2013, 2018; Spier et al. 2013; Sutula et al. 2017). The toxin-producing cyanobacterium *Cylindrospermopsis raciborskii* has also been observed in the Delta (cf. Mioni et al. 2011) despite it being uncommon in North America (Vico et al. 2020). While generally considered a tropical or subtropical taxon, this organism has been documented to have expanded in temperate regions, and it is considered an invasive species (Dyble et al. 2002; Briand et al. 2004; Paerl and Huisman 2008).

Assessment of cyanoHABs in the upper estuary has relied primarily on intermittent monitoring (e.g., Lehman et al. 2005, 2013, 2020), and has focused nearly exclusively on quantification of *Microcystis* and microcystins until recently. *Microcystis* was reported as early as 1920 (Allen et al. 1920) as part of a diverse cyanobacterial assemblage but was not dominant. While there is always a concern that cells and toxins were presumed absent because of a lack of sampling, as noted, it was not dominant during more routine sampling between 1975 and 1990 (Lehman and Smith 1991). Both cyanobacteria and dinoflagellate biomass have increased between 1975 and 1993 throughout the upper estuary, with a coincident decrease in diatom biomass (Lehman 2000). Before 1999, the colonial (bloom) morphology of *Microcystis* was not observed in the upper estuary (Lehman et al. 2005). However, blooms have been recorded in the upper estuary from 1999 onward (Lehman 2000; Lehman et al. 2005, 2008, 2010; Spier et al. 2013), and have become an increasing threat. Biomass and toxins from *Microcystis* have been documented as present through at least 180 km of waterways from freshwater to brackish water environments, and microcystins are routinely detected, even in the full-salinity waters of SFB (Peacock et al. 2018). The emergence of anatoxin and saxitoxins is even more recent; these toxins were not detected during *Microcystis* blooms before 2016 (cf. Lehman et al. 2021), suggesting that the simultaneous presence of toxin-producing

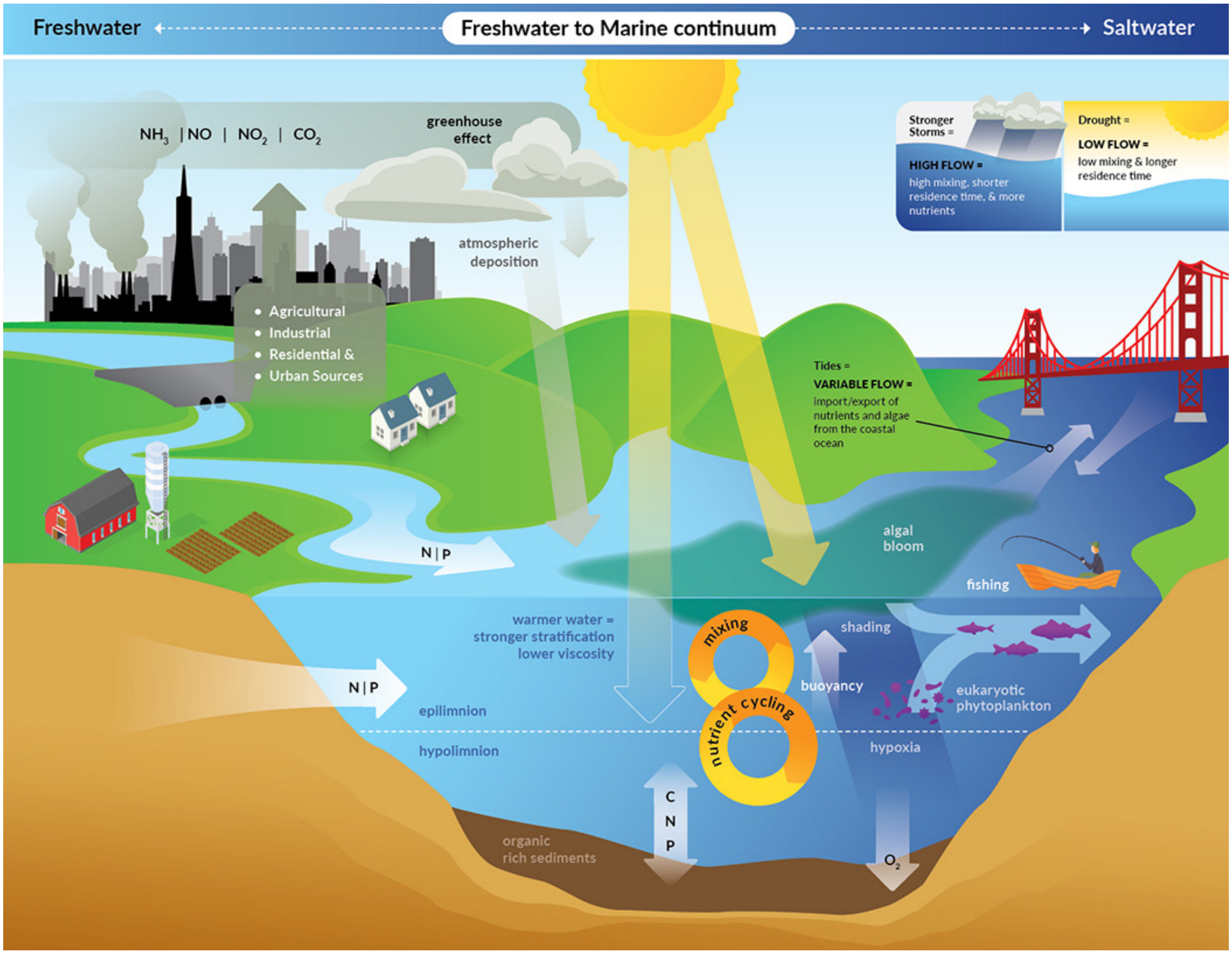

Figure 2 Conceptual model of factors that control HABs in estuarine systems

Dolichospermum, *Aphanizomenon*, and *Microcystis* in the upper estuary is becoming more common (Lehman et al. 2021).

The Delta is not unique in occurrence of multiple toxigenic organisms when compared to other California water bodies. Recent studies have demonstrated that potential toxic cyanobacterial genera and their associated toxins are widespread (e.g., Fetscher et al. 2015; Tatters et al. 2019), and that there is significant transfer of cells and toxins across the freshwater-to-marine continuum (Figure 2; e.g., Gibble and Kudela 2014; Preece et al. 2017; Peacock et al. 2018; Tatters et al. 2021), such that what are considered freshwater taxa have effects downstream, while many freshwater and marine taxa exhibit fairly wide salinity tolerances (see “Salinity and Temperature”).

In brackish and marine waters of the estuary, multiple toxigenic organisms have also been identified and/or the presence of toxins has been confirmed (Table 1). In addition to *Pseudo-nitzschia* spp. (domoic acid), Alexandrium catenella (paralytic shellfish toxins, or PSTs), and *Microcystis* aeruginosa (microcystins), the dinoflagellates Dinophysis spp. (diarrhetic shellfish toxins), Karenia mikmotoi (haemolytic and cytotoxic effects), Karlodinium veneficum (karlotoxins), Heterocapsa triquetra (fish-killing), and the raphidophyte Heterosigma akashiwo (fish-killing; exact toxic mechanism is not

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Table 1 Potential HABs from San Francisco Bay and the Sacramento–San Joaquin Delta, known toxins or effect, and salinity range. ATX = anatoxin-*a*; PbTx = brevetoxin; CYN = cylindrospermopsin; DA = domoic acid; DST = diarrhetic shellfish toxins (okadaic acid, pectenotoxins); ICX = ichthyotoxin; LYN = lyngbyatoxin; MC = microcystin; STX = saxitoxin and analogues.

			Salinity range		
Genus	**Group**	**Potential toxin(s) or effect**	**Low (0 to 4)**	**Moderate (4 to 16)**	**High (16+)**
Anabaena	Cyanobacteria	ATX, CYN, MC, STX	X	X	
Anabaenopsis	Cyanobacteria	MC	X	X	X
Aphanizomenon	Cyanobacteria	ATX, CYN, STX	X	X	
Cylindrospermopsis	Cyanobacteria	ATX, CYN, STX	X		
Cylindrospermum	Cyanobacteria	ATX, MC	X		
Dolichospermum	Cyanobacteria	ATX, CYN, MC, STX	X	X	
Lyngbya	Cyanobacteria	CYN, LYN, STX	X	X	X
Microcystis	Cyanobacteria	MC	X	X	X
Oscillatoria	Cyanobacteria	ATX, CYN, MC, STX	X	X	X
Phormidium	Cyanobacteria	ATX, MC	X	X	X
Planktolyngbya	Cyanobacteria	ATX, MC	X	X	X
Planktothrix	Cyanobacteria	ATX, MC	X	X	
Pseudo-nitzschia	Diatom	DA		X	X
Akashiwo	Dinoflagellate	Bird mortality			X
Alexandrium	Dinoflagellate	STX		X	X
Dinophysis	Dinoflagellate	DST			X
Heterocapsa	Dinoflagellate	ICX		X	X
Karenia	Dinoflagellate	PbTx			X
Karlodenium	Dinoflagellate	ICX	X	X	X
Heterosigma	Raphidophyte	ICX		X	X

known) are all present, as is the dinoflagellate *Akashiwo sanguinea*, which does not produce a toxin but produces instead a low-molecular-weight compound that disrupts the waterproofing effect of bird feathers, leading to mortality from hypothermia. *Akashiwo* blooms have resulted in widespread bird mortalities in California (Jessup et al. 2009), and *Akashiwo* has on occasion produced spectacular blooms in South Bay (Cloern et al. 2005). All the marine taxa are recognized as being persistent members of California coastal assemblages and were likely always present in SFB, although there is evidence that opening of the South Bay Salt Ponds may have resulted in geographical expansion of some previously rare or unreported organisms (see "Case Studies from the San Francisco Estuary").

Relevant Harmful Algal Bloom Toxins

Harmful Algal Bloom toxins are diverse; many of these compounds are secondary metabolites with unknown function for the producing organism, and there are many more metabolites that have toxic effects on some but not all organisms. Toxins of concern include the most common compounds that have a known, deleterious effect on fish, birds, and mammals. In many cases, a given genus may produce multiple toxins (Table 1), but for a specific geographic region it is more common for a subset of toxins to be present. Following Table 1, the major toxin groupings will be addressed, recognizing that for some groups, such as microcystins, there are well over 100 known variants (Vesterkvist et al. 2012). The most commonly monitored (and observed) toxin in the upper estuary belongs to this group, while in the marine environment domoic acid and paralytic shellfish toxins are considered to be the most prevalent in California and more broadly on the West Coast of the United States (Lewitus et al. 2012).

The toxins are listed below in alphabetical order, with **Anatoxin-*a***, **Cylindrospermopsin**, **Lyngbyatoxin**, and **Microcystins** being predominantly freshwater in origin. When relevant, the lethal dose for 50% of a human population (LD50; typically determined using mice as model organisms) is provided.

Anatoxin-*a* (ATX). Anatoxin-*a* is an alkaloid that inhibits neuromuscular receptors by disrupting cellular ion channels, resulting in muscle failure and sometimes death. There are several variants including homoanatoxin-*a* and dihydroanatoxin-*a*, and homodihydroanatoxin-*a* (Puddick et al. 2021). ATX production occurs in multiple cyanobacteria genera (Table 1) and in California is often associated with benthic mat-forming species (Bouma–Gregson et al. 2018). ATX exposure is typically through ingestion of cells or water, Iding drinking water. Anatoxins have generally been reported at trace concentrations in the Delta (Lehman et al. 2005, 2010) but were persistent and reached ~0.24 $\mu g\,L^{-1}$ in 2016 in the upper estuary (Lehman et al. 2021). LD50: 200 to 375 $\mu g\,kg^{-1}$ (Carmichael and Boyer 2016).

Brevetoxin (PbTx). This is a lipid-soluble cyclic polyether produced by the marine dinoflagellate *Karenia*, with multiple analogues identified in various species and strains of the genus. These compounds open voltage-gated sodium ion channels in cell membranes, leading to an influx of sodium, and subsequent cytotoxicity. Brevetoxin exposure occurs through direct ingestion, aerosolization, and accumulation in the food web, particularly in shellfish (Fleming et al. 2011). LD50:200 to 6600 $\mu g\,kg^{-1}$, depending on exposure route (Kirkpatrick et al. 2004).

Cylindrospermopsin (CYN). Cylindrospermopsin is a hepatotoxic polyketide-derived alkaloid originally associated with the cyanobacterial genus *Cylindrospermopsis*, but also identified in a wide range of genera (Table 1). The molecule is highly water soluble, and exposure is typically through ingestion of water, although contact exposure has also been documented through, for example, recreational activities (Moreira et al. 2013). It is considered to be cytotoxic, dermatotoxic, genotoxic, and hepatotoxic. LD50: 2.1 $mg\,kg^{-1}$ (short-term exposure) or 200 $\mu g\,kg^{-1}$ (chronic exposure; Carmichael and Boyer 2016).

Diarrhetic Shellfish Toxin (DST). This group of polyketide marine toxins include okadaic acid and the analog dinophysistoxin and are primarily associated in California with the marine dinoflagellate Dinophysis (Shultz et al. 2019). This class of compounds are protein phosphatase inhibitors. Exposure is typically through consumption of shellfish and leads to food poisoning-like symptoms. DST was detected in the majority (>91%) of shellfish samples collected in 2012 and 2014 in SFB, with concentrations up to 146 µg/kg (Peacock et al. 2018). LD50:192 $\mu g\,kg^{-1}$ (Van Dolah 2000).

Domoic Acid (DA). This small-molecular-weight amino acid is primarily produced by the marine diatom *Pseudo-nitzschia*. It is a neurotoxin that interacts with and activates neural receptors in vertebrates and is more specifically an antagonist for glutamate receptors. Acute symptoms include food poisoning-like conditions as well as confusion, loss of memory, brain damage, and death. Exposure is nearly always through direct consumption of algae or filter-feeding organisms that accumulate the toxin. While primarily a marine toxin, it has frequently been detected in the brackish waters of SFB; concentrations in bivalves ranged from ~20 to 565 $\mu g\,kg^{-1}$, and DA was present in 100% of tested mussels in 2012 and 2014 (Peacock et al. 2018). LD50:35 to 70 $mg\,kg^{-1}$ for oral ingestion (Van Dolah 2000).

Ichthyotoxin (ICX). This is a catchall for secondary metabolites produced by algae that are harmful to fish and other organisms. It includes specific toxins such as karlotoxin (produced by the brackish/marine dinoflagellate *Karlodinium*) as well as compounds such as hydrogen peroxide produced by the raphidophyte *Heterosigma*. Ichthyotoxic effects are observed in both freshwater and marine systems, but the causative compounds are often not identified.

Lyngbyatoxin (LYN). This compound and its derivatives and related toxins (aplysiatoxin, debromoaplysiatoxin) are alkaloids that most commonly cause contact dermatitis ("swimmer's itch"), although ingestion can also lead to gastrointestinal irritation and food poisoning-like symptoms (Jiang et al. 2014). This toxin is primarily marine, associated with benthic cyanobacteria, but at least one case of contact dermatitis in a dog exposed to *Lyngbya* in California lake waters has been reported, with the causative toxin identified as debromoaplysiatoxin (Puschner et al. 2017). These compounds have not specifically been identified in the estuary system. LD50:250 $\mu g\,kg^{-1}$ (Carmichael and Boyer 2016).

Microcystins (MCs). This group of toxins are small, monocyclic peptides that undergo frequent substitutions of L-amino acids, leading to the numerous variants; toxicity depends on the specific substitution, and is often reported as equivalents to MCLR, the most common compound. All microcystins have an Adda amino acid group, and some detection methods specifically target this part of the molecule (e.g., US EPA 2016). The genes for MC production are widespread in cyanobacteria genera (Table 1). MCs act by inhibiting protein phosphatase in cells, leading to liver damage, tumors, and death (cf. Preece et al. 2017). Microsystin exposure is typically through ingestion of cells or water, including drinking water. Microcystins are frequently detected in estuarine sampling of phytoplankton as well as in marine bivalves within SFB, with concentrations exceeding 3.5 $\mu g\,L^{-1}$ in the estuary and up to 18.9 $\mu g\,kg^{-1}$ in SFB bivalves (Peacock et al. 2018; Lehman et al. 2021). LD50:25 to 1000 $\mu g\,kg^{-1}$ (Carmichael and Boyer 2016).

Paralytic Shellfish Toxins (PSTs). This is a broad group of compounds related to saxitoxin that occur in both cyanobacteria and marine dinoflagellates. Toxicity depends on the specific chemical structure and is typically reported as STX (saxitoxin) equivalents. These are potent neurotoxins that act as selective, reversible, voltage-gated sodium channel blockers. Exposure leads to a variety of neurological symptoms culminating in death. Exposure can occur through direct consumption of algae, trophic transfer through, for example, shellfish, as well as consumption of the dissolved toxin in water and drinking water. PSTs in the upper estuary were low but present in sampling from 2014 to 2018 (Lehman et al. 2021). In SFB, PSTs are routinely detected in marine bivalves at up to 34 $\mu g\,kg^{-1}$ (Peacock et al. 2018). LD50:10 to 30 $\mu g\,kg^{-1}$ STX equivalents (Carmichael and Boyer 2016).

Regulatory Monitoring and Guidance

For many of the HAB organisms there is no consistent monitoring for associated toxins, and the specific toxin(s) and mechanisms are poorly characterized for some organisms (e.g., *K. mikimotoi*, *H. akashiwo*). Routine monitoring for DA, MCs, DSTs, and (with less frequency) PSTs has taken place as part of the US Geological Survey's shipboard time-series that spans 145 km from lower South Bay to the lower Sacramento River since 2011 (Cloern and Schraga 2016; Sutula et al. 2017; Peacock et al. 2018). The San Francisco Estuary Institute has expanded this effort to include bivalves (mussels) at several sites in SFB (Peacock et al. 2018). For the upper estuary, consistent sampling for cells and toxins (MC, ATX, STX) has been ongoing since 2014 (Lehman et al. 2021). Before that date, sampling was project-specific. While limited to a subset of toxins and focusing primarily on the marine end of the system, results from both multi-decade time-series of HAB presence and abundance (Figure 3) and occurrence of toxins (Figure 4) demonstrate that harmful algal taxa are nearly always present at low to moderate abundance and concentration in the system, and that classically "marine" and "freshwater" toxins readily exchange across the full 145-km range and salinity gradient of the USGS cruises (Sutula et al. 2017).

Assessment of cyanoHABs in the freshwater Delta has relied primarily on intermittent monitoring (e.g., Lehman et al. 2005, 2013, 2020), and has focused nearly exclusively on quantification of *Microcystis* and microcystins. Some monitoring has been done for other common toxins, including ATX and CYN (Mioni et al. 2011; Lehman et al. 2021; Figure 5A), but those toxins were absent in the first study, and highly variable in the second study. More consistent monitoring (Lehman et al. 2021; Figure 5A) identified several co-occurring toxins but also confirmed that, for now, MCs are likely the dominant and most relevant cyanotoxins of interest for the upper estuary. There is currently no formal monitoring of cyanoHABs in the upper estuary, although numerous research groups and agencies routinely collect relevant data on blooms, MCs, and associated environmental parameters (Thronson et al. 2020). The much higher levels of MCs compared to Lehman et al. (2021) are documented in Discovery Bay and Stockton Basin (Figure 5B), highlighting the importance of reduced mixing and long retention time.

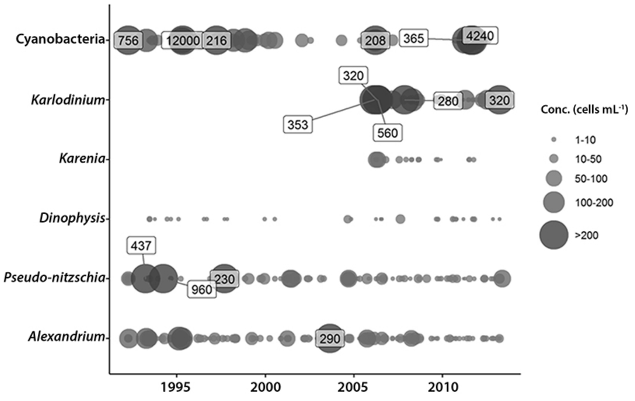

Figure 3 Abundances of six HAB taxa in SFB from 1993 to 2014. *Symbols* indicate abundance (cells mL^{-1}) by cruise. The sample with the highest cell density is indicated for cruises with HAB enumerated at multiple locations. Inset values give abundance at stations > 200 cells mL^{-1}. *Source: Sutula et al. 2017, with data from Nejad et al. 2017.*

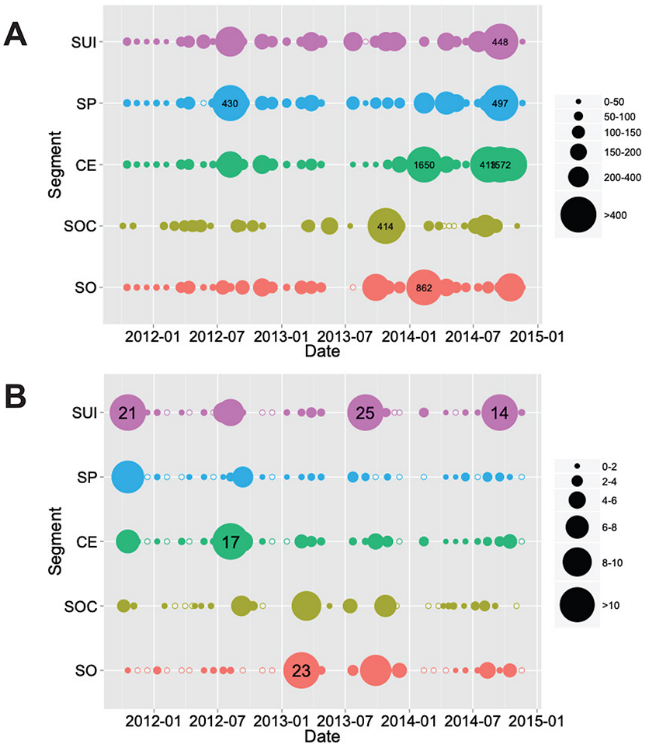

Figure 4 Concentration of domoic acid (ng g^{-1}; **A**) and microcystins (ng g^{-}; **B**) from Solid Phase Adsorption Toxin Tracking (SPATT) passive samplers deployed in the R/V Polaris surface mapping system for South Bay (SO), South/Central Bay (SOC), Central Bay (CE), San Pablo Bay (SP), and Suisun Bay and eastern stations (SUI). *Circles* indicate domoic acid concentration; for SPATT greater than 400 ng g^{-}, the inset numeric value indicates the concentration. *Source: Sutula et al. 2017. White circles with no color* indicate samples with toxins below the minimum detection limit.

Regulatory guidance for HAB organisms (abundance) and toxin levels is generally lacking except for a few specific organisms and toxins (Table 2); for convenience, toxin thresholds are provided in regulatory units (which vary by toxin and source) and as equivalent ppm. For human consumption of fish and shellfish, the US Food and Drug Administration (FDA) regulates the marine toxins DA and PSTs with thresholds of 20 mg kg^{-1} (20 ppm) and 80 µg 100 g^{-1} (0.8 ppm), respectively. The FDA also provides guidance for DSTs of 0.16 mg kg^{-1} (0.16 ppm). There is no regulatory toxin level for MCs in fish and shellfish, although California's Office of Environmental Health and Hazard Assessment (OEHHA) set a guidance level for MCs in fish tissue for human consumption at 10 µg kg^{-1} (0.01 ppm; OEHHA 2012). Multiple agencies also provide guidelines for presence of marine HAB organisms in water bodies, but criteria vary widely, depending on both the organism and agency. As an example, for California, a commonly accepted threshold of concern for *Pseudo-nitzschia* is 10,000 cells L^{-1} (Anderson et al. 2016), while the mere presence of *Alexandrium* is considered problematic (Sutula et al. 2017).

Cyanotoxins have a wide range of guidance and thresholds. (Table 2). The most widely

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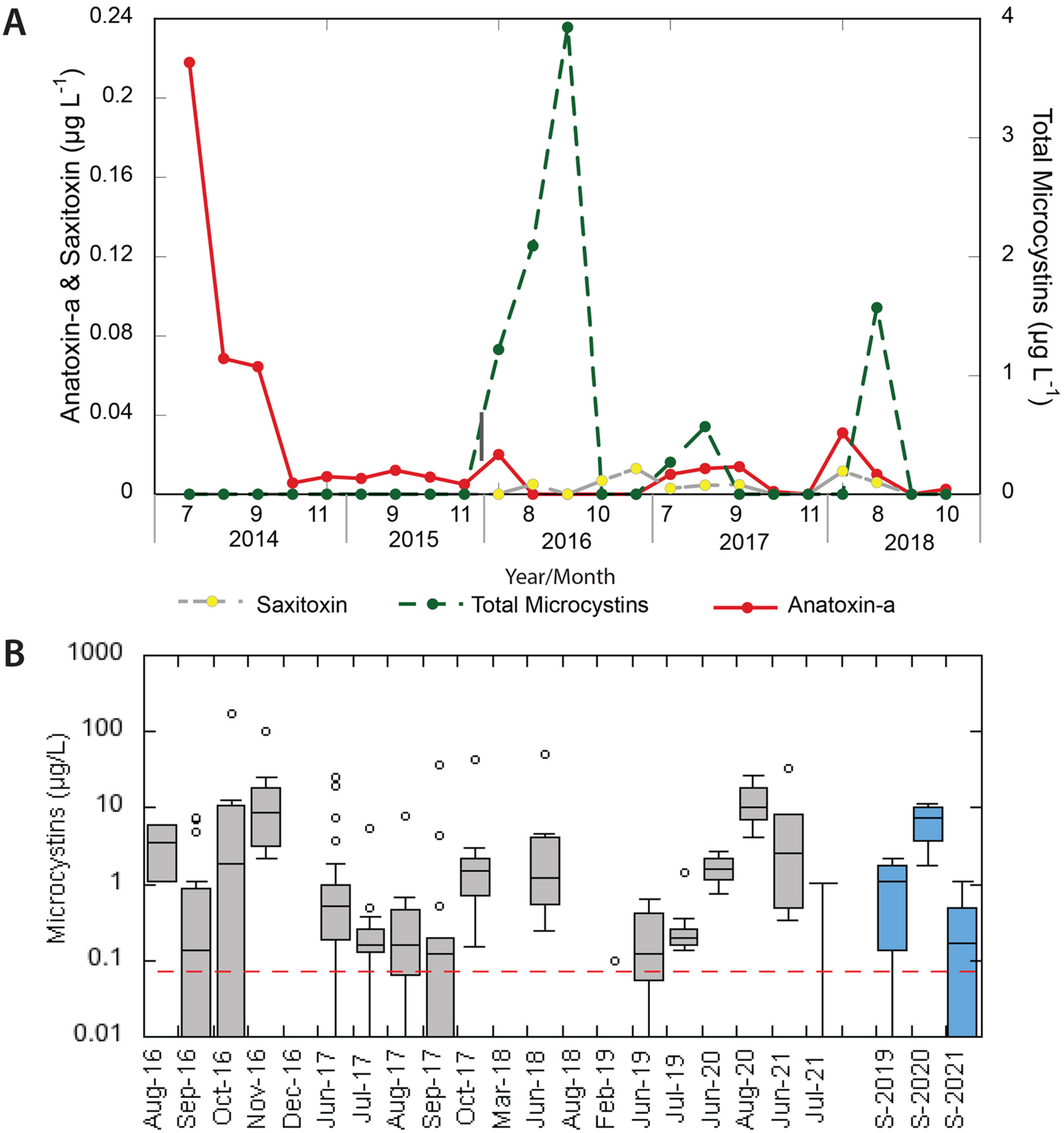


Figure 5 Average concentration of cyanotoxins (**A**) reported by Lehman et al. (2021) for ten sites in the upper estuary, and (**B**) concentrations of microcystins for Discovery Bay (*grey*; monthly sampling) and Stockton Basin (*blue*; annual summer sampling for June through August). The horizontal dashed line indicates the 0.8 µg L^{-1} alert level. Surface scum was not included in either data set, but for **Panel B**, scum values were 189 +/− 128 µg L^{-1} ($n = 3$; Discovery Bay) and 274 +/− 361 µg L^{-1} ($n = 6$; Stockton) for the mean and standard deviation of collected scum samples.

Table 2 Potential HABs from the San Francisco Estuary. Alert levels based on cell abundance used in other regions, and regulatory limits or guidance.

Organism	Alert level	Regulatory guidance for toxins	Reference
Alexandrium	Presence	800 µg kg^{-1} STX equivalents	*http://www.scotland.gov.uk/Publications/2011/03/16182005/37*
Dinophysis	100 to 1,000 L^{-1}	160 µg kg^{-1} DST	*http://www.scotland.gov.uk/Publications/2011/03/16182005/37*; Vlamis et al. 2014
Karenia mikimotoi	5,000 L^{-1}	800 µg kg^{-1} PbTx	NSSP
Karlodinium veneficum	5,000 L^{-1}		Alan Plaice (2015 pers. comm. with RMK unreferenced, see “Notes”)
Pseudo-nitzschia	10,000 to 50,000 L^{-1}	20,000 µg kg^{-1} (30,000 µg kg^{-1} for Dungeness crab viscera)	Cal-HABMAP; Shumway et al. 1995; Anderson et al. 2009
Cyanobacteria* (Recreational Contact Health Thresholds)	4 to 100 x 10^{6} L^{-1}	ATX: Detection, 20, 90 µg L^{-1} (Caution, Warning, Danger) CYN:1, 4, 17 µg L^{-1} (Caution, Warning, Danger) MC:0.8, 6, 20 µg L^{-1} (Caution, Warning, Danger)	*https://mywaterquality.ca.gov/habs/resources/habs_response.html*
Cyanobacteria* (USEPA Recreational Water Quality Criteria)	None	15 µg L^{-1} CYN 8 µg L^{-1} MCs	USEPA 2019; Water-quality criteria allow states to list a waterbody as impaired on the 303(d) list of impaired waterbodies.
Cyanobacteria[a] (USEPA Drinking Water Health Advisory Thresholds)	None	MCs:0.3 µg L^{-1} (children) MCs:1.6 µg L^{-1} (adults)	USEPA 2015a, 2015b. Based on a 10-day exposure.
Cyanobacteria* (human consumption)		ATX:5000 µg kg^{-1} CYL:70 µg kg^{-1} MCs:10 µg kg^{-1}	OEHHA 2012

a. There are numerous, conflicting guidelines for cyanoHABs; the limits most relevant to California are included.

recommended guidelines are from the World Health Organization (WHO), which set recreational guidance and action levels for cyanobacteria, chlorophyll-*a*, and microcystin-LR using a tiered approach (WHO 2006). The WHO guidance levels are categorized into low, medium, or high probability of adverse health effects. The **low** category has guidance levels of < 20,000 cyanobacteria cells mL^{-1}, < 10 µg L^{-1} (< 0.01 ppm) of microcystin-LR, or < 10 µg L^{-1} of chlorophyll-*a*. The **medium** category has levels of 20,000 to 100,000 cells mL^{-1}, 10 to 20 µg L^{-1} (0.01–0.02 ppm) microcystin-LR, or 10 to 50 µg L^{-1} of chlorophyll-*a*. The **high**-probability category has levels of 100,000 to 10,000,000 cells mL^{-1}, 20 to 2,000 µg L^{-1} (0.02–2 ppm) microcystin-LR, and 50 to 5,000 µg L^{-1} of chlorophyll-*a*. For drinking water specifically, chlorophyll-*a* levels are lowered to 1, 12, and 24 µg L^{-1} respectively for all cyanotoxins (WHO 2020a, 2020b, 2020c, 2020d).

The US Environmental Protection Agency (USEPA) recently introduced guidelines and criteria for both recreational contact and water consumption. For recreational contact, water quality criteria and swimming advisories were set at 8 µg L^{-1} (0.008 ppm) for MCs and 15 µg L^{-1} (0.015 ppm) for CYN (USEPA 2019). The USEPA established health advisory thresholds based on a 10-day exposure for drinking water in MCs and CYN at 1.6 and 3.0 µg L^{-1} (0.016 and 0.03 ppm), respectively, which is reduced to 0.3 and 0.7 µg L^{-1} (0.0003 and 0.0007 ppm) for MCs and CYN in children (USEPA 2015a, 2015b).

Most states have separate guidance for recreational contact with cyanotoxins that deviates significantly from both USEPA and WHO guidelines. Mehinto et al. (2021) summarized the range of health thresholds for each state as 0.8 to 20 µg L^{-1} for MCs (Kansas and Utah use 2,000 µg L^{-1}

or 2 ppm), 1 to 20 $\mu g L^{-1}$ (0.001 to 0.02 ppm) for CYN, detection to 300 $\mu g L^{-1}$ (0.3 ppm) for ATX, and 0.8 to 75 $\mu g L^{-1}$ (0.008 to 0.075 ppm) for STX. California includes quantitative health threshold guidelines for both biomass and cyanotoxins, including MCs, CYN, and ATX, but does not have any thresholds for STX. California's thresholds represent the most conservative end of the range used by most states, including USEPA criteria, and uses a tiered system with three different thresholds for recreation: Caution, Warning Tier, and Danger Tier. The caution level for ATX, CYL, and MCs is detection, 1, and 0.8 $\mu g L^{-1}$, respectively (or detection, 0.001, and 0.0008 ppm). Warning Tier I thresholds are set at 20, 4, and 6 $\mu g L^{-1}$ (0.02, 0.004, and 0.006 ppm) for ATX, CYL, and MCs, respectively. Danger Tier 3 levels are set at 90, 17, and 20 $\mu g L^{-1}$ (0.09, 0.017, and 0.02 ppm). Separate guidelines are provided for consumption by livestock and pets (OEHHA 2012), including subchronic and acute thresholds for MCs and CYL (but not ATX). Subchronic action levels are set at 2 and 10 $\mu g L^{-1}$ (0.002 and 0.01 ppm) for MCs and CYL, 50 and 60 $\mu g L^{-1}$ (0.05 and 0.06 ppm) for cattle, and 100 and 200 $\mu g L^{-1}$ (0.1 and 0.2 ppm) for dogs, respectively. Note that other than for cyanotoxins, there are no separate guidelines for chronic exposure to the majority of HAB toxins.

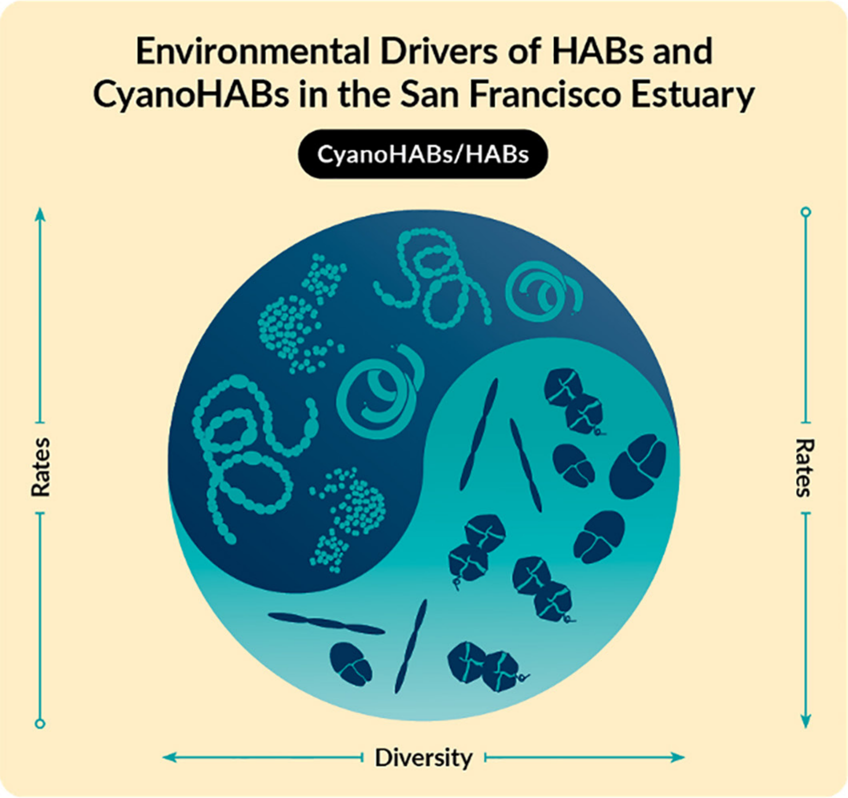
ENVIRONMENTAL DRIVERS OF HARMFUL ALGAL BLOOMS IN THE SAN FRANCISCO ESTUARY

Landscape-Scale Drivers: The Air and Watershed Perspective

Phytoplankton are the main primary producers at the base of all aquatic food chains (Malone et al. 1999; Cloern 2001). These organisms drive biogeochemical cycling, convert inorganic and organic nutrients to organic matter, and are critical to the health and functioning of aquatic ecosystems. In a healthy system, there is considerable diversity in the phytoplankton community, driven by the availability of light, nutrients, temperature, and other abiotic factors such as salinity. The global expansion and intensification of harmful algal blooms in both freshwater and marine systems is well documented (cf. Anderson 2021; Gobler 2020; Hallegraeff et al. 2021; Paerl and Barnard 2020) and is generally a response to favorable environmental conditions for a particular subset of the phytoplankton community. This response is highly site-specific (e.g., Anderson et al. 2021), and often driven by multiple environmental factors at local, regional, and global scales (Figure 2; Wells et al. 2015; Glibert et al. 2018; Paerl et al. 2018; Burford et al. 2020), which should not be surprising given the diversity of HAB organisms and habitats (cf. Glibert et al. 2018).

Despite this complexity, a few environmental drivers are known to play a disproportionate role in promoting HABs, particularly for temperate coastal estuaries such as the estuary (Figures 2, 6). These drivers include changes in the hydrological cycle, which includes but is not limited to changes in drought intensity (Lehman et al. 2020) and residence time (Paerl et al. 2011, 2018; Kudela et al. 2020), eutrophication, or nutrient pollution (Paerl 1988; Anderson et al. 2002, 2008; Heisler et al. 2008; Sutula et al. 2017; Cloern et al. 2020), and the effects of climate change, often acting as a co-factor with nutrients and hydrology (Paerl and Paul 2012; Glibert 2020; Gobler 2020; Griffith and Gobler 2020; Wells et al. 2015, 2020). Berg and Sutula (2015) specifically identified stratification and residence time, water temperature, water column irradiance, and salinity, and nutrients as the five main drivers that control cyanoHAB events in the upper estuary, although these same factors apply equally well to the marine waters of the SFB (Cloern and Dufford 2005; Cloern et al. 2017; Wells et al. 2015). These factors are described in more detail below, beginning with the air and watershed, followed by hydrological controls, bio-physiological interactions (temperature, salinity, irradiance), and, lastly, nutrient concentration and form.

While HAB events are often considered to be a “local” problem, the land, the air, and freshwater and marine receiving waters are linked components of the ecosystem—all of which contribute to or are influenced by the presence of HABs (Figure 2; Paerl et al. 2018). The airshed is often overlooked but is a critical component in promotion of HABs (particularly cyanoHABs)



Positive
- High nutrients
- Low turblence
- Long residence time
- High (adequate) light
- Warm temperatures
- Low grazing rates
- Seed population/Refugia
- Low biodiversity

Modulating Factors
- Strong environmental gradients (e.g. stratfication, salinity)
- Nutrient forms & ratios
- Diverse habitats/connectivity
- Selective grazing
- Predator/Prey co-evolution
- High (adequate) light
- Toxin production

Negative
- Low nutrients*
- High turblence
- Short residence time
- Low light
- High grazing rates
- High biodiversity
- Extreme temperatures (warm/cold)

Figure 6 Environmental factors known to contribute to HABs in the San Francisco Estuary. The *increasing (decreasing) rates arrows* indicate the physiological/evolutionary traits associated with individual genera and taxa which may promote or suppress a bloom, while the *diversity arrow* indicates that the emergent community composition is also controlled by what organisms are present.

and negative effects (Plaas and Paerl 2021). Atmospheric deposition of anthropogenically generated N (NHx, NOx, organic N) has been directly related to numerous HAB events (Paerl 1997). Atmospheric N deposition is increasing globally (cf. Glibert et al. 2018). There is a dearth of specific estimates for atmospheric N deposition in the SFB watershed (Cloern et al. 2020), with even less known about deposition for the upper estuary, but McKee and Gluchowski (2011) provide a conservative estimate of 393 metric tons per year total N for South Bay (south of the Bay Bridge) alone, and 16 metric tons per year of total P. For comparison, Cloern et al. (2020) estimated stormwater accounts for 3,942 and 474.5 metric tons of N and P, respectively, per year for the entire SFB. Scaling up by a factor of 10 from South Bay to the full region, atmospheric N deposition would be approximately equal to stormwater, while atmospheric P deposition is considerable but proportionally lower.

Moving into the watershed, Paerl et al. (2018) highlighted the importance of holistically examining linkages from source waters such as headwater streams, through riverine systems, into the estuary, and finally to the coastal marine receiving waters. The estuary watershed covers more than 194,000 km^2. A recent survey of over 1,200 wadeable stream segments in California identified the common occurrence of multiple potentially toxic benthic cyanobacteria, with co-located cyanotoxins found in one-third of all sites (Fetscher et al. 2015), suggesting that cyanoHABs and associated toxins could readily be imported to the Bay–Delta system even if there were no resident populations (see Bouma–Gregson et al. 2017 for a comparable assessment of the Eel River). At the coastal end of the watershed, there are steadily increasing reports of freshwater toxins appearing in marine systems. For example, cyanotoxins have accumulated in aquatic marine birds (Lehman et al. 2010; Gibble et al. 2017) and bioaccumulate in estuarine and marine shellfish (Garcia et al. 2010; Miller et al. 2010; Preece et al. 2015; Gibble et al. 2016; Peacock et al. 2018; Tatters et al. 2019; Biré et al. 2020). Higher-trophic-level marine mammals have been exposed to microcystins, such as cetaceans in southern California (Danil et al. 2021), sea otters in Monterey Bay (Miller et al. 2010), and bottlenose dolphins in Florida (Brown et al. 2018), as well as the recent dramatic example of bald eagle poisoning associated with a recently described cyanotoxin (Breinlinger et al. 2021). Marine HABs can also extend considerable distances into brackish and freshwater systems, with estuarine circulation acting to mix toxins from marine and fresh waters throughout the waterway (Figure 4; cf. Peacock et al. 2018).

Hydrology, Mixing, and Downstream Transport

Many HAB organisms in the Bay–Delta system benefit from reduced mixing and stratification (i.e., stagnant conditions). *Microcystis* is relatively slow-growing compared to other algae (Paerl and Otten 2013), but it regulates buoyancy to balance demands for photosynthetic carbon fixation and acquisition of nutrients—particularly P—from the subsurface and benthic environments, respectively (Harke et al. 2016), leading to characteristic surface scums when *Microcystis* and similar genera aggregate in light to moderately wind-mixed surface waters. Intensification of vertical stratification combined with plentiful nutrients are particularly favorable conditions for development and persistence of scum-forming blooms (Jöhnk et al. 2008; Paerl and Huisman 2008; Huisman et al. 2018). One important aspect of this regulated buoyancy is that cells and colonies can be transported by a variety of processes, including directional transport by currents, vertical distribution caused by turbulence, and directed vertical movement through buoyancy. Some cyanobacterial genera—including *Microcystis, Planktothrix, Anabaena, Aphanizomenon*, and *Dolichospermum*—produce gas vesicles and carbohydrate storage products (Oliver 1994). The carbohydrate storage products provide negative buoyancy, and the amount produced varies as a function of species, irradiance, and growth rates (Visser et al. 1997; Wallace and Hamilton 1999). As a result, cyanobacteria are rarely randomly distributed and often aggregate in surface slicks, patches, and downwind, as well as in backwaters and areas of reduced flow (Puddick et al. 2016), particularly in areas experiencing high light, warm water temperatures, and elevated nutrients (Lehman et al. 2005). While benthic cyanobacterial mats have not yet been identified as a serious threat in SFB, other colonial cyanoHABs such as *Anabaena* have been observed to disperse great distances downstream through formation of oxygen bubbles in normally benthic *Anabaena* mats (Bouma–Gregson et al. 2017).

Given the particular combination of eco-physiological characteristics and attributes of *Microcystis*, what typifies all the places it is found is weak vertical, turbulent mixing and thus temperature-stratified water columns (Huisman et al. 2004, 2018; Paerl et al. 2018). The reason for this is that much as how turbulence affects sediment flocs, turbulent shear breaks up colonies, reducing flotation speeds (O'Brien et al. 2004), and mixes cells over the entire depth (Hozumi et al. 2019), i.e., to depth with lower light levels, or as is common in the Bay–Delta, to depths where respiration rates are greater than rates of primary production (e.g., Lucas et al. 1999; Cloern and Jassby 2012). When mixing is weak, conditions are ideal for the development of *Microcystis* blooms because of their high degree of buoyancy. Conversely, if mixing is relatively strong, *Microcystis* blooms tend not to develop; most likely because they will then have to compete with other, faster-growing, non-cyanobacterial taxa (chlorophytes, diatoms). Weak mixing regimes are often associated with localized environments such as Discovery Bay and downtown Stockton; these and other localized regions likely serve as sources of *Microcystis* biomass and toxins for the entire Delta and downstream. These are generally expected to be regions where currents are especially weak, reducing turbulent mixing and flushing, and allowing summertime stratification to develop.

In estuarine and marine waters, stratification is also an important factor that regulates abundance of HABs (Figure 2). Flagellates (predominantly dinoflagellates) are well-adapted to a "swim" strategy that allows them to aggregate at the surface and vertically migrate to the subsurface to acquire nutrients (Smayda 1997). This includes several common genera from SFB, notably *Alexandrium catenella* (Rines et al. 2010) and *Akashiwo sanguinea* (Cloern et al. 2005; Peacock and Kudela 2014). Other members of the community such as *Dinophysis* and *Pseudo-nitzschia* are frequently associated with "thin layers" or aggregations in the subsurface on density gradients (Rines et al. 2010; Berdalet et al. 2014). Besides making routine monitoring and detection difficult, these responses to the physical/chemical structure of the environment add to the difficulty of predicting the occurrence of HAB events in SFB (Cloern and Dufford 2005; Cloern et al. 2010).

Salinity and Temperature

Salinity and temperature are closely related to density stratification, but these factors also exert control at the organismal level. Estuaries exhibit strong gradients in salinity from freshwater to the coastal receiving waters, and algal communities shift along that gradient. Cloern and Dufford (2005) examined data extending from lower South Bay to the confluence of the Sacramento and San Joaquin rivers for 1992–2001; they identified 500 distinct phytoplankton taxa that clustered into six seasonal patterns, ranging from spring bloom diatoms to groups imported from either the freshwater or marine sources. The HAB dinoflagellate *Alexandrium catenella* was identified as a canonical "episodic" organism that would appear sporadically. CyanoHABs were generally restricted to freshwater or moderately brackish water (salinity $<<10$) and included *Anabaena, Oscillatoria*, and *Aphanizomenon* (*Microcystis* was not identified in the data set), although salinity per se does not exclude all cyanoHABs. For example, the N_2 fixing and toxin-producing bloom-former *Nodularia* has been found in hypersaline lakes (e.g., Pyramid Lake, Nevada (Galat et al. 1990) and estuarine/coastal waters ranging in salinity from ~10 to full strength seawater (>32) (Moisander and Paerl 2000; Moisander et al. 2002). Fortunately, *Nodularia* has not (yet) been detected in the estuary.

From that study, marine HABs—including *Alexandrium*, *Dinophysis*, and *Heterocapsa*—were generally present in brackish- to full-coastal-salinity waters (~10 to 33 salinity). While there is an ecological boundary at approximately salinity of 10, this boundary is porous, particularly for downstream transport of toxins rather than intact cells. Kudela et al. (2020) demonstrated that *Pseudo-nitzschia* grew in salinity as low as 18, with increasing levels of DA as salinity decreased.

Lehman (2000) and Lehman et al. (2005, 2008) reported *Microcystis* in salinity ranging from 0.1 to 18. Preece et al. (2017) conducted a thorough review of cyanoHABs along the freshwater-to-marine continuum, and noted that several common genera—including *Anabaena, Anabaenopsis*, *Microcystis*, and *Oscillatoria*—are particularly salt tolerant. While individual experimental results vary widely, Tonk et al. (2007) demonstrated that *M. aeruginosa* could survive up to 9 days in salinity of 17.5, while both Miller et al. (2010) and Gibble et al. (2016) reported that intact *Microcystis* colonies were present 48 hours after transfer to full-strength seawater (salinity of ~33). Salinities up to 10 generally do not inhibit microcystin production (cf. Preece et al. 2017), and toxins can persist for at least 21 days in full-salinity seawater (Miller et al. 2010). *Microcystis* tolerance to salinity has been linked to spontaneous mutations within the population (Melero–Jimenez et al. 2019). These mutations result in selection for salt-tolerant strains, and this selection is enhanced by gradual exposure to increasing salinity, as would occur with downstream transport in an estuarine system (Melero–Jimenez et al. 2020). A main point from these studies is that the salinity gradient in the estuary is not a particularly effective barrier for either marine or freshwater HABs, consistent with the observed presence of multiple toxins throughout the SFB (Figure 4).

Temperature is also a strong modulator of organismal growth and toxicity, but highly depends on the specific organism and related environmental conditions. Laboratory studies have shown that, in general, cyanobacteria exhibit optimal growth at higher temperatures (often above 25 °C) compared to dinoflagellates and diatoms (Paerl et al. 2011; Griffith and Gobler 2020). As a result, the general consensus is that climate change (warming) will exacerbate cyanoHABs globally (Paerl and Huisman 2008; Harke et al. 2016; Burford et al. 2020; Griffith and Gobler 2020).

In contrast to prokaryotic cyanobacteria, eukaryotic HAB organisms may exhibit very different physiological responses to rising temperatures. SFB is currently dominated by diatoms (Cloern and Dufford 2005), and *Pseudo-nitzschia* (a DA producer) is of particular concern. While toxic outbreaks along the US West Coast have been linked to warming temperatures (McKibben et al. 2017; Trainer et al. 2020), there is also evidence that *Pseudo-nitzschia* toxicity is suppressed above a temperature of about 20 °C (Smith et al. 2018). During the massive DA event in 2015 (McCabe et al. 2016), very little toxicity was observed in SFB despite extraordinarily high levels on the coast (Peacock et al. 2018). This may have been caused by higher estuarine temperatures exceeding the thermal optimum of *Pseudo-nitzschia* during the concurrent marine heat wave (Kudela et al. 2020). In contrast, dinoflagellates may benefit from the cascading environmental effects of increasing temperature, such as enhanced stratification. Another marine heat wave in September 2004, for example, allowed a massive red tide of *Akashiwo sanguinea* to develop in the South Bay in response to thermal stratification (Cloern et al. 2005). That species also exhibits a thermal optimum considerably higher (~30 °C; Boyd et al. 2013) than most diatoms (Paerl et al. 2011).

Irradiance

Irradiance is a fundamental controlling factor on phytoplankton in both freshwater and marine waters of the estuary (e.g., Cole and Cloern 1984; Cloern 1999; Lehman et al. 2013; Strong et al. 2021). Phytoplankton have multiple strategies for modulating exposure to sunlight, but the two most relevant are vertical migration through active swimming, and buoyancy regulation. Dinoflagellates are strong vertical migrators, which allows them under some conditions to produce spectacular blooms (e.g., Cloern et al. 2005). Cyanobacterial buoyancy can provide a competitive advantage over other phytoplankton such as diatoms, which typically rely on turbulence, and cyanobacterial blooms as well as dinoflagellate blooms are dominant in stagnant, well-stratified waters (Reynolds et al. 1987; Paerl and Fulton 2006; Ji and Franks 2007; Huisman et al. 2018; Xiao et al. 2018; Wu et al. 2021). The cyanobacterial blooms can also increase local heating of surface waters, leading to a positive feedback mechanism that continues to promote dominance once the bloom is established (Huisman 2018).

Under future climate change, irradiance is generally expected to vary only slightly to moderately from present conditions at global scales (Wells et al. 2015), but there are examples of climate-driven shifts that affect both SFB and the upper estuary. Cloern et al. (2007) reported positive monthly trends in suspended particulate material (SPM) for 1978–2005, although there was no significant trend in salinity, which can alter bloom frequency by stratifying the water column and reducing light limitation. They concluded that light limitation remained essentially unchanged for the period 1978–2005. Other analyses estimate an annual decrease in SPM of 1.2 to 1.3% yr^{-1} in Suisun Bay (Wright and Schoellhamer 2004; Cloern and Jassby 2012), resulting in order of a 50% increase in algal growth (primary productivity) potential in Suisun Bay (Cloern and Jassby 2012). Given the strong statistical relationship between biomass and HAB species (Sutula et al. 2017), the implication is that the long-term trend of decreasing SPM will indirectly promote HAB events in strongly light-limited regions of the estuary. This declining trend is generally attributed to hydrological control of the San Joaquin and Sacramento rivers (Cloern and Jassby 2012), but Bever et al. (2018) also documented a decadal (1995–2015) reduction in SPM as a result of declining wind speed, attributed to decadal changes in the Pacific Decadal Oscillation (Bever et al. 2018). Hestir et al. (2016) also suggested a positive-feedback mechanism in which increased light penetration led to more submerged aquatic vegetation, which accelerates the decline in SPM. Overall, several mechanisms could lead to increased water clarity, leading to more algal productivity and HAB potential in light-limited regions of the estuary such as Suisun Bay.

Nutrient Concentrations and Forms

The estuary has historically been resistant to classic eutrophication issues, including stimulation of HABs (Sutula et al. 2017; Cloern et al. 2020). Two recent summaries of the estuary (Beck et al. 2018) and SFB (Cloern et al. 2020) provide relevant information on concentration, forms, and trends for nutrients in this region. For the estuary, nitrogen concentrations have consistently decreased since the mid- to late-1990s, but with a long-term trend of increasing ammonium in Suisun starting in the 1970s and extending until at least the 2000s. Subsequent ammonium reductions are associated with wastewater treatment plant (WTP) upgrades (Beck et al. 2018), including a major upgrade to the Sacramento Regional Wastewater Treatment Plant (SRWTP), which included the Biological Nutrient Removal project, reducing ammonium to undetectable levels, beginning in mid-2021. For SFB, the main sources of external nutrients are from WTPs and refineries, stormwater runoff, and outflow from the Delta (Cloern et al. 2020), although, as noted above, atmospheric deposition is poorly constrained as an additional source (Figure 2). Cloern et al. (2020) place SFB in the 87th and 91st percentile for N and P loading respectively, compared to 162 other estuaries worldwide. The overall conclusion is that nutrient concentrations exhibit seasonal and interannual variability related to multiple drivers, but the estuary and SFB are generally highly nutrient enriched, leading to the assumption that phytoplankton (including HABs) are, in general, not limited by N and P. It should be noted however that the nutrient-eutrophication connection in the estuary and SFB proper is quite different from the Delta, where relatively long residence times can lead to significant uptake of nutrients by high phytoplankton biomass, triggering periods of nutrient limitation. Some regions are also influenced by submerged and floating aquatic vegetation (Christman et al. this issue; Brown et al. 2016). While both can influence nutrient concentrations in the water column either through direct uptake or through processes such as denitrification in the sediment (Racchetti et al. 2017; Wang et al. 2021), the relative sink or source terms are highly variable in space and time, and thus likely to have an overall moderate effect on nutrient concentrations in the upper estuary (Cornwell et al. 2014).

While nutrient concentrations and loads are indisputably high, there has been considerable interest in whether the form or ratio of nutrients influences phytoplankton community composition (Figure 2). In particular, there has been an ongoing debate as to whether elevated ammonium inhibits diatoms (Dugdale et al. 2007; Glibert et al. 2014), and whether the ratio—rather than the absolute concentration—of N and P are driving community composition and growth (e.g., Glibert et al. 2011; Glibert 2020). The SRWTP has been the single largest point source of N, resulting in very high (> 70 to 80 µM) levels of ammonium (Glibert et al. 2014; Strong et al. 2021). Observations of impaired water quality and potential HABs in Suisun (Dugdale et al. 2007; Parker et al. 2012; Glibert et al. 2014) and both conceptual (Glibert et al. 2010) and numerical (Dugdale et al. 2013) models have supported the concept that elevated ammonium suppresses nitrate utilization by (especially) diatoms, leading to potential HAB events by selecting for organisms such as chlorophytes or cyanobacteria that readily uptake and assimilate ammonium.

In contrast to those studies, several lab and field experiments have found little to no evidence to support these hypotheses. For example, Berg et al. (2017) found no difference in laboratory growth rates as a function of N form for chlorophytes vs. diatoms isolated from SFB, and that diatoms grew well on elevated ammonium concentrations. Berg et al. (2019) subsequently showed that the common estuarine diatoms *Thalassiosira weisflogii* and *Entomoneis paludosa* were not inhibited by ammonium concentrations typical for Suisun Bay. Specifically addressing the role of the SRWTP, Strong et al. (2021) conducted a series of controlled experiments where water collected upstream or downstream of the outfall was amended with nitrate, ammonium, or effluent. Results showed a preference for ammonium with no inhibition, and fastest growth by diatoms in all treatments. The author concluded that light is the primary factor limiting phytoplankton in the estuary. Lee et al. (2015) also showed that while *Microcystis* isolated from SFB prefers ammonium, it readily grows on all forms of N with no inhibition of nitrate uptake by ammonium.

Two other studies of direct relevance to the planned upgrades to WTPs were conducted by Esparza et al. (2014) and Kraus et al. (2017). The first study showed no inhibition of diatoms after acute exposure to ammonium (> 1000 µmol ammonium compared to background concentrations of ~2 µmol ammonium), and, in fact, ammonium strongly stimulated diatom growth. In the second study, the SRWTP diverted discharge of effluent for 2 days in spring and autumn, allowing investigators to conduct a Lagrangian experiment with and without enhanced nutrients. Results from these studies did not support the hypothesis that wastewater effluent and high ammonium concentrations result in downstream declines in phytoplankton, and there was also no detectable shift toward HABs in this study, with diatoms dominant with or without effluent. It is also important to note that ammonium-N may be rapidly taken up by phytoplankton or cycled to nitrate (Damashek et al. 2016), making total N reductions at least as important as a shift in form, consistent with a National Research Council (NRC 2011) report.

These studies are consistent with the rich literature on nutrient form for marine HABs as well (Kudela et al. 2010). Many HAB organisms will readily use multiple N forms and are well adapted to use nutrients at both low and high concentrations, which is somewhat unusual compared to the canonical view that some groups are low-nutrient adapted and others are not (cf. Kudela et al. 2010). A separate issue is whether nutrient form favors toxin production. Some strains of *Pseudo-nitzschia*, for example, are more toxic when grown on reduced forms of N such as urea (Kudela et al. 2008). Similarly, Chen et al. (2019) demonstrated that *Microcystis* produces more MC precursors when utilizing ammonium, suggesting that growth on ammonium may favor toxicity, while a recent study by Wagner et al. (2021) found that toxicity of *Microcystis* was highest with elevated nitrate and micronutrients, but N addition—regardless of form—stimulated biomass and toxicity, and there is extensive literature from Lake Erie that demonstrates reduced toxicity of *Microcystis* with decreasing N concentrations (Davis et al. 2010; Gobler et al. 2016; Wilhelm et al. 2021).

It is difficult to project short-term experiments to permanent shifts in nutrient form and concentration, but a variety of studies consistently identify factors other than (or in addition to) nutrients as primary drivers of phytoplankton composition and therefore HABs. This is also supported by several statistical analyses of cyanoHAB abundance in the estuary and environmental conditions. For example, Mioni et al. (2011) identified temperature as the primary driver associated with cyanoHAB biomass and toxins but did not include stratification or residence time. Lehman et al. (2008) also found temperature to be a driver, with a minor contribution from nutrient concentrations or ratios using canonical correlation analysis. In that study, streamflow (residence time) was also identified as a major controlling factor in biomass accumulation. More recent work has identified ammonium as a primary N source for *Microcystis* blooms (Lee et al. 2015; Lehman et al. 2015) but also identified nutrient concentrations as non-limiting and therefore not directly regulating blooms (Lehman et al. 2017). Lehman et al. (2020) concluded that while N and P can certainly be important factors for controlling bloom magnitude (e.g., Harke et al. 2016), in the estuary, flow (residence time and supply rates) is more important, given that the estuary is very nutrient enriched (Jassby 2008). Nutrient loads, rather than absolute concentration, are also critical, given that numerous transformations occur (e.g., nitrification, denitrification, etc; Zehr and Kudela 2016), and it is not clear that all of the pathways are fully accounted for in existing nutrient budgets (Cloern et al. 2020).

In summary, nutrient form and concentration have been documented in specific studies as regulators of both marine HAB and cyanoHAB biomass and toxicity but are not consistently identified as the primary drivers of blooms or toxicity in the estuary. For the Delta specifically, nutrient regulation of blooms can be important, given the very long residence times that allow blooms to deplete the ambient inorganic nutrients; however, once established, there is good evidence that *Microcystis* blooms can be maintained for long periods by regenerated nutrients such as ammonium (Lehman et al. 2013, 2015, 2021; Lee et al. 2015) in areas of long residence time. The overriding conclusion from multiple studies is that both the upper estuary and SFB are, on average, nutrient enriched, leading to other environmental factors—such as flow, temperature—controlling the presence of HABs. However, there is also clear evidence of a potential for more HABs, given the elevated nutrient concentrations (Sutula et al. 2017; Cloern et al. 2020), and a clear statistical link between increasing nutrients, increasing chlorophyll, and therefore increasing HABs in the SFB (Sutula et al. 2017). While these drivers are interconnected, two key components are the availability of excess nutrients (providing the potential for blooms) and hydrological forcing, which influences both residence time and stratification, which in turn modulates light availability. Tidal flushing is likely a key regulator that minimizes blooms in the SFB, while long residence time is likely a key regulator that promotes blooms in the upper estuary.

DIRECT AND INDIRECT EFFECTS FROM HARMFUL ALGAL BLOOMS

Human and Animal Exposure

Harmful algae can result in negative consequences to humans and the environment through a variety of effects, including nuisance and noxious blooms, reduction in dissolved oxygen, physical disruption of filter-feeding organisms, and disruption of ecosystem function, but direct human toxicity is typically of greatest concern (Carmichael 1981; Hallegraeff 2014; Lopes et al. 2019). While toxin concentrations in SFB, the estuary, and the Delta are generally low, several toxins—including DA, DSTs, and microcystins—regularly exceed regulatory limits (e.g., Baxa et al. 2010; Lehman et al. 2005, 2017, 2021; Peacock et al. 2018). As noted above, a significant challenge in determining the effect of these toxins is the lack of consistent long-term monitoring, particularly for toxins that are assumed to be primarily marine or freshwater but are routinely found throughout the system (e.g., Figure 4).

Human exposure to algal toxins in the SFB and estuary is poorly documented. Preece et al. (2017) reviewed and summarized potential exposure mechanisms for freshwater toxins, including direct consumption of contaminated water and trophic transfer through the food web, while Plaas and Paerl (2021) summarized other exposure routes such as aerosolization, and Backer et al. (2008) documented recreational exposure in a small lake. Exposure to marine toxins also undoubtedly occurs through recreational harvesting of shellfish (Peacock et al. 2018) and downstream transport of both freshwater and marine toxins to commercial fisheries (Gibble et al. 2016; Preece et al. 2017). While exposure clearly occurs, the lack of consistent environmental sampling has made it difficult to definitively document chronic or acute exposure. An exception is the study of Zhang et al. (2015) who conducted a county-level assessment of chronic exposure to cyanobacterial blooms and unexpectedly high incidence of non-alcoholic liver disease (presumed to be associated with exposure to hepatotoxins such as microcystin). Central California, centered on the estuary and northern Monterey Bay, was the only identified region for the western US where there was clear evidence for chronic human exposure, strongly suggesting that cyanoHABs are a serious and ongoing threat for this system.

A fundamental question which is not well documented is whether chronic exposure to low levels of one or more toxins is of concern, as opposed to acute exposure (i.e., above the regulatory limit for those toxins that have regulatory guidance). Given the association between blooms and chronic liver disease in humans (Zhang et al. 2015), the answer is almost certainly yes, it is of concern, but very few studies address this issue. Sub-lethal chronic doses may produce serious health compromises in humans and marine mammals (Capper et al. 2013; Ferriss et al. 2017), but very few studies have directly addressed chronic exposure to HABs (Young et al. 2020). Of particular concern is chronic exposure of humans and domestic animals to microcystins through drinking water (Greer et al. 2018; Massey et al. 2018), although other potential routes also exist, such as transfer through plants irrigated with contaminated water (Crush et al. 2008; Corbel et al. 2014; Zhang et al. 2021). Given that the Delta exports large amounts of surface water to the California Central Valley for irrigation and animal stocks, it is quite likely that the Delta is also exporting microcystins along with water, leading to potential chronic exposure via drinking water, crop consumption, and livestock consumption.

The airshed must also be considered as a mechanism for transport and contact of aerosolized cyanoHAB toxins. Respiratory symptoms are among the most frequently recorded complaints after suspected inhalation of water during cyanoHAB events, suggesting that the inhalation of cyanotoxins in spray aerosol may activate inflammatory responses in the human body. Stewart et al. (2006) found that recreational water users were more than twice as likely to report mild respiratory symptoms when exposed to cyanoHABs than those unexposed, and Backer et al. (2010) detected microcystins in onshore air samples and the nasal passages of recreational lake users during two cyanoHAB events.

An important consideration is how HAB events affect communities that have differing social-ecological status. Chronic exposure to microcystins is associated with hepatocellular cancer and colorectal cancer, which are expected to disproportionately affect populations with contaminated drinking water and poor access to healthcare (Meneely and Elliot 2013). Poste et al. (2011) noted that fish consumption can lead to acute exposure to microcystins, suggesting that subsistence fishers may be particularly affected, while Peacock et al. (2018) noted that elevated levels of multiple toxins in shellfish from SFB led to the possibility of enhanced risk to subsistence and recreational harvesters. The social-ecological effects of cyanoHAB events in the US are sparsely documented (cf. Carmichael and Boyer 2016) and there is little to no information about the SFE specifically, but Dodds et al. (2009) estimated the combined economic loss from freshwater cyanoHABs in the US to exceed $2.2 billion annually, including economic loss for recreational use and waterfront real estate, costs to treat drinking water, and associated costs to remediate lost biodiversity. Marine HABs also affect marine fisheries differently; closures related to the massive DA event along the US West Coast in 2015–2016 affected smaller vessels and operations from SFB more than larger commercial operators (Jardine et al. 2020), demonstrating that socioeconomic issues affect human populations at all levels, but are most pronounced for those of lower socioeconomic status.

The first documented microcystin poisoning of a dog in California after it ingested water from a stagnant tide pool occurred in the SFB area in 1991 (DeVries et al. 1993). A comprehensive assessment of canine cyanotoxin poisoning in the United States identified possible poisoning by anatoxin-*a* and microcystins in numerous California counties that are part of the estuary watershed; the authors concluded that canine exposure to algal toxins is probably common and largely unreported (Backer et al. 2013). Livestock are also exposed to cyanotoxins through consumption of mats or drinking water but surprisingly little information is available about livestock toxicosis or death in the estuary watershed. OEHHA (2012) provides regulatory guidance for livestock, and fatalities are documented globally (e.g., Stewart et al. 2008). A retrospective study of 1,199 livestock fatalities in California also highlights the importance of microcystins but did not document specific exposure (Varga and Puschner 2012). As with the prevalence and occurrence of toxic algae in aquatic systems, the magnitude of microcystins and their effect on livestock is therefore likely seriously under-reported.

Trophic Transfer and Ecosystem Disruptions

In contrast to the lack of documented cases of HAB toxicity in canines and humans for the estuary, more literature documents trophic transfer of freshwater and marine toxins. Lehman et al. (2005, 2008, 2010) documented the presence of microcystins in zooplankton and suggested that the presence of liver lesions in juvenile Striped Bass indicated recent exposure (Lehman et al. 2010). Acuña et al. (2020) subsequently documented the presence of microcystins in the Threadfin Shad with subsequent accumulation of toxins in gut and liver tissues. Several groups have also documented the presence of microcystins in bivalve grazers throughout the freshwater-to-marine continuum (Gibble et al. 2016; Preece et al. 2017; Peacock et al. 2018).

As summarized by Preece et al. (2017) there are now numerous examples of microcystins being exported from lakes and rivers to estuarine and marine receiving waters. Moving into SFB, at least four toxins (DA, PSTs, DSTs, MCs) have been documented simultaneously in bivalves (Peacock et al. 2018), with both marine and freshwater toxins detected across the full range of salinity. Organisms at higher trophic levels have largely been untested, but down the coast in Monterey Bay microcystins have been associated with Southern Sea otter mortality (Miller et al. 2010), demonstrating that exported toxins are routinely accumulating in multiple prey items. Within SFB specifically, other organisms have not routinely been tested, but recent analysis of anchovy collected in the SFB also show persistent but low levels of both DA and microcystins (Senn and Kudela, unpublished, see "Notes"). Given the key role of anchovies in trophic transfer of phycotoxins (Doucette et al. 2006) this strongly suggests that multiple vectors can transfer toxins throughout the food web, including to humans (Gibble et al. 2016; Preece et al. 2017; Peacock et al. 2018), marine mammals (Lefebvre et al. 2001), seabirds (Gibble et al. 2021), and fish (Acuña et al. 2020).

In addition to direct and indirect toxicity, HAB events may also lead to changes in ecosystem functioning within the upper estuary. Ger et al. (2009; Ger, Teh et al. 2010) identified calanoid copepods as particularly sensitive to microcystins in feeding studies, and subsequently demonstrated that ingestion of toxic *Microcystis* by the copepod *Pseudodiaptomus forbesi* was directly related to population declines of that copepod, potentially disrupting the estuary ecosystem when populations of *P. forbesi* are suppressed. A follow-on study (Lehman et al. 2021) suggests that *Microcystis* blooms may exert bottom-up pressure on the ecosystem by favoring small rather than large zooplankton. Intriguingly, this study suggests *Microcystis* blooms may alter bacterial abundance and diversity, leading to a trophic cascade wherein the bacterial community favors nanoflagellates and other protist grazers, which in turn selects for small copepods such as *Limnoithona* and *Paracylopina* that feed on those protist grazers. Otten et al. (2017) also identified a significant loss of microbial community biodiversity in the estuary coincident with *Microcystis* blooms, although whether the causative effects were solely from microcystins or from toxins in combination with other secondary metabolites was unclear.

More focused research needs to confirm these broad assertions, however. Some targeted studies have shown that the dominant copepod *P. forbesi* is largely food-limited. Increased growth of zooplankton is also statistically linked to increased cyanobacteria concentrations in the estuary despite the low food quality of those prey items (Owens et al. 2019). Other studies have reported suppression of *P. forbesi* in the presence of toxic cells (Ger et al. 2009; Ger, Teh et al. 2010), as well as less susceptibility to toxic *Microcystis* for *P. forbesi* compared to *Eurytemora affinis* from behavioral selection of prey items (Ger, Arneson et al. 2010). These seemingly contradictory results may be a result of adaptation and co-evolution by predators (zooplankton) and prey (algae, including toxic algae). Ger et al. (2016) provide an extensive global review and concluded that prolonged exposure to cyanobacteria shifts zooplankton communities toward better-adapted species and ultimately selects for genotypes more tolerant to exposure.

More broadly, *Microcystis* blooms in the estuary have been associated with shifts toward less productive food web conditions. Glibert et al. (2010) proposed that increasing ammonium concentrations led to a fundamental shift away from diatoms (pre-1982) and towards cyanobacteria (post-2000) in Suisun Bay (Glibert et al. 2010), directly contributing to the documented Pelagic Organism Decline (POD; Sommer et al. 2007). Although the statistical methods employed in that study have been criticized (Cloern et al. 2012) and rebutted (Lancelot et al. 2012), it is nonetheless clear that a shift toward cyanobacteria—including *Microcystis*—would be expected to lower food quality for herbivorous grazers (Winder et al. 2018).

Lehman et al. (2010) documented potential negative effects on fishery production, linking *Microcystis* to the POD, and suggested that *Microcystis* might be particularly influential because of negative effects on copepods and Cladocera. Subsequent studies suggested subtle shifts in community assemblage dependent on *Microcystis* distribution (Otten et al. 2017; Lehman et al. 2018). Thus, it is expected that long-term increases in cyanobacterial abundance in the estuary would negatively affect ecosystem health and the food web function, irrespective of the presence of toxins. As noted by Ger et al. (2016), however, there are numerous knowledge gaps and biases in the existing literature, and better designed lab and field experiments are needed to separate co-evolution from the effects of individual factors that are often tested on relatively short time-scales. While not clearly defined in the estuary, recent studies in Lake Erie document the profound and often surprising effects of *Microcystis* blooms on food web structure, including high levels of microcystin accumulation prey-fishes, leading to further food web effects (Briland et al. 2020). While effects from *Microcystis* are particularly well documented, note that other HAB genera, both freshwater and marine, can negatively affect food webs through both direct (toxicit) and more subtle indirect responses, highlighting the need to better understand the consequences of increasing HAB probability in the estuary (Burkholder et al. 2018).

THE FUTURE OF HARMFUL ALGAL BLOOMS IN THE SAN FRANCISCO BAY-DELTA SYSTEM

Future Projections

The estuary is one of the best-studied estuaries in the world, providing a rich historical and scientific baseline for understanding how the ecosystem has changed through time (Cloern 1996; Cloern and Jassby 2012; NRC 2012; Cloern et al. 2017; Luoma and Muscatine 2019), and it is clear that the long-term health of the system is in decline (e.g., Sommer et al. 2007; Healey et al. 2008; Glibert et al. 2011; Sutula et al. 2017). It is also well documented that cyanoHABs emerged as a serious threat in the last few decades (Lehman et al. 2005). For marine HABs, it is more difficult to determine long-term trends given the paucity of data, but many potentially harmful algal genera have been identified in the SFB for decades (e.g., Cloern and Dufford 2005), and the

SFB is directly connected to patterns and processes occurring along the open coast (Cloern et al. 2007, Raimonet and Cloern 2017).

There is no question that this system, along with all aquatic ecosystems, will continue to evolve under the pressures of both climate-driven changes as well as global change (e.g., increased human populations leading to increased cultural eutrophication; Wells et al. 2015), and that this will directly and indirectly affect HABs. Accurately predicting these changes, or even identifying a global increase in HAB events in response to climate (Hallegraeff et al. 2021), is challenging at best (Wells et al. 2015, 2020). It is much more tractable to predict future changes of the internal and external drivers that influence HABs in the estuary system. Given that many of the dominant ecological processes in the estuary are controlled by flow and temperature—including HABs—considerable effort has been made to predict how the system will respond in the future. Another factor to consider is that HABs in the SFB are connected to the coastal ocean (Sutula et al. 2017), linking long-term changes in the Pacific to the fate of the SFB. California's Fourth Climate Change Assessment provides estimates of important drivers for the region (Ackerly et al. 2018). This includes projections of regional warming of 2 to 4 °C by 2050; an increase in frequency, intensity, and duration of marine heat waves; increasing high-precipitation events; and increasing sea levels, all of which could influence HABs and their penetration upstream.

Hallegraeff et al. (2021) caution against making broad assertions about increases in HAB events globally but provide evidence that at regional scales increases are evident. Within the California Current System one such change has been in the magnitude and intensity of *Pseudo-nitzschia* events (McCabe et al. 2016; McKibben et al. 2017). These events have been directly linked to climate change, and more specifically to increasing temperature (McKibben et al. 2017; Trainer et al. 2020). As blooms increase along the outer coast, a reasonable expectation is that blooms or bloom potential will also increase within the SFB (Sutula et al. 2017), although it is also possible that the SFB will become too warm to support some HAB organisms (Kudela et al. 2020). Other predicted changes such as more frequent and severe marine heat waves (Laufkötter et al. 2020) provide ideal conditions for HAB organisms to emerge from background concentrations when the physical environment (e.g., stratification, temperature) becomes favorable (Cloern et al. 2005). In freshwater, the increasing frequency and intensity of drought conditions (Cayan et al. 2009; Dettinger et al. 2016) will also exacerbate cyanoHAB events (Lehman et al. 2020).

Knowles and Cronkite–Ratcliff (2018) predict that a combination of sea level rise and impaired flow will result in further intrusion of saltwater upstream, potentially expanding the suitable habitat for marine HABs. At the same time, extremes in drought and high-flow events will increase. Past evidence suggested that wet years may favor cyanobacteria other than *Microcystis* while dry years exacerbate the presence of cyanoHABs (Mioni et al. 2011; Lehman et al. 2013, 2017), but more recent evidence demonstrates that, once established, *Microcystis* is persistent across extremes in wet and dry years (Lehman et al. 2020), likely as a result of the formation of refugia in backwaters during high-flow years. This is also consistent with Paerl et al. (2016) who identified a "perfect storm" scenario for cyanoHABs wherein excessive episodic rainfall events, followed by droughts, promote large nutrient input pulses followed by lengthy residence times, enabling blooms to develop and proliferate. The combination of increased temperature and nutrients may also result in larger and more toxic cyanoHAB events (cf. Paerl and Barnard 2020). Finally, long-term trends of increasing water clarity caused by a combination of reduced sediment load, invasive filter feeders, and changes in flow (Beck et al. 2018; Bever et al. 2018) may alleviate light limitation of HABs, a key controlling mechanism for the SFB and estuary in our current environment (Cole and Cloern 1984; Cloern 1999; Lehman et al. 2013; Strong et al. 2021).

While the SFB and estuary are not currently considered nutrient limited, efforts to greatly reduce nutrient inputs could result in precipitation and flow becoming even more dominant drivers in the future. Knowles and Cronkite–Ratcliff (2018) assumed that water management goals and actions would remain the same in the face of a changing climate, but many management options could be implemented (cf. Sommer 2020). Some, such as increased use of gates and barriers, may have unintended consequences for HABs through modification of retention time and flow, leading back to the "perfect storm" scenario. While Kimmerer et al. (2019) found that a temporary drought barrier installed on the False River in 2015 did not contribute to HABs, a follow-on study (Hartman et al. 2021) that compared the drought years 2015 and 2021 identified localized toxic *Microcystis* blooms in Franks Tract in 2021 that may be attributed to increased temperature and retention with the installation of a temporary drought barrier, even though at the landscape scale the years with a barrier (2015, 2021) did not enhance HAB events relative to years without a barrier in place.

Regardless of the management actions taken, it is improbable that HABs will decrease under predicted climate scenarios, although the specific organisms may change as optimal habitat shifts, and cyanoHABs in particular will be exacerbated in almost any scenario (Burford et al. 2020; Lehman et al. 2020; Paerl and Barnard 2020). It is difficult to predict which HAB organisms will be winners or losers in response to climate change (Wells et al. 2015; Wells and Karlson 2018). However, based on historical data it is clear that HAB organisms in the SFB and upper estuary generally increase with increasing total biomass (Lehman et al. 2008; Sutula et al. 2017).

Case Studies from the San Francisco Estuary

Consideration of hydrology and flow must consider the potential effect of changing climate but also management decisions, such as the opening of the South Bay Salt Pond System. The Bay–Delta has been extensively modified by a series of dams and levees, which in conjunction with consistent freshwater withdrawal has led to significant changes in the natural hydrological regime of the system (NRC 2012) California's San Francisco Bay Delta Estuary remains biologically diverse and functions as a central element in California's water supply system. Uncertainties about the future, actions taken under the federal Endangered Species Act (ESA. Furthermore, global climatic changes have resulted in protracted droughts, increased surface temperatures, and intensified water-column stratification (cf. Paerl and Paul 2012; Wells et al. 2015). These processes reduce freshwater flow, resulting in increased water residence times and opportunities for the relatively slow-growing cyanoHABs to outcompete faster-growing and generally less problematic eukaryotic algae, e.g., diatoms, chlorophytes, and flagellates (Paerl and Otten 2013).

Successive wet and dry years in the region have resulted in a more nuanced understanding of how flow regulates *Microcystis* blooms. Extreme drought conditions in 2014 resulted in the highest biomass, toxin concentration, and longest duration of *Microcystis* blooms since its appearance in 1999 (Lehman et al. 2017). The bloom was correlated with extreme water temperatures, long residence time, elevated nutrients, and both low stream flow and reduced agricultural diversions. After 2014, blooms reappeared in 2015 when the drought had intensified, with both drier and warmer conditions (Lehman et al. 2018). Based on 2014, the expectation was that the blooms would be even more severe, but 2015 was at least an order of magnitude smaller than 2014 (Lehman et al. 2017) even though median water-quality parameters were similar between years (Lehman et al. 2018). In 2015, primary correlates were water temperature, outflow, and the distance landward from the Pacific Ocean where the bottom salinity is 2 (X2). The general conclusion was that the structure and function of the estuary was fundamentally different between years, with somewhat subtle changes in surface vs. subsurface *Microcystis* colonies, and shifts in the community assemblage (Otten et al. 2017; Lehman et al. 2018).

After these successive droughts, 2017–2018 was a record wet year. The expectation was that high flow and reduced temperatures would flush the estuary, resulting in greatly reduced *Microcystis* biomass and toxin production. However, this hypothesis was rejected when *Microcystis* remained persistent even during an extreme wet year (Lehman et al. 2020), although it was also noted that the persistent blooms occurred in retentive or isolated parts of the estuary. Water temperature and the position of X2 again emerged as important controlling factors. Consistent with the discussion above, it was hypothesized that highly retentive backwaters and their sediments could be a persistent source and refuge for *Microcystis* during a high-flow year (Lehman et al. 2020). The overall conclusion was that once *Microcystis* and other cyanoHABs are established in the estuary, blooms will persist so long as other limiting factors such as temperature and nutrients remain conducive to growth.

A less well documented example of hydrological modification enhancing HABs is related to the reconnection of the South Bay salt ponds to the main circulation of the SFB (Grenier and Davis 2010). Sampling in South Bay Salt Pond A18 during 2006 (Thébault et al. 2008) revealed the presence of six phytoplankton taxa that can potentially cause harmful algal blooms: *Alexandrium* spp. and *Karenia mikimotoi*, the raphidophyte *Chattonella marina*, and cyanophytes *Anabaenopsis* spp. and *Anabaena* spp. Microscopic analysis of samples collected by USGS monitoring in 2006 and 2008 revealed seven additional species of phytoplankton that, at bloom abundances in other shallow marine systems, have disrupted aquatic food webs; caused mortality of invertebrates, fish, and birds; or human illness. These include *Karlodinium veneficum*, *Chattonella marina*, and *Heterosigma akashiwo*. These organisms are predominantly found in the South Bay and may be emerging threats. There are relatively few observations before the opening of the salt ponds but analysis of the historical USGS data (Figure 3) strongly suggests that the sudden appearance of *K. mikimotoi* and *K. veneficum* shortly after the initiation of the South Bay Salt Pond Restoration Project is not coincidental. Both organisms have been gradually spreading throughout the South Bay, suggesting that the salt ponds are analogous to the retentive backwaters that promote the persistence of cyanoHABs in the estuary. Given the persistence of these marine HABs, it is also likely that, once established, these HABs will be a persistent part of the SFB ecosystem. Taken together, these unexpected outcomes highlight the difficulty of predicting exactly how the estuary will respond to short- and long-term climate and global change, and is an important reminder that future projections, while useful, are not infallible.

Comparison with Chesapeake Bay

It is important examine the management actions and resulting trends in Chesapeake Bay as a comparative system for the San Francisco Bay and Estuary. Doubled nitrogen loadings from 1945 to 1980 resulted in increases in total phytoplankton biomass and reduced water clarity, followed by moderate improvements after aggressive nutrient management was introduced in the early 1980s (Harding et al. 2016). Nutrient reductions have led to increasing sensitivity to wet vs. dry years (Harding et al. 2015, 2016), with wet years generally associated with more diatom blooms. At the same time, salinity intrusions and warming are predicted to allow expansion and persistence of the marine HAB organisms *Karlodinium* and *Prorocentrum* in the Chesapeake (Li et al. 2020) while *Microcystis* persists in the freshwater regions, with downstream contamination of fish and shellfish (Bukaveckas et al. 2018), comparable to the SFB (Peacock et al. 2018). Najjar et al. (2010) generalized the expected effects of climate change on the Chesapeake Bay system and concluded that future conditions would be very similar to the estuary system, with increased heat waves and precipitation intensity, increased summertime stratification, increasing phytoplankton production, and increasing HAB events, but cautioned that the trajectory of these changes is highly variable, depending on factors under direct human control (such as emissions of CO_2 and nutrient loads). In the Chesapeake, aggressive nutrient management, enhanced salinity intrusion, and increased sensitivity to wet and dry years amplified HAB events in intensity, persistence, and toxicity while shifting environmental conditions both spatially and temporally (expanding the ecological window of opportunity [Moore et al. 2009]). While SFB is not the Chesapeake Bay, the documented changes in that system perhaps illustrate how other mid-latitude estuarine systems could respond.

Kennish (2021) summarized expected changes in coastal and estuarine environments in response to climate change and included several relevant summary points. First, estuaries are disproportionately affected through the combination of increasing human population and coastal development. Second, multiple drivers can act synergistically with non-linear and cascading responses, making specific predictions difficult. Third, management options should focus on the resilience of the entire ecosystem. Wainger et al.

(2017) used the Chesapeake Bay as an example for how to improve system resilience to improve multiple metrics, including HABs. They concluded that estuarine restoration generates multiple ecosystem service benefits, but only a small fraction of them can be monetized, given lack of data and knowledge on how linear and non-linear interactions function. They caution that reliance on regulation of Total Maximum Daily Loads (TMDLs) may not result in gradual improvement of the system. However, TMDLs may enhance resilience to novel or amplified pressures—such as those expected from climate change—by reducing stress in key ecosystem components such as submerged aquatic vegetation and fish populations, leading to increased resilience to novel stressors. A key take-home message from Chesapeake Bay is that there are no simple solutions and there will be unexpected consequences, but minimizing stressors that are controllable at the regional level such as nutrient loads will have long-term tangible and intangible benefits, including reducing the threat of HABs.

MANAGEMENT RESPONSES, RECOMMENDATIONS, AND KNOWLEDGE GAPS

Current Monitoring Efforts

Marine HABs have long been of concern in California, and there are numerous monitoring programs and regulatory guidelines in place to manage and mitigate deleterious consequences of HABs in California and more broadly in the US (e.g., Lewitus et al. 2012; Frolov et al. 2013; Kudela et al. 2015a, 2015b; Anderson et al. 2021). With increasing recognition that freshwater HABs are also a serious issue (e.g., Lehman et al. 2005, 2010, 2020; Fetscher et al. 2015; Brooks et al. 2017; Urquehart et al. 2017), Assembly Bill 834 established a freshwater and estuarine HAB program (FHAB Program) through the California State Water Resources Control Board. The FHAB program has established an assessment and support strategy that includes three components: response to HAB events, field assessment and ambient monitoring programs, and risk assessment for potential HAB events (Anderson–Abbs et al. 2016). The California State Water Resource Control Board emphasizes effective communication and meaningful participation by California Native American Tribes, whether federally recognized or not. Recently, the FHAB program developed a monitoring strategy that includes six recommended actions that the California State Water Boards and other federal, state, and local agencies, stakeholders, and the California Native American Tribes, who have sovereign jurisdiction over their waterways, could take to implement a statewide monitoring program that would complement and coordinate with existing statewide marine and estuarine HAB monitoring efforts (Smith et al. 2021a).

Within the estuary, there are nutrient management and research strategic plans to develop a long-term approach to address nutrient effects, including HABs, and to identify and fill data gaps. SFB has a nutrient management strategy that documents the technical studies required to support nutrient-management decisions (Feger et al. 2012), and has developed an organizational structure, comprising several committees and advisory groups of stakeholders, regulators, and scientists to implement the strategy (Smith et al. 2021a). Similarly, the Delta has a Nutrient Research Plan adopted by the Central Valley Water Board (Resolution No. R5-2018-0059) which highlights the data gaps and research and monitoring needs to address nutrient and related HAB issues in the Delta (Cooke et al. 2018).

Management actions to address nutrient issues in the Delta have included regulatory requirements to upgrade wastewater treatment through National Pollutant Discharge Elimination System (NPDES) permits. In 2000, virtually none of the Delta WTPs provided nutrient removal. Over the past 20 years, the Delta municipalities have been upgrading their WTPs to provide advanced treatment such as tertiary filtration and biological nutrient removal (BNR) to meet new disinfection and nutrient-removal requirements in their NPDES permits. The Sacramento County Regional Sanitation District, which makes up about 60% of the total WTP discharge flow to the Delta, recently completed construction of BNR facilities that resulted in a substantial reduction of nutrients to the Delta. The City of Stockton has also implemented similar upgrades to their WTPs to meet current NPDES requirements. Today 99.8% of wastewater discharged to the Delta is treated to remove nutrients. While all these efforts represent important steps forward, there has been a lack of coordination amongst the various local, state, and federal regulatory agencies to develop clear and consistent management guidelines and monitoring activities to address the interconnected freshwater-to-marine continuum.

For the estuary, considerable monitoring efforts already exist for HABs but are arguably not well coordinated. The California Department of Water Resources (CDWR) currently conducts monitoring which routinely samples many of the variables of interest known to influence cyanoHABs—including turbidity, temperature, and total phytoplankton biomass—but toxins are not routinely collected (Berg and Sutula 2015). In SFB, the USGS and the CDWR routinely collect the same environmental data (with the addition of nutrients) and began toxin testing for DA and microcystins in late 2011 (Peacock et al. 2018). The CDWR also routinely monitors nutrients in the system. The San Francisco Estuary Institute (estuaryI) has attempted to provide a broader, freshwater-to-marine perspective, primarily through a series of special projects as part of the San Francisco Bay Nutrient Management Strategy. This has included support for the USGS toxin sampling, method development using new monitoring tools such as the Imaging Flow CytoBot (IFCB; Cloern 2018) and molecular methods (Otten et al. 2017), and the development of a California-wide satellite remote sensing tool for cyanoHABs in collaboration with NOAA and California State Water Boards as part of the Freshwater Harmful Algal Bloom Program (*https://fhab.sfei.org/).* Numerous short-term research projects funded by multiple agencies and programs also focus on HABs in the region. This includes multiple regional monitoring programs in both the SFB and Sacramento–San Joaquin Delta (SFB Regional Monitoring Program, Delta Regional Monitoring Program, Interagency Ecological Program, Delta Science Program) that include a HAB component; however, most of these efforts are uncoordinated and lack appropriate funding to produce long-term data sets. The challenge remains to integrate these disparate monitoring efforts into a coordinated action plan with an associated cohesive and comprehensive monitoring approach to HABs.

Mitigation Strategies

Mitigation of HABs is a primary goal for multiple agencies, including NOAA, which has an active program focused on Prevention, Control, and Mitigation of HABs (PCMHAB; Fed Regist 2009). There are also several excellent reviews of mitigation strategies for both freshwater and marine HABs (e.g., Paerl et al. 2016; Huisman et al. 2018; Sellner et al. 2018; Pal et al. 2020). Methods can be broadly divided into physical (flocculation, mixing, harvesting), chemical (e.g., copper, barley straw, hydrogen peroxide), and biological (e.g., macroalgae, introduced grazers, bacteria, and viruses) control for both freshwater and marine blooms (Huisman et al. 2018; Kidwell 2015). All of these methods have been used successfully in some environments (typically small lakes; Huisman et al. 2018) but are very much location-specific. For the estuary, most of these methods are impractical given the size of the estuary and the cost associated with implementation, except for targeted sites such as Discovery Bay and the Stockton Basin, where increased physical mixing or reductions in retention time could reduce these chronically toxic regions (Figure 5B).

Beyond these targeted remediation strategies, two successful examples of mitigating HABs include the Chesapeake, where aggressive nutrient reduction and estuarine restoration appear to be viable (see Comparison with Chesapeake Bay) and the Great Lakes, where a combination of reductions in total N and P (Steffen et al. 2014; Paerl et al. 2020) and effective monitoring and forecasting through the use of satellite remote sensing and hydrological models (Stumpf et al. 2016) have also proven useful. In both cases, the most effective long-term solution focused on reducing nutrient loads as the most tractable for mitigation of an entire ecosystem. Nutrient reduction is already ongoing in the estuary (see "Salinity and Temperature"). Thus, a long-term strategy may be to monitor and manage the overall eutrophic status of the estuary ecosystem, particularly through the lens of nutrient loads (Cloern et al. 2020) since this is one driver that can be regulated by human activity at a regional scale.

On shorter time-frames, development of an integrated monitoring and forecasting system is possible but does not yet exist for the estuary. Immediately offshore, the California Harmful Algae Risk Mapping system (C-HARM; Anderson et al. 2016) successfully forecasts DA risk but has not been downscaled to the SFB. The EPA also supports the Cyanobacteria Assessment Network (CyAN; see also the California CyanoHAB Network of the California Water Quality Monitoring Council; *https://mywaterquality.ca.gov/habs/data_viewer/*), which provides satellite remote sensing data of cyanobacterial biomass for the contiguous United States inland waters, but large parts of the estuary are not applicable given the spatial resolution and the physical structure of the river and channel network (Schaeffer et al. 2020). The estuary is data-rich (Davis et al. 2006; Trowbridge et al. 2016), and there are examples of data frameworks that would support integration of HABs into water-quality forecasts (e.g., the USGS Next Generation Water Observing System; Eberts et al. 2019). Examples also exist where observational data are combined with new statistical methods to produce risk assessments (Myer et al. 2021; Beck et al. 2022); these tools could be developed for the estuary focusing specifically on HABs if coordination and funding were sufficient to implement such a strategy.

Management Recommendations

A variety of recommendations, plans, and strategies directly or indirectly provide guidance for a comprehensive HAB program that are applicable to the estuary system (Berg and Sutula 2015; Anderson–Abbs et al. 2016; Holleman et al. 2017; Paerl et al. 2018; Smith et al. 2021a, 2021b; Howard et al. 2022). The FHAB Program Monitoring Strategy makes clear recommendations at the state level (Figure 7; Smith et al. 2021a). The statewide approach (Figure 7) has several components, including ambient monitoring, incident response, special studies (*shown in green*), monitoring partners (*shown in blue*), and infrastructure to support the program (*shown in gray*). These components produce monitoring products that include predictive models; assessment of status, trends and drivers; and data for public health protection, all of which inform management decisions and actions to protect beneficial uses of water (recreation, fish consumption, aquatic life, drinking water, and Tribal uses). The six main recommendations of the FHAB Monitoring Strategy include (1) developing and implementing coordinated partner monitoring, (2) strengthening the remote sensing program, (3) implementing field surveys to protect human health, (4) conducting focused assessments of HAB drivers, (5) strengthening and continuing bloom event response program, and (6) integrating HAB monitoring elements into California State Water Board programs, permits, and policies. Of relevance to the estuary system are the coordinated partner monitoring, remote sensing program, focused assessments, and field survey recommendations, as well as the monitoring products and management actions. The goal of coordinated partner monitoring is to promote and support local-level monitoring efforts to leverage the limited resources available and to establish a shared monitoring framework. The partners can be California Native American Tribes, local health departments, parks and recreational organizations, drinking water agencies, waterbody managers, scientific non-governmental organizations, and science groups. HAB status, trends and drivers, and predictive models of HAB occurrence and drivers collectively will produce assessments of freshwater HABs and field information to protect public health and inform management actions (Smith et al. 2021a).

Paerl et al. (2018) introduced recommendations for a holistic approach to mitigate HAB expansion across the freshwater-to-marine continuum. Most HAB monitoring programs are not designed to capture the transfer of HAB organisms and toxins because they generally

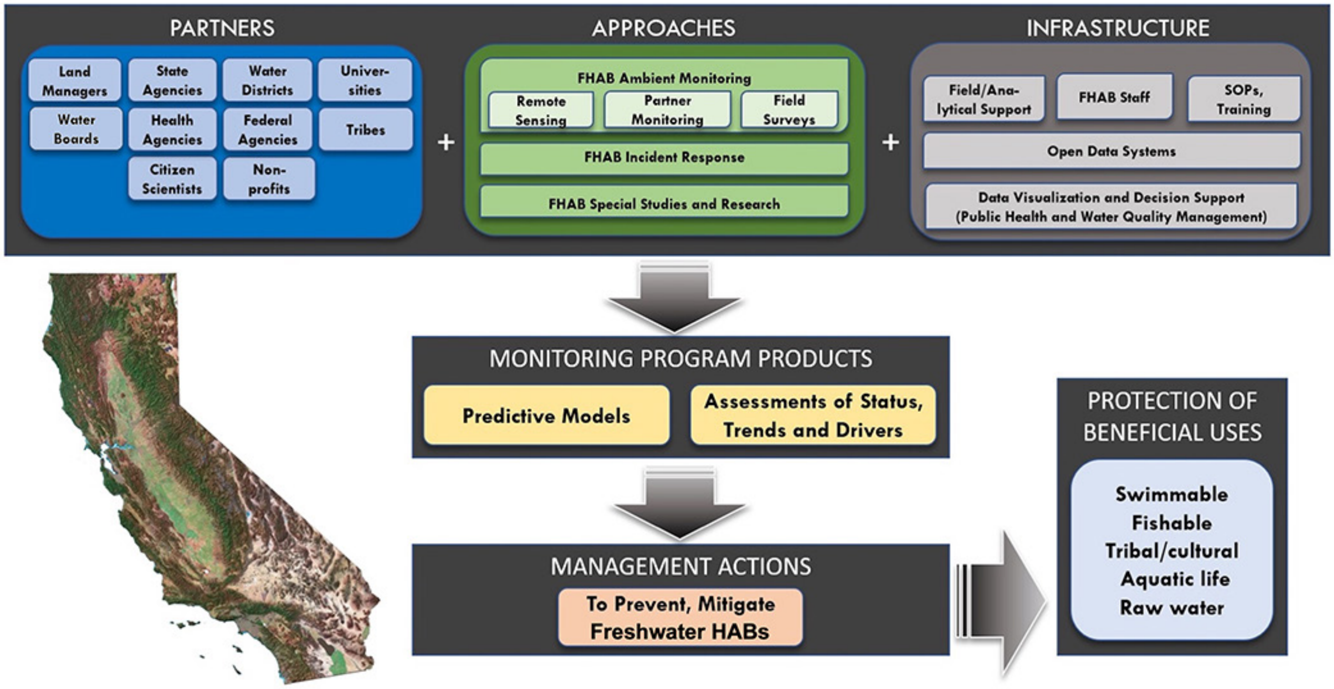


Figure 7 Proposed freshwater HAB management strategy for the State of California. Adapted from Smith et al. 2021a

focus on a single waterbody or a few waterbodies within the larger watershed or ecosystem. Management and mitigation strategies need to focus holistically on the watershed, be inclusive of all hydrologically interconnected waterways, and treat the freshwater-to-marine continuum as one interconnected system (Paerl et al. 2018). Additionally, Paerl et al. (2018) emphasized the need to reduce both N and P inputs for long-term mitigation of HABs, and that mitigation efforts downstream will be unsuccessful without coordinated efforts upstream. This is relevant to the estuary system because most of the programs described above are uncoordinated, yet there is a clear hydrologic connectivity between SFB and the Delta. Monitoring and management efforts have not been applied to the whole interconnected system, but rather to artificial boundaries that reflect organizational, jurisdictional, or perceived ecological boundaries or funding sources. Howard et al. (2022) built upon the Paerl et al. (2018) concepts and guidance, and developed more detailed recommendations for HAB monitoring across the freshwater-to-marine continuum to inform management and mitigation strategies, and to address the physical and hydrological challenges encountered across the freshwater-to-marine continuum.

Three overarching recommendations emerge from this analysis which can be tiered from short- to long-term goals: (1) **a combination of sample types and matrices should be employed** to effectively monitor HAB toxin dynamics and transport across the watershed, (2) **multiple toxins should be routinely monitored from these samples**, and cyanotoxins should be included in estuarine and marine monitoring efforts. (3) **HAB monitoring should be coordinated and cohesive** across hydrologically interconnected waterbodies, and should cross organizational and political boundaries and jurisdictions.

Traditional HAB monitoring programs rely on grab samples, which severely underestimate ephemeral or episodic events; when those data are compared to more comprehensive or integrated sampling, HABs and toxins can be

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perceived to be rarely present when they are ubiquitous (Fetscher et al. 2015; Peacock et al. 2018; Tatters et al. 2019, 2021). One cost-effective method that has been employed in both the SFB and more recently the upper estuary is the use of passive samplers such as Solid Phase Adsorption Toxin Tracking, or SPATT (Mioni et al. 2012; Kudela 2017; Peacock et al. 2018). SPATT has proven particularly useful as a complement to traditional grab samples because it provides an integrative measurement of the presence of toxins that grab samples cannot. SPATT deployed in areas that otherwise would be under-sampled (as because of resource limitations, difficult access, hydrological challenges, or other reasons) significantly expands HAB toxin surveillance (Gibble and Kudela 2014; Howard et al. 2017; Peacock et al. 2018; Tatters et al. 2019). A significant barrier to broader adoption of SPATT and other passive samplers is that the data are not directly comparable to grab samples, and therefore cannot easily be used for regulatory thresholds (Kudela 2017). While efforts are underway to calibrate passive samplers (e.g., D'Angelo 2019; Yao et al. 2019; Wang et al. 2022), technical hurdles remain, and it would be more effective to use multiple sampling modalities.

It is now accepted that multiple toxins exist throughout the freshwater-to-marine continuum. Many cyanobacteria species can produce one or more toxins (cf. Berg and Sutula, 2015), resulting in mixtures of different cyanotoxins and metabolites, many of which are not fully documented but which may result in deleterious effects (Otten et al. 2017). The frequency of co-occurrence of cyanotoxins is not well defined since the number of comprehensive toxin surveys that measure multiple cyanotoxins are still limited; even fewer studies look for the presence of freshwater and marine toxins (Peacock et al. 2018; Tatters et al. 2019, 2021).

The lack of comprehensive and coordinated monitoring is not unique to this system, and better coordination would benefit multiple programs and goals focused on beneficial use and water quality. This first recommendation requires multiple agencies and organizations to work together using the same standard operating procedures and quality assurance plans.

Second, a tiered monitoring program could be implemented by combining satellite remote sensing (Zhang et al. 2015; Urquhart et al. 2017) to monitor status and trends synoptically with deployment of passive samplers (Mioni et al. 2012; Kudela 2017; Peacock et al. 2018) at key locations, and traditional grab sampling for regulatory purposes to inform management action. A significant advantage of this approach is that the combination of remote sensing and passive sampling is ideal for development of historical trends and predictive models (Kudela 2011; Kudela et al. 2015a; Schaeffer and Myer 2020).

Based on the above strategies and publications, there are three management recommendations for the estuary:

1. **Develop a comprehensive HAB monitoring and management strategy for the estuary ecosystem.** The strategy should include shared science and management goals and should be based on the relevant strategies and recommendations provided from other sources. First, HAB monitoring should be holistically coordinated throughout the estuary ecosystem to characterize one interconnected system. This coordinated approach should include multiple organizations and partners to leverage all available resources in the estuary. Second, multiple approaches that include a combination of in situ sampling with satellite-based remote sensing should be included to provide a comprehensive picture of HABs and toxins. The in situ sampling should include multiple sample types and measurements to characterize effects on beneficial uses such as recreational contact and harvesting for human consumption. Third, multiple toxins should be measured and should include all relevant and potential toxins present (Table 1), including both marine toxins and cyanotoxins.

2. **Both short and long-term management and mitigation approaches should be developed.** Shorter-term

mitigation approaches and technologies should be developed and should be the focus of future funding efforts. These short-term mitigation approaches can help alleviate annual hotspots such as Stockton and Discovery Bay. Additionally, federal and state regulatory agencies such as the U.S. Fish and Wildlife Service and the California Department of Fish and Wildlife should be included in this effort to identify additional information or special studies needed to develop a permitting approach for successful implementation of these mitigation approaches. To inform management actions, appropriate long-term approaches to mitigate HABs include the reduction of both N and P as well as implementation of a coordinated monitoring program that can provide a comprehensive data set across the freshwater-to-marine continuum. The reduction of nutrient loads from WWTPs is likely the most effective long-term mitigation strategy for SFB given the clear linkage between increasing biomass and increasing HABs (Sutula et al. 2017), while nutrient reductions from multiple loading sources may be necessary for the upper estuary (Sutula et al. 2017; Cloern et al. 2020; Saleh and Domagalski, 2021). While a nutrient mitigation strategy would not be cost-effective for HABs alone, reduction in the potential for increased HAB events is certainly one potential positive outcome.

3. **Water quality objectives and health standards should be developed for toxins that are not well regulated, and for exposure to multiple toxins.** While some federal water-quality criteria and guidance exist (Table 2), there is a lack of regulatory standards and objectives for multiple routes of exposure of cyanotoxins (see Regulatory Monitoring and Guidance). In addition, all existing water-quality criteria and thresholds are based on exposure to a single toxin; however, multiple toxins have been documented in SFB (Peacock et al. 2018), while there is a lack of observational data on multiple toxins in the upper estuary. Given the lack of observational data, it is not surprising that the effect of simultaneous exposure to multiple toxins is also poorly documented. While the effects may be additive given that disparate toxins have differing modes of action, there is growing evidence that combinations of toxins can be more intense than exposure to individual toxins. For example, Chia et al. (2019) demonstrated that exposure to microcystins and anatoxin-*a* simultaneously exerts a strong selective pressure on phytoplankton community composition, promoting or inhibiting chlorophytes vs. cyanobacteria. Alarcan et al. (2018) similarly reported that marine lipophilic toxins in combination can have both synergistic and antagonistic effects but are poorly defined. For marine toxins common to SFB, studies suggest no or unknown synergistic effects for simultaneous exposure to DA and brevetoxin in bottlenose dolphins (Fire et al. 2011) or brevetoxins, okadaic acid, and saxitoxin in manatees and turtles (Capper et al. 2013). The combined effect, even if not synergistic, likely results in a reduction in marine mammal fitness through immunosuppression (Capper et al. 2013). Of additional concern are the unknown effects of simultaneous exposure to cyanotoxins and phycotoxins in the presence of microplastics, heavy metals, and pesticides (Metcalf and Codd 2020). There is a clear need for controlled studies to understand how these stressors interact, if at all; however, from a monitoring and management perspective, a simple first step is to routinely assess or monitor for the presence of multiple stressors and use the data to develop appropriate health-based thresholds for multiple toxin and stressor exposure for humans and wildlife.

Knowledge Gaps

There are multiple knowledge gaps in our understanding of HABs globally (e.g., Burford et al. 2020; Wells et al. 2015, 2020). There are also several critical gaps for the estuary specifically. First and foremost, there is a lack of consistent monitoring of both phytoplankton and toxins. While phytoplankton have been regularly quantified by several agencies (including USGS and IEP), underlying technical issues with how the counts were completed introduce undesirable uncertainty in the historical data sets. Second, the fate and transfer of toxins within the airshed, watershed, and food web are largely unknown, leading to large gaps in our understanding of potential exposure routes and effects on both the ecosystem and humans. This is particularly important for this system if we consider the estuary as a freshwater-to-marine continuum, with effects and consequences for nutrient loading or toxicity, for example, often far removed from source locations. This is exacerbated by the fractionated and overlapping interests of the various regulatory and monitoring agencies. There are also clear opportunities. The estuary has numerous monitoring programs and a wealth of historical and current environmental data. New tools are being developed regionally and nationally with clear potential application to the estuary ecosystem. The number one priority should be to effectively integrate these programs to create an effective monitoring and forecasting system that will inform decision-making now and into the future. Specific gaps and recommendations for addressing these needs include the following.

- There is a lack of data on the presence of multiple toxins and the implications of exposure to a suite of toxins. Consistently monitor for multiple toxin classes using a variety of sampling modalities and characterize the threat of chronic and acute toxin exposure to both animals and humans throughout the food web.

- There is conflicting evidence on how the food web (particularly the microbial web and grazers) will respond on short vs. long (evolutionary) time-scales to the presence of toxic algae. More experimental work needs to incorporate the potential for co-evolution.

- Long-term mitigation strategies include hydrological modification and reduction in nutrients, but it is not clear that our ability to predict environmental responses (e.g., Lehman et al. 2017; Kimmerer et al. 2019) is acceptable. Care should be taken in extrapolating past events to the future without better understanding of how these environmental drivers modulate HABs in the system.

- For the most part, HAB abundance models spanning the freshwater-to-marine continuum (Paerl et al. 2018) are lacking, as are watershed-scale models that focus on HABs. Nonetheless, it may be possible to leverage recent federal initiatives such as the NOAA Water Initiative and the USGS Next Generation Water Observing System to facilitate advances in this area.

- To improve predictability on short- and long-term time-scales, we require both observational data and additional lab and field experimentation, particularly for HAB organisms other than *Microcystis*, to inform mechanistic models that can be used with downscaled climate predictions to anticipate threats and identify potential mitigation strategies.

- These same research studies and models should holistically evaluate and quantify the magnitude of nutrient reduction required to limit HAB occurrence and severity in the estuary, and the potential effects to beneficial phytoplankton and resulting community composition in response to nutrient reductions.

SUMMARY AND CONCLUSIONS

SFB and the upper estuary have historically been resistant to acute HAB events, but this resistance is weakening (Sutula et al. 2017; Lehman et al. 2020). While HABs have not been a focus of most monitoring and management programs in the region, it is now clear that there is a wide array of marine and freshwater HABs in the region that are realized threats today and act as potential threats in the future. Under current conditions, HABs in both freshwater and marine ends of the system are strongly influenced by temperature, salinity, and irradiance, as well as circulation (residence time), all of which are changing rapidly. As these changes occur, nutrient concentration and form may become an increasingly important drivers; with the expected nutrient reductions associated with upgrades to the regional WTPs, nutrient forms and limiting nutrient concentrations, in particular, may become increasingly important as drivers of physiological response (e.g., toxin production) and community composition. Successfully predicting how HABs will respond to climate change and global change is complicated, particularly given the very limited number of modeling studies that specifically address HABs (Ralston and Moore 2020). Despite the current lack of quantitative predictability, HABs will remain an important part of the estuary system and are likely to increase in frequency, intensity, and duration.

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REFERENCES

Ackerly D, Jones A, Stacey M, Riordan B. 2018. San Francisco Bay Area summary report. California's Fourth Climate Change Assessment. Publication number: CCCA4-SUM-2018-005. [accessed 2022 Jul 22]. *https://www.energy.ca.gov/sites/default/files/2019-11/Reg_Report-SUM-CCCA4-2018-005_SanFranciscoBayArea_ADA.pdf*

Ajani P, Brett S, Krogh M, Scanes P, Webster G Armand, L. 2013. The risk of harmful algal blooms (HABs) in the oyster-growing estuaries of New South Wales, Australia. Env Mon Assess. [accessed 2022 Jul 22];185:5295–5316. *https://doi.org/10.1007/s10661-012-2946-9*

Alarcan J, Biré R, Hégarat LL, Fessard V. 2018. Mixtures of lipophilic phycotoxins: exposure data and toxicological assessment. Mar Drugs. [accessed 2022 Jul 22]16:46. *https://doi.org/10.3390/md16020046*

Allen WE. 1920. A quantitative and statistical study of the plankton of the San Joaquin. University of California Publications in Zoology. [accessed 2022 Jul 22];22:1–292. Available from: *https://www.biodiversitylibrary.org/item/44205#page/5/mode/1up*

Anderson CR, Berdalet E, Kudela RM, Cusack CK, Silke J, O'Rourke E, Dugan D, McCammon M, Newton JA, Moore SK, et al. 2019. Scaling up from regional case studies to a global harmful algal bloom observing system. Front Mar Sci. [accessed 2022 Jul 22];6:250. *https://doi.org/10.3389/fmars.2019.00250*

Anderson CR, Kudela RM, Kahru M, Chao Y, Rosenfeld LK, Bahr FL, Anderson DM, Norris TA. 2016. Initial skill assessment of the California harmful algae risk mapping (C-HARM) system. Harmful Algae. [accessed 2022 Jul 22];59:1–18. *https://doi.org/10.1016/j.hal.2016.08.006*

Anderson DM, Burkholder JM, Cochlan WP, Glibert PM, Gobler CJ, Heil CA, Kudela RM, Parsons ML, Rensel JJ, Townsend D, et al. 2008. Harmful algal blooms and eutrophication: examining linkages from selected coastal regions of the United States. Harmful Algae. [accessed 2022 Jul 22];8(1):39–53.

Anderson DM, Fensin E, Gobler CJ, Hoeglund AE, Hubbard KA, Kulis DM, Landsberg JH, Lefebvre KA, Provoost P, Richlen ML, Smith JL. 2021. Marine harmful algal blooms (HABs) in the United States: history, current status and future trends. Harmful Algae. [accessed 2022 Jul 22];102:101975. *https://doi.org/10.1016/j.hal.2021.101975*

Anderson DM, Glibert PM, Burkholder JM. 2002. Harmful algal blooms and eutrophication: nutrient sources, composition, and consequences. Estuaries. [accessed 2022 Jul 22];25(4):704–726. *https://doi.org/10.1007/BF02804901*

Anderson CR, Siegel DA, Kudela RM, Brzezinski MA. Empirical models of toxigenic Pseudo-nitzschia blooms: potential use as a remote detection tool in the Santa Barbara Channel. Harmful Algae. [accessed 2022 Jul 22];8(3):478-92. *https://doi.org/10.1016/j.hal.2008.10.005*

Anderson–Abbs B, Howard M, Taberski K, Worcester K. 2016. California freshwater harmful algal blooms assessment and support strategy. SWAMP-SP-SB-2016-0001. [accessed 2022 Jul 22]. Available from: *https://www.waterboards.ca.gov/rwqcb2/water_issues/programs/SWAMP/HABstrategy_phase%201.pdf*

Backer L, Carmichael W, Kirkpatrick B, Williams C, Irvin M, Zhou Y, Johnson T, Nierenberg K, Hill V, Kieszak S, et al. 2008. Recreational exposure to low concentrations of microcystins during an algal bloom in a small lake. Mar Drugs. [accessed 2022 Jul 22];6:389–406. *https://doi.org/10.3390/md6020389*

Backer LC, McNeel SV, Barber T, Kirkpatrick B, Williams C, Irvin M, Zhou Y, Johnson TB, Nierenberg K, Aubel M, et al. 2010. Recreational exposure to microcystins during algal blooms in two California lakes. Toxicon. [accessed 2022 Jul 22];55:909–921. *https://doi.org/10.1016/j.toxicon.2009.07.006*

Backer LC, Landsberg JH, Miller M, Keel K, Taylor TK. 2013. Canine cyanotoxin poisonings in the United States (1920s–2012): Review of suspected and confirmed cases from three data sources. Toxins. [access 2022 Jul 22];5(9):1597-628. *https://doi.org/10.3390/toxins5091597*

[BACWA] Bay Area Clean Water Agencies. 2018. Potential nutrient reduction by treatment optimization, sidestream treatment, treatment upgrades, and other means. BACWA Nutrient Reduction Study final report. June 22, 2018. [accessed 2022 Jul 22]. Available from: *https://bacwa.org/wp-content/uploads/2018/06/BACWA_Final_Nutrient_Reduction_Report.pdf*

Barnard MA, Chaffin JD, Plaas HE, Boyer GL, Wei B, Wilhelm SW, Rossignol KL, Braddy JS, Bullerjahn GS, Bridgeman TB, et al. 2021. Roles of nutrient limitation on western Lake Erie CyanoHAB toxin production. Toxins. [accessed 2022 Jul 22];13(1):47. *http://www.doi.org/10.5281/zenodo.4281127*

Baxa DV, Kurobe T, Ger KA, Lehman PW, Teh SJ. 2010. Estimating the abundance of toxic Microcystis in the San Francisco Estuary using quantitative real-time PCR. Harmful Algae. [accessed 2022 Jul 22];9:342–349. *https://doi.org/10.1016/j.hal.2010.01.001*

Beck MW, de Valpine P, Murphy R. Wren I, Chelsky A, Foley M, Senn DB. 2022. Multi-scale trend analysis of water quality using error propagation of generalized additive models. Sci Tot Env. [accessed 2022 Jul 22];802:149927. *https://doi.org/10.1016/j.scitotenv.2021.149927*

Beck MW, Jabusch TW, Trowbridge PR, Senn DB. 2018. Four decades of water quality change in the upper San Francisco Estuary. Estuarine Coast Shelf Sci. [accessed 2022 Jul 22];212:11–22. *https://doi.org/10.1016/j.ecss.2018.06.021*

Berdalet E, McManus MA, Ross ON, Burchard H, Chavez FP, Jaffe JS, Jenkinson IR, Kudela R, Lips I, Lips U, et al. 2014. Understanding harmful algae in stratified systems: review of progress and future directions. Deep Sea Res II. Topical Studies Oceanogr. [accessed 2022 Jul 22];101:4–20. *https://doi.org/10.1016/j.dsr2.2013.09.042*

Berg GM, Driscoll S, Hayashi K, Kudela R. 2019. Effects of nitrogen source, concentration, and irradiance on growth rates of two diatoms endemic to northern San Francisco Bay. Aquat Biol. [accessed 2022 July 22];28:33-43. *https://doi.org/10.3354/ab00708*

Berg GM, Driscoll S, Hayashi K, Ross M, Kudela R. 2017. Variation in growth rate, carbon assimilation, and photosynthetic efficiency in response to nitrogen source and concentration in phytoplankton isolated from upper San Francisco Bay. J Phycol. [accessed 2022 Jul 22];53:664–679. *https://doi.org/10.1111/jpy.12535*

Berg M, Sutula M. 2015. Factors affecting the growth of cyanobacteria with special emphasis on the Sacramento–San Joaquin Delta. Southern California Coastal Water Research Project Technical Report 869. [accessed 2022 Jul 22]. Available from: *https://amarine.com/wp-content/uploads/2018/01/Cyano_Review_Final.pdf*

Bever AJ, MacWilliams ML, Fullerton DK. 2018. Influence of an observed decadal decline in wind speed on turbidity in the San Francisco Estuary. Estuaries Coasts. [accessed 2022 Jul 22];41(7):1943–1967. *https://doi.org/10.1007/s12237-018-0403-x*

Biré R, Bertin T, Dom I, Hort V, Schmitt C, Diogène J, Lemée R, De Haro L, Nicolas M. 2020. First evidence of the presence of anatoxin-a in sea figs associated with human food poisonings in France. Mar Drugs. [accessed 2022 Jul 22];18(6):285. *https://doi.org/10.3390/md18060285*

Boesch D, Burreson E, Dennison W, Houde E, Kemp–M, Kennedy V, Newell R, Paynter K, Orth R, Ulanowicz R. 2001. Factors in the decline of coastal ecosystems. Science. [accessed 2022 Jul 22];293:1589–1591. *https://doi.org/10.1126/science.293.5535.1589*

Bouma–Gregson K, Power ME, Bormans, M. 2017. Rise and fall of toxic benthic freshwater cyanobacteria (Anabaena spp.) in the Eel River: buoyancy and dispersal. Harmful Algae. [accessed 2022 Jul 22];66:79–87. *https://doi.org/10.1016/j.hal.2017.05.007*

Bouma–Gregson K, Kudela RM, Power ME. 2018. Widespread anatoxin-a detection in benthic cyanobacterial mats throughout a river network. PLoS One [accessed 2022 Jul 22];13(5):e0197669. *https://doi.org/10.1371/journal.pone.0197669*

Boyd PW, Rynearson TA, Armstrong EA, Fu F, Hayashi K, Hu Z, Hutchins DA, Kudela RM, Litchman E, Mulholland MR, et al. 2013. Marine phytoplankton temperature versus growth responses from polar to tropical waters–outcome of a scientific community-wide study. PloS ONE. [accessed 2022 Jul 22];8(5):e63091. *https://doi.org/10.1371/journal.pone.0063091*

Breinlinger S, Phillips TJ, Haram BN, Mareš J, Yerena JAM, Hrouzek P, Sobotka R, Henderson WM, Schmieder P, Williams SM, et al. 2021. Hunting the eagle killer: a cyanobacterial neurotoxin causes vacuolar myelinopathy. Science. [accessed 2022 Jul 22];371:aax9050. *https://doi.org/10.1126/science.aax9050*

Briand J, Leboulanger C, Humbert J, Bernard C, Dufour P. 2004. *Cylindrrospermopsis raciborskii* (cyanobacteria) invasion at mid-latitudes: selection, wide physiological tolerance, or global warming? J Phycol. [accessed 2022 Jul 22];40:231–238. *https://doi.org/10.1111/j.1529-8817.2004.03118.x*

Bricker SB, Longstaff B, Dennison W, Jones A, Boicourt K, Wicks C, Woerner J. 2008. Effects of nutrient enrichment in the nation's estuaries: a decade of change. Harmful Algae. [accessed 2022 Jul 22];8(1):21–32. *https://doi.org/10.1016/j.hal.2008.08.028*

Briland RD, Stone JP, Manubolu M, Lee J, Ludsin SA. 2020. Cyanobacterial blooms modify food web structure and interactions in western Lake Erie. Harmful Algae. [accessed 2022 Jul 22];92:101586. *https://doi.org/10.1016/j.hal.2019.03.004*

Brooks BW, Lazorchak JM, Howard MD, Johnson MVV, Morton SL, Perkins DA, Reavie ED, Scott GI, Smith SA, Steevens JA. 2017. In some places, in some cases, and at some times, harmful algal blooms are the greatest threat to inland water quality. Env Tox Chem. [accessed 2022 Jul 22];36(5):1125. *https://doi.org/10.1002/etc.3801*

Brown A, Foss A, Miller M.A, Gibson Q. 2018. Detection of cyanotoxins (microcystins/nodularins) in livers from estuarine and coastal bottlenose dolphins (*Tursiops truncatus*) from Northeast Florida. Harmful Algae. [accessed 2022 Jul 22];76:22–34. *https://doi.org/10.1016/j.hal.2018.04.011*

Brown LR, Kimmerer W, Conrad JL, Lesmeister S, Mueller–Solger, A. 2016. Food webs of the Delta, Suisun Bay, and Suisun Marsh: an update on current understanding and possibilities for management. San Franc Estuary Watershed Sci. [accessed 2022 Jul 22];14(3). *https://doi.org/10.15447/sfews.2016v14iss3art4*

Bukaveckas PA, Frankli R, Tassone S, Trache B, Egerton T. 2018. Cyanobacteria and cyanotoxins at the river–estuarine transition. Harmful Algae. [accessed 2022 Jul 22];76:11–21. *https://doi.org/10.1016/j.hal.2018.04.012*

Burford MA, Carey CC, Hamilton DP, Huisman J, Paerl HW, Wood SA, Wulff A. 2020. Perspective: advancing the research agenda for improving understanding of cyanobacteria in a future of global change. Harmful Algae. [accessed 2022 Jul 22];91:101601. *https://doi.org/10.1016/j.hal.2019.04.004*

Burkholde M, Shumway SE, Glibert, M. 2018. Food web and ecosystem impacts of harmful algae. Chapter 7 In: Shumway SE, Burkholder JM, Morton SL, editors. Harmful algal blooms: a compendium desk reference. [accessed 2022 Jul 22]:243–336. *https://doi.org/10.1002/9781118994672.ch7*

Calandrino ES, Paerl HW. 2011. Determining the potential for the proliferation of the harmful cyanobacterium *Cylindrospermopsis raciborskii* in Currituck Sound, North Carolina. Harmful Algae. [accessed 2022 Jul 22];11:1–9. *https://doi.org/10.1016/j.hal.2011.04.003*

Capper A, Flewelling LJ, Arthur K. 2013. Dietary exposure to harmful algal bloom (HAB) toxins in the endangered manatee (*Trichechus manatus latirostris*) and green sea turtle (*Chelonia mydas*) in Florida, USA. Harmful Algae. [accessed 2022 Jul 22];28:1–9. *https://doi.org/10.1016/j.hal.2013.04.009*

Carmichael WW. 1981. Freshwater blue-green algae (cyanobacteria) toxins—a review. In: Carmichael WW, editor. The water environment: algal toxins and health. [accessed 2022 Jul 22];1–13. Boston (MA): Springer. *https://doi.org/10.1007/978-1-4613-3267-1_1*

Carmichael WW, Boyer GL. 2016. Health impacts from cyanobacteria harmful algae blooms: implications for the North American Great Lakes. Harmful Algae. [accessed 2022 Jul 22];54:194–212. *https://doi.org/10.1016/j.hal.2016.02.002*

Cayan D, Tyree M, Dettinger M, Hidalgo H, Das T, Maurer E, Bromirski PD, Flick RE. 2009. Climate change scenarios and sea level rise estimates for the California 2009 climate change scenarios assessment. [accessed 2022 Jul 22]. Available from: *https://escholarship.org/content/qt8vh4m2qz/qt8vh4m2qz.pdf*

Chen Q, Wang M, Zhang J, Shi W, Mynett AE, Yan H, Hu L. 2019. Physiological effects of nitrate, ammonium, and urea on the growth and microcystins contamination of *Microcystis aeruginosa*: implication for nitrogen mitigation. Water Res. [accessed 2022 Jul 22];163:114890. *https://doi.org/10.1016/j.watres.2019.114890*

Chia MA, Kramer BJ, Jankowiak JG, Bittencourt-Oliveira MDC, Gobler CJ. 2019. The individual and combined effects of the cyanotoxins, anatoxin-a and microcystin-LR, on the growth, toxin production, and nitrogen fixation of prokaryotic and eukaryotic algae. Toxins. [accessed 2022 Jul 22];11:43. *https://doi.org/10.3390/toxins11010043*

Cloern JE. 1982. Does the benthos control phytoplankton biomass in South San Francisco Bay (USA)? Mar Ecol Prog Ser. [accessed 2022 Jul 22];9:191e202.

Cloern JE. 1996. Phytoplankton bloom dynamics in coastal ecosystems: a review with some general lessons from sustained investigation of San Francisco Bay, California. Rev Geophys. [accessed 2022 Jul 22];34:127–168. *https://doi.org/10.1029/96rg00986*

Cloern JE. 1999. The relative importance of light and nutrient limitation of phytoplankton growth: a simple index of coastal ecosystem sensitivity to nutrient enrichment. Aquatic Ecol. [accessed 2022 Jul 22];33:3–15. *https://doi.org/10.1023/a:1009952125558*

Cloern JE. 2001. Our evolving conceptual model of the coastal eutrophication problem. Mar Ecol Prog Ser. [accessed 2022 Jul 22];210:223–253. *https://doi.org/10.3354/meps210223*

Cloern JE, Hieb KA, Jacobson T, Sansó B, Di Lorenzo E, Stacey MT, Largier JL, Meiring W, Peterson WT, Powell TM, Winder M. Biological communities in San Francisco Bay track large-scale climate forcing over the North Pacific. Geophys Res Lett. [accessed 2022 Jul 22];37(21). *https://doi.org/10.1029/2010GL044774*

Cloern JE. 2018. Why large cells dominate estuarine phytoplankton. Limnol Oceanogr. [accessed 2022 Jul 22];63:S392–S409. *https://doi.org/10.1002/lno.10749*

Cloern JE, Dufford R. 2005. Phytoplankton community ecology: principles applied in San Francisco Bay. Mar Ecol Prog Ser. [accessed 2022 Jul 22];285:11–28. *https://doi.org/10.3354/meps285011*

Cloern JE, Jassby AD. 2012. Drivers of change in estuarine–coastal ecosystems: discoveries from four decades of study in San Francisco Bay. Rev Geophys. [accessed 2022 Jul 22];50:RG4001. *https://doi.org/10.1029/2012RG000397*

Cloern JE, Jassby AD, Carstensen J, Bennett WA, Kimmerer W, Mac Nally R, Schoellhamer DH, Winder M. 2012. Perils of correlating CUSUM-transformed variables to infer ecological relationships (Breton et al. 2006; Glibert 2010). Limnol Oceanogr. [accessed 2022 Jul 22];57(2):665–668. *https://doi.org/10.4319/lo.2012.57.2.0665*

Cloern JE, Jassby AD, Thompson JK, Hieb KA. 2007. A cold phase of the East Pacific triggers new phytoplankton blooms in San Francisco Bay. PNAS. [accessed 2022 Jul 22];104:8561–18565. *https://doi.org/10.1073/pnas.0706151104*

Cloern JE, Schraga T. 2016. USGS measurements of water quality in San Francisco Bay (CA), 1969–2015. US Geological Survey data release. [accessed 2022 Jul 22]. *https://doi.org/10.5066/F7TQ5ZPR*

Cloern JE, Schraga TS, Lopez CB, Knowles N, Labiosa RG, Dugdale R. 2005. Climate anomalies generate an exceptional dinoflagellate bloom in San Francisco Bay. Geophys Res Lett. [accessed 2022 Jul 22];32. *https://doi.org/10.1029/2005gl023321*

Cloern JE, Schraga TS, Nejad E, Martin C. 2020. Nutrient status of San Francisco Bay and its management implications. Estuaries Coasts. [accessed 2022 Jul 22];43:1299–1317. *https://doi.org/10.1007/s12237-020-00737-w*

Cloern JE, Jassby AD, Schraga TS, Nejad E, Martin C. 2017. Ecosystem variability along the estuarine salinity gradient: Examples from long-term study of San Francisco Bay. Limnol Oceanogr. [accessed 2022 Jul 22];62(S1):S272-91. *https://doi.org/10.1002/lno.10537*

Cooke J, Joab C, Lu Z. 2018. Delta nutrient research plan. [accessed 2022 Jul 22]. Rancho Cordova (CA): Regional Water Quality Control Board, Central Valley Region. Available from: *https://www.waterboards.ca.gov/centralvalley/water_issues/delta_water_quality/delta_nutrient_research_plan/2018_0802_dnrp_final.pdf*

Corbel S, Mougin C, Bouaïcha N. 2014. Cyanobacterial toxins: modes of actions, fate in aquatic and soil ecosystems, phytotoxicity and bioaccumulation in agricultural crops. Chemosphere. [accessed 2022 Jul 22];96:1–15. *https://doi.org/10.1016/j.chemosphere.2013.07.056*

Cornwell JC, Glibert PM, Owen MS. 2014. Nutrient fluxes from sediments in the San Francisco Bay Delta. Estuaries Coasts. [accessed 2022 Jul 22];37:1120–1122. *https://doi.org/10.1007/s12237-013-9755-4*

Crespo BG, Keafer BA, Ralston DK, Lind H, Farber D, Anderson DM. 2011. Dynamics of Alexandrium fundyense blooms and shellfish toxicity in the Nauset Marsh System of Cape Cod (Massachusetts, USA). Harmful Algae. [accessed 2022 Jul 22];12:26–38. *https://doi.org/10.1016/j.hal.2011.08.009*

Damashek J, Casciotti KL, Francis CA. 2016. Variable nitrification rates across environmental gradients in turbid, nutrient-rich estuary waters of San Francisco Bay. Estuaries Coasts. [accessed 2022 Jul 22];39:1050–1071. *https://doi.org/10.1007/s12237-016-0071-7*

D'Angelo E. 2019. Development and evaluation of a sensitive, Diffusive Gradients in Thin-Films (DGT) method for determining microcystin-LR concentrations in freshwater and seawater. Harmful Algae. [accessed 2022 Jul 22];89:101668. *https://doi.org/10.1016/j.hal.2019.101668*

Danil K, Berman M, Fram E, Preti A, Fire SE, Leighfield T, Carrett J, Carter ML, Lefebvre K. 2021. Marine algal toxins and their vectors in southern California cetaceans. Harmful Algae. [accessed 2022 Jul 22];103:102000. *https://doi.org/10.1016/j.hal.2021.102000*

Davis JA, Taberski K, Buchan K, Tucker D, Flegal AR, Gunther AJ. 2006. The regional monitoring program: science in support of managing water quality in the San Francisco Estuary. [accessed 2022 Jul 22]. Oakland (CA): San Francisco Estuary Institute. SFEI Contribution #435. Available from: *https://www.sfei.org/sites/default/files/biblio_files/Management_Synthesis_Final_83006.pdf*

Davis T, Harke M, Marcoval M, Goleski J, Orano-Dawson C, Berry D, Gobler, CJ. 2010. Effects of nitrogenous compounds and phosphorus on the growth of toxic and non-toxic strains of *Microcystis* during cyanobacterial blooms. Aquat Microb Ecol. [accessed 2022 Jul 22];61:149^{-1}62. *https://doi.org/10.3354/ame01445*

Dettinger M, Anderson J, Anderson M, Brown LR, Cayan D, Maurer E. 2016. Climate change and the Delta. San Franc Estuary Watershed Sci. [accessed 2022 Jul 22];14(3). *https://doi.org/10.15447/sfews.2016v14iss3art5*

Diaz RJ, Rosenberg R. 2008. Spreading dead zones and consequences for marine ecosystems. Science. [accessed 2022 Jul 22];321(5891):926–929. *https://doi.org/10.1126/science.1156401*

Dugdale R, Wilkerson F, Parker AE, Marchi A, Taberski K. 2012. River flow and ammonium discharge determine spring phytoplankton blooms in an urbanized estuary. Estuar Coast Shelf Sci. [accessed 2022 Jul 22];115(10):187–199. *https://doi.org/10.1016/j.ecss.2012.08.025*

Dugdale RC, Wilkerson FP, Hogue VE, Marchi A. 2007. The role of ammonium and nitrate in spring bloom development in San Francisco Bay. Estuar Coast Shelf Sci. [accessed 2022 Jul 22];73(1–2):17–29. *https://doi.org/10.1016/j.ecss.2006.12.008*

Dugdale RC, Wilkerson FP, Parker AE. 2013. A biogeochemical model of phytoplankton productivity in an urban estuary: the importance of ammonium and freshwater flow. Ecol Model. [accessed 2022 Jul 22];263:291. *https://doi.org/10.1016/j.ecolmodel.2013.05.015*

Dyble J, Paerl HW, Neilan BA. 2002. Genetic characterization of *Cylindrospermopsis raciborskii* (Cyanobacteria) isolates from diverse geographic origins based on nifH and cpcBA-IGS nucleotide sequence analysis. Appl Env Microbiol. [accessed 2022 Jul 22];68, 2567–2571. *https://doi.org/10.1128/aem.68.5.2567-2571.2002*

Eberts M, Wagne CR, Woodside MD. 2019, Water priorities for the nation—the US Geological Survey Next Generation Water Observing System. US Geological Survey Fact Sheet. [accessed 2022 Jul 22];2019–3046. 2 p. *https://doi.org/10.3133/fs20193046*

Esparza ML, Farrell AE, Craig DJ, Swanson C, Dhaliwal BS, Berg GM. 2014. Impact of atypical ammonium concentrations on phytoplankton abundance and composition in fresh versus estuarine waters. Aquat Biol. [accessed 2022 Jul 22];21(3):191–204. *https://doi.org/10.3354/ab00588*

Federal Register. 2009. Implementation of New Competitive Prevention, Control, and Mitigation of Harmful Algal Blooms (HAB) Program and Regional Rotation of the Existing and New National Competitive HAB Programs. May 04, 2009. A Notice by the National Oceanic and Atmospheric Administration. Fed Regist [accessed 2022 Jul 22];74:20465–20469. Available from: *https://www.federalregister.gov/articles/2009/05/04/E9-10187/implementation-of-new-competitive-prevention-control-and-mitigation-of-harmful-algal-blooms-hab*

Feger N. 2017. San Francisco Bay nutrients project. Oakland (CA): San Francisco Bay Regional Water Quality Control Board. [accessed 2022 Jul 22]. Available from: *https://www.waterboards.ca.gov/sanfranciscobay/water_issues/programs/planningtmdls/amendments/estuarynne.html*

Feger N, Senn D, Sutula M. 2012. Nutrient management strategy for San Francisco Bay. [accessed 2022 Jul 22]. Available from: *https://www.waterboards.ca.gov/sanfranciscobay/water_issues/programs/planningtmdls/amendments/estuarineNNE/Nutrient_Strategy%20November%202012.pdf*

Fetscher AE, Howard MDA, Stancheva R, Kudela RM, Stein ED, Sutula MA, Busse LB, Sheath RG. 2015. Wadeable streams as widespread sources of benthic cyanotoxins in California, USA. Harmful Algae. [accessed 2022 Jul 22];49:105–116. *https://doi.org/10.1016/j.hal.2015.09.002*

Fire SE, Wang Z, Byrd M, Whitehead HR, Paternoster J, Morton SL. 2011. Co-occurrence of multiple classes of harmful algal toxins in bottlenose dolphins (*Tursiops truncatus*) stranding during an unusual mortality event in Texas, USA. Harmful Algae. [accessed 2022 Jul 22];10:330–336. *https://doi.org/10.1016/j.hal.2010.12.001*

Fleming LE, Kirkpatrick B, Backe LC, Walsh,CJ, Nierenberg K, Clark J, Reich A, Hollenbeck J, Benson J, Cheng YS, et al. 2011. Review of Florida red tide and human health effects. Harmful Algae. [accessed 2022 Jul 22];10(2):224–233. *https://doi.org/10.1016/j.hal.2010.08.006*

Frolov S, Kudela RM, Bellingham JG. 2013. Monitoring of harmful algal blooms in the era of diminishing resources: a case study of the US West Coast. Harmful Algae. [accessed 2022 Jul 22];21:1–12. *https://doi.org/10.1016/j.hal.2012.11.001*

Galloway JN, Dentener FJ, Capone DG, Boyer EW, Howarth RW, Seitzinger SP, Asner GP, Cleveland CC, Green PA, Holland EA, et al. 2004. Nitrogen cycles: past, present, and future. Biogeochemistry. [accessed 2022 Jul 22];70:153–226. *https://doi.org/10.1007/s10533-004-0370-0*

Garcia AC, Bargu S, Dash P, Rabalais NN, Sutor M, Morrison W, Walker ND. 2010. Evaluating the potential risk of microcystins to blue crab (*Callinectes sapidus*) fisheries and human health in a eutrophic estuary. Harmful Algae. [accessed 2022 Jul 22];9(2):134–143. *https://doi.org/10.1016/j.hal.2009.08.011*

Ger KA, Arneson P, Goldman CR, Teh SJ. 2010. Species specific differences in the ingestion of Microcystis cells by the calanoid copepods *Eurytemora affinis* and *Pseudodiaptomus forbesi*. J Plankt Res. [accessed 2022 Jul 22];32(10):1479 Freshwater Biol 1484. *https://doi.org/10.1093/plankt/fbq071*

Ger KA, Teh SJ, Baxa DV, Lesmeister S, Goldman CR. 2010. The effects of dietary *Microcystis aeruginosa* and microcystin on the copepods of the upper San Francisco Estuary. Freshw Biol. [accessed 2022 Jul 22];55:1548–1559. *https://doi.org/10.1111/j.1365-2427.2009.02367.x*

Ger KA, Teh SJ, Goldman CR. 2009. Microcystin-LR toxicity on dominant copepods *Eurytemora affinis* and *Pseudodiaptomus forbesi* of the upper San Francisco Estuary. Sci Tot Env. [accessed 2022 Jul 22];407:4852–4857. *https://doi.org/10.1016/j.scitotenv.2009.05.043*

Ger KA, Urrutia–Cordero, P, Frost PC, Hansson LA, Sarnelle O, Wilson AE, Lürling M. 2016. The interaction between cyanobacteria and zooplankton in a more eutrophic world. Harmful Algae. [accessed 2022 Jul 22];54:128–144. Freshwater Biol. *https://doi.org/10.1016/j.hal.2015.12.005*

Ghalambor CK, Grosholtz ED, Jeffries KM, Largier JK, McCormick SD, Sommer T, Velotta J, Whitehead A. 2021. Ecological effects of climate-driven salinity variation in the San Francisco Estuary: can we anticipate and manage the coming changes? San Franc Estuary Watershed Sci. [accessed 2022 Jul 22];19(2). *https://doi.org/10.15447/sfews.2021v19iss2art3*

Gibble CM, Hayashi K, Kudela RM. 2017. The use of blood collection cards for assessing presence of microcystin in marine and estuarine birds. J Wildlife Rehab. [accessed 2022 Jul 22];37(1) *https://theiwrc.org/journal-article-search/*

Gibble CM, Kudela RM. 2014. Detection of persistent microcystin toxins at the land–sea interface in Monterey Bay, California. Harmful Algae [accessed 2022 Jul 22];39:146-53. *https://doi.org/10.1016/j.hal.2014.07.004*

Gibble CM, Kudela RM, Knowles S, Bodenstein B, Lefebvre KA. 2021. Domoic acid and saxitoxin in seabirds in the United States between 2007 and 2018. Harmful Algae. [accessed 2022 Jul 22];103:101981. *https://doi.org/10.1016/j.hal.2021.101981*

Gibble CM, Peacock MB, Kudela RM. 2016. Evidence of freshwater algal toxins in marine shellfish: implications for human and aquatic health. Harmful Algae. [accessed 2022 Jul 22];59:59-66. *https://doi.org/10.1016/j.hal.2016.09.007*

Glibert PM. 2010. Long-term changes in nutrient loading and stoichiometry and their relationships with changes in the food web and dominant pelagic fish species in the San Francisco Estuary, California. Rev Fish Sci. [accessed 2022 Jul 22];18:211–232. *https://doi.org/10.1080/10641262.2010.492059*

Glibert PM. 2020. Harmful algae at the complex nexus of eutrophication and climate change. Harmful Algae. [accessed 2022 Jul 22];91:101583. *https://doi.org/10.1016/j.hal.2019.03.001*

Glibert PM, Berdalet E, Burford MA, Pitcher GC, Zhou M, editors. 2018. Global ecology and oceanography of harmful algal blooms. [accessed 2022 Jul 22]. New York (NY): Springer. *https://doi.org/10.1007/978-3-319-70069-4*

Glibert PM, Beusen AH, Harrison JA, Dürr HH, Bouwman AF, Laruelle GG. 2018. Changing land-, sea-, and airscapes: sources of nutrient pollution affecting habitat suitability for harmful algae. In: Glibert PM, Berdalet E, Burford MA, Pitcher GC, Zhou M, editors. Global ecology and oceanography of harmful algal blooms. New York (NY): Springer. [accessed 2022 Jul 22];232:53–76.

Glibert PM, Dugdale RC, Wilkerson F, Parker AE, Alexander J, Antell E, Blaser S, Johnson A, Lee J, Lee T, et al. 2014. Major—but rare—spring blooms in 2014 in San Francisco Bay Delta, California, a result of the long-term drought, increased residence time, and altered nutrient loads and forms. J Exp Mar Biol Ecol. [accessed 2022 Jul 22];460:8–18. *http://doi.org/10.1016/j.jembe.2014.06.001*

Gobler CJ. 2020. Climate change and harmful algal blooms: insights and perspective. Harmful Algae. [accessed 2022 Jul 22];9:101731. *https://doi.org/10.1016/j.hal.2019.101731*

Gobler CJ, Burkholder JM, Davis TW, Harke MJ, Johengen T, Stow CA, Van De Waal DB. 2016. The dual role of nitrogen supply in controlling the growth and toxicity of cyanobacterial blooms. Harmful Algae. [accessed 2022 Jul 22];54:87–97. *https://doi.org/10.1016/j.hal.2016.01.010*

Greer B, Meneely JP, Elliott CT. 2018. Uptake and accumulation of Microcystin-LR based on exposure through drinking water: an animal model assessing the human health risk. Sci Rep. [accessed 2022 Jul 22];8:4913. *https://doi.org/10.1038/s41598-018-23312-7*

Grenier JL, Davis JA. 2010. Water Quality in South San Francisco Bay, California: Current condition and potential issues for the South Bay Salt Pond Restoration Project. In: Whitacre, D (editor). Reviews of environmental contamination and toxicology. Vol. 206. New York (NY): Springer. Rev Env Cont Tech. [accessed 2022 Jul 22];206:115–147. *https://doi.org/10.1007/978-1-4419-6260-7_6*

Griffith AW, Gobler CJ. 2020. Harmful algal blooms: a climate change co-stressor in marine and freshwater ecosystems. Harmful Algae. [accessed 2022 Jul 22];91:101590. *https://doi.org/10.1016/j.hal.2019.03.008*

Hall, NS, Paerl HW, Peierls BL, Whippl AC, Rossignol KL. 2013. Effects of climatic variability on phytoplankton community structure and bloom development in the eutrophic, microtidal, New River Estuary, North Carolina, USA. Estuar Coast Shelf Sci. [accessed 2022 Jul 22];117:70–82. *https://doi.org/10.1016/j.ecss.2012.10.004*

Hallegraeff GM. 2014. Harmful algae and their toxins: progress, paradoxes and paradigm shifts. In: Rossini GP, editor. Toxins and biologically active compounds from microalgae: Volume 1. Origin, chemistry and detection. [accessed 2022 Jul 22]. Boca Raton (FL): CRC Press. 530 p.

Hallegraeff GM, Anderson DM, Belin C, Bottein MYD, Bresnan E, Chinain M, Enevoldsen H, Iwataki M, Karlson B, McKenzie CH, et al. 2021. Perceived global increase in algal blooms is attributable to intensified monitoring and emerging bloom impacts. Comm Earth Env. [accessed 2022 Jul 22];2(117). *https://doi.org/10.1038/s43247-021-00178-8*

Harding LW, Adolf JE, Mallonee ME, Miller WD, Gallegos CL, Perry ES, Johnson JM, Sellner KG, Paerl HW. 2015. Climate effects on phytoplankton floral composition in Chesapeake Bay. Estuar Coast Shelf Sci. [accessed 2022 Jul 22];162:53–68. *https://doi.org/10.1016/j.ecss.2014.12.030*

Harding Jr LWH, Mallonee ME, Perry ES, Miller WD, Adolf JE, Gallegos CL, Paerl HW. 2016. Variable climatic conditions dominate recent phytoplankton dynamics in Chesapeake Bay. Sci Reps. [accessed 2022 Jul 22];6:23773. *https://doi.org/10.1038/srep23773*

Harke MJ, Steffen MM, Gobler CJ, Otten TG, Wilhelm SW Wood SA, Paerl HW. 2016. A review of the global ecology, genomics, and biogeography of the toxic cyanobacterium, *Microcystis* spp. Harmful Algae. [accessed 2022 Jul 22];54:4–20. *https://doi.org/10.1016/j.hal.2015.12.007*

Hartman R, Ateljavich E, Berg M, Bouma–Gregson K, Bosworth D, Rasmussen N, Flynn T, Pennington T. 2021. Report on the impact of the emergency drought barrier on harmful algal blooms and aquatic weeds in the Delta. Sacramento (CA): Department of Water Resources. [accessed 2022 Jul 22]. 86 p + appendix. Available from: *https://www.waterboards.ca.gov/waterrights/water_issues/programs/drought/tucp/docs/2021/20211215_dwremail_cond8.pdf*

Healey M, Dettinger M, Norgaard R. 2008. Introduction: new perspectives on science and policy in the Bay–Delta. The state of Bay–Delta science 2008. Sacramento (CA): The CALFED Science Program. [accessed 2022 Jul 22]. p. 1. Available from: *http://www.science.calwater.ca.gov/publications/sbds.html*

Heisler J, Glibert PM, Burkholder JM, Anderson DM, Cochlan W, Dennison WC, Dortch Q, Gobler CJ, Heil CA, Humphries E, et al. 2008. Eutrophication and harmful algal blooms: a scientific consensus. Harmful Algae. [accessed 2022 Jul 22];8(1):3–13. *https://doi.org/10.1016/j.hal.2008.08.006*

Hestir EL, Schoellhamer DH, Greenberg J, Morgan–King T, Ustin SL. 2016. The effect of submerged aquatic vegetation expansion on a declining turbidity trend in the Sacramento–San Joaquin River Delta. Estuaries Coasts. [accessed 2022 Jul 22];39(4):1100–1112. *https://doi.org/10.1007/s12237-015-0055-z*

Holleman R, MacVean L, Mckibben M, Sylvester Z, Wren I, Senn D. 2017. Nutrient Management Strategy Science Program. Richmond (CA): San Francisco Estuary Institute. estuaryI Contribution No. 879. [accessed 2022 Jul 22]. Available from: *https://www.sfei.org/documents/nutrient-management-strategy-science-program*

Howard MD, Nagoda C, Kudela RM, Hayashi K, Tatters A, Caron DA, Busse L, Brown J, Sutula M, Stein ED. 2017. Microcystin prevalence throughout lentic waterbodies in coastal southern California. Toxins [accessed 2022 Jul 22];9(7):231. *https://doi.org/10.3390/toxins9070231*

Howard MD, Smith J, Caron DA, Kudela RM, Loftin K, Hayashi K, Fadness R, Fricke S, Kann J, Roethler M, et al. 2022. Integrative monitoring strategy for marine and freshwater harmful algal blooms and toxins across the freshwater-to-marine continuum. Int Environ Assess Mgmt. [accessed 2022 Jul 22]. *https://doi.org/10.1002/ieam.4651*

Hozumi A, Ostrovsky I, Sukenik A, Gildor H. 2019. Turbulence regulation of Microcystis surface scum formation and dispersion during a cyanobacteria bloom event. Inland Waters. [accessed 2022 Jul 22];10(1):51–70. *https://doi.org/10.1080/20442041.2019.1637681*

Huisman J, Codd GA, Paerl HW, Ibelings BW, Verspagen JMH, Visse PM. 2018. Cyanobacterial blooms. Nat Rev Microbiol. [accessed 2022 Jul 22];16:471–483. *https://doi.org/10.1038/s41579-018-0040-1*

Huisman J, Sharples JJ, Stroom JM, Visser PM, Kardinaal EA, Verspagen JMH, Sommeijer B. 2004. Changes in turbulent mixing shift competition for light between phytoplankton species. Ecology. [accessed 2022 Jul 22];85(11):2960–2970. *https://doi.org/10.1890/03-0763*

Jardine SL, Fisher MC, Moore SK, Samhouri JF. 2020. Inequality in the economic impacts from climate shocks in fisheries: the case of harmful algal blooms. Ecol Econ. [accessed 2022 Jul 22];176:106691. *https://doi.org/10.1016/j.ecolecon.2020.106691.*

Jassby A. 2008. Phytoplankton in the upper San Francisco Estuary: recent biomass trends, their causes, and their trophic significance. San Franc Estuary Watershed Sci. [accessed 2022 Jul 22];6(1). *https://doi.org/10.15447/sfews.2008v6iss1art2*

Jessup DA, Miller MA, Ryan JP, Nevins HM, Kerkering HA, Mekebri A, Crane DB, Johnson TA, Kudela RM. 2009. Mass stranding of marine birds caused by a surfactant-producing red tide. PLoS ONE. [accessed 2022 Jul 22];4(2):e4550. *https://doi.org/10.1371/journal.pone.0004550*

Ji R, Franks PJ. 2007. Vertical migration of dinoflagellates: model analysis of strategies, growth, and vertical distribution patterns. Mar Ecol Prog Ser. [accessed 2022 Jul 22];344:49–61. *https://doi.org/10.3354/meps06952*

Jiang W, Zhou W, Uchida H, Kikumori M, Irie K, Watanabe R, Suzuki T, Sakamoto B, Kamio M, Nagai H. 2014. A new lyngbyatoxin from the Hawaiian cyanobacterium *Moorea producens*. Mar Drugs. [accessed 2022 Jul 22];12(5):2748–2759. *https://doi.org/10.3390/md12052748*

Jöhnk KD, Huisman JE, Sharples J, Sommeijer BE, Visser PM, Stroom JM, 2008. Summer heatwaves promote blooms of harmful cyanobacteria. Glob Change Biol. [accessed 2022 Jul 22];14(3):495–512. *https://doi.org/10.1111/j.1365-2486.2007.01510.x*

Kennish MJ. 2021. Drivers of change in estuarine and coastal marine environments: an overview. Open J Ecol. [accessed 2022 Jul 22];11:224–239. *https://doi.org/10.4236/oje.2021.113017*

Kidwell D. 2015. Mitigation of harmful algal blooms: the way forward. Environ Sci. PICES Press. [accessed 2022 Jul 22];23(2):22.

Kimmerer W, Wilkerson F, Downing B, Dugdale R, Gross ES, Kayfetz K, Khanna S, Parker AE, Thompson J. 2019. Effects of drought and the emergency drought barrier on the ecosystem of the California Delta. San Franc Estuary Watershed Sci. [accessed 2022 Jul 22];17(3). *https://doi.org/10.15447/sfews.2019v17iss3art2*

Kirkpatrick B, Fleming LE, Squicciarini D, Backer LC, Clark R, Abraham W, Benson J, Cheng YS, Johnson D, Pierce R, et al. 2004. Literature review of Florida red tide: implications for human health effects. Harmful Algae. [accessed 2022 Jul 22];3(2):99–115. *https://doi.org/10.1016%2Fj.hal.2003.08.005*

Knowles N, Cronkite–Ratcliff C. 2018. Modeling managed flows in the Sacramento/San Joaquin Watershed, California, under scenarios of future change for CASCaDE2. Menlo Park (CA): US Department of the Interior, US Geological Survey. Open-File Report 2018-1101. [accessed 2022 Jul 22]. *https://doi.org/10.3133/ofr20181101*

Kudela RM. 2011. Characterization and deployment of Solid Phase Adsorption Toxin Tracking (SPATT) resin for monitoring of microcystins in fresh and saltwater. Harmful Algae. [accessed 2022 Jul 22];11:117–125. *https://doi.org/10.1016/j.hal.2011.08.006*

Kudela R, Berdalet E, Urban E. 2015. Harmful algal blooms: a scientific summary for policy makers. [accessed 2022 Jul 22]. Paris: IOC/UNESCO.

Kudela RM, Bickel A, Carter M, Howard MDA, Rosenfeld L. 2015b. The monitoring of harmful algal blooms through ocean observing: the development of the California Harmful Algal Bloom Monitoring and Alert Program. In: Liu Y, Kerkering H, Weisbert RH, editors. Coastal ocean observing systems. [accessed 2022 Jul 22]. Boston (MA): Academic Press. p. 58–75.

Kudela RM, Hayashi K, Caceres CG. 2020. Is San Francisco Bay resistant to *Pseudo-nitzschia* and domoic acid? Harmful Algae. [accessed 2022 Jul 22];92:101617. *https://doi.org/10.1016/j.hal.2019.05.010*

Kudela RM, Lane JQ, Cochlan WP. 2008. The potential role of anthropogenically derived nitrogen in the growth of harmful algae in California, USA. Harmful Algae. [accessed 2022 Jul 22];8(1):103–110. *https://doi.org/10.1016/j.hal.2008.08.019*

Kudela RM, Seeyave S, Cochlan WP. 2010. The role of nutrients in regulation and promotion of harmful algal blooms in upwelling systems. Prog Oceanogr. [accessed 2022 Jul 22];85(1–2):122–135. *https://doi.org/10.1016/j.pocean.2010.02.008*

Kurobe T, Baxa DV, Mioni CE, Kudela RM, Smythe TR, Waller S, Chapman AD, Teh SJ. 2013. Identification of harmful cyanobacteria in the Sacramento–San Joaquin Delta and Clear Lake, California by DNA barcoding. SpringerPlus. [accessed 2022 Jul 22];2:491. *https://doi.org/10.1186/2193-1801-2-491*

Kurobe T, Lehman PW, Haqu ME, Sedda T, Lesmeiste S, Teh S. 2018. Evaluation of water quality during successive severe drought years within *Microcystis* blooms using fish embryo toxicity tests for the San Francisco Estuary, California. Sci Tot Env. [accessed 2022 Jul 22];610–611:1029–1037. *https://doi.org/10.1016/j.scitotenv.2017.07.267*

Lancelo C, Grosjean, P Rousseau, V, Breton E, Glibert PM. 2012. Rejoinder to “Perils of correlating CUSUM-transformed variables to infer ecological relationships (Breton et al. 2006; Glibert 2010)”. Limnol Oceanogr. [accessed 2022 Jul 22];57(2):669–670. *https://doi.org/10.4319/lo.2012.57.2.0669*

Laufkötter C, Zscheischler J, Frölicher TL. 2020. High-impact marine heatwaves attributable to human-induced global warming. Science. [accessed 2022 Jul 22];369:1621–1625. *https://doi.org/10.1126/science.aba0690*

Lee J, Parker AE, Wilkerson FP, Dugdale RC. 2015. Uptake and inhibition kinetics of nitrogen in *Microcystis aeruginosa*: results from cultures and field assemblages collected in the San Francisco Bay Delta, CA. Harmful Algae. [accessed 2022 Jul 22];47:126–140. *https://doi.org/10.1016/j.hal.2015.06.002*

Lehman PW. 2000. The influence of climate on phytoplankton community biomass in San Francisco Bay Estuary. Limnol Oceanogr. [accessed 2022 Jul 22];45(3):580–590. *https://doi.org/10.4319/lo.2000.45.3.0580*

Lehman PW, Boyer G, Hall C, Waller S, Gehrts K. 2005. Distribution and toxicity of a new colonial *Microcystis aeruginosa* bloom in the San Francisco Bay Estuary, California. Hydrobiologia. [accessed 2022 Jul 22];541(1):87–99. *https://doi.org/10.1007/s10750-004-4670-0*

Lehman PW, Kendall C, Guerin MA, Young, MB, Silva SR, Boyer GL. Teh SJ. 2015. Characterization of the *Microcystis* bloom and its nitrogen supply in San Francisco Estuary using stable isotopes. Estuaries Coasts. [accessed 2022 Jul 22];38(1):165–178. *https://doi.org/10.1007/s12237-014-9811-8*

Lehman PW, Kurobe T, Huynh K, Lesmeister S, Teh SJ. 2021. Covariance of phytoplankton, bacteria, and zooplankton communities within *Microcystis* blooms in San Francisco Estuary. Front Microbiol. [accessed 2022 Jul 22];12:1184. *https://doi.org/10.3389/fmicb.2021.632264*

Lehman PW, Kurobe T, Lesmeister S, Baxa D, Tung A, Teh SJ. 2017. Impacts of the 2014 severe drought on the *Microcystis* bloom in San Francisco Estuary. Harmful Algae. [accessed 2022 Jul 22];1;63:94–108. *https://doi.org/10.1016/j.hal.2017.01.011*

Lehman PW, Kurobe T, Lesmeister S, Lam C, Tung A, Xiong M, Teh SJ. 2018. Strong differences characterize *Microcystis* blooms between successive severe drought years in the San Francisco Estuary, California, USA. Aquat Microb Ecol. [accessed 2022 Jul 22];81(3):293–299. *http://dx.doi.org/10.3354/ame01876*

Lehman PW, Kurobe T, Teh SJ. 2020. Impact of extreme wet and dry years on the persistence of *Microcystis* harmful algal blooms in San Francisco Estuary. Quat Int. [accessed 2022 Jul 22];621:16–25. *https://doi.org/10.1016/j.quaint.2019.12.003*

Lehman PW, Smith RW. 1991. Environmental factors associated with phytoplankton succession for the Sacramento–San Joaquin Delta and Suisun Bay estuary, California. Estuar Coast Shelf Sci. [accessed 2022 Jul 22];32(2):105–128. *https://doi.org/10.1016/0272-7714(91)90009-Z*

Lehman PW, Teh SJ, Boyer GL, Nobriga ML, Bass E, Hogle, C. 2010. Initial impacts of *Microcystis aeruginosa* blooms on the aquatic food web in the San Francisco Estuary. Hydrobiologia. [accessed 2022 Jul 22];637(1):229–248. *https://doi.org/10.1007/s10750-009-9999-y*

Lewitus AJ, Horner RA, Caron DA, Garcia–Mendoza E, Hickey BM, Hunter M, Huppert DD, Kudela RM, Langlois GW, Largier JL, et al. 2012. Harmful algal blooms along the North American west coast region: history, trends, causes, and impacts. Harmful Algae. [accessed 2022 Jul 22];19:133–159. *https://doi.org/10.1016/j.hal.2012.06.009*

Li M, Zhu W, Guo L, Hu J, Chen H, Xiao M. 2016. To increase size or decrease density? Different Microcystis species has different choice to form blooms. Sci Rep. [accessed 2022 Jul 22];6:37056. *https://doi.org/10.1038/srep37056*

Lopes VM, Costa PR, Rosa R. 2019. Effects of harmful algal bloom toxins on marine organisms, In: Duarte B, Isabel Caçador C, editors. Ecotoxicology of marine organisms. [accessed 2022 Jul 22]. Boca Raton (FL): CRC Press. p. 42–88. *https://doi.org/10.1201/b22000*

Lucas L, Cloern JE, Koseff JR. Monismith SG, Thompson JK. 1999. Processes governing phytoplankton blooms in estuaries: Part I: the local production-loss balance. Mar Ecol Prog Ser. [accessed 2022 Jul 22];186:1–15. *https://doi.org/10.3354/meps187001*

Luoma SN, Muscatine LD. 2019. Sixteen years of San Francisco Estuary and watershed science: a retrospective. San Franc Estuary Watershed Sci. [accessed 2022 Jul 22];17(4). *https://doi.org/10.15447/sfews.2019v17iss4art1*

Malone TC, Malej A, Harding LW Jr, Smodlaka N, Turner RE, editors. 1999. Coastal and estuarine studies. Ecosystems at the land-sea margin, drainage basin to coastal sea. [accessed 2022 Jul 22];55. Washington (DC): American Geophysical Union. *https://doi.org/10.1029/CE055*

McKee LJ, Gluchowski DC. 2011. Improved nutrient load estimates for wastewater, stormwater and atmospheric deposition to South San Francisco Bay (south of the Bay Bridge). A Watershed Program report prepared for the Bay Area Clean Water Agencies (BACWA). [accessed 2022 Jul 22]. Oakland (CA): San Francisco Estuary Institute. Available from: *https://www.sfei.org/sites/default/files/biblio_files/Report_Nutrient_load_to_South_Bay_2011-8-31_FINAL.pdf*

McKibben SM, Peterson W, Wood AM, Trainer VL, Hunter M, White AE. 2017. Climatic regulation of the neurotoxin domoic acid. PNAS. [accessed 2022 Jul 22];114(2):239–244. *https://doi.org/10.1073/pnas.1606798114*

Mehinto AC, Smith J. Wenger E, Stanton B, Linville R, Brooks BW, Sutula MA, Howard MDA. 2021. Synthesis of ecotoxicological studies on cyanotoxins in freshwater habitats —evaluating the basis for developing thresholds protective of aquatic life in the United States. Sci Tot Env. [accessed 2022 Jul 22];795:148864. *https://doi.org/10.1016/j.scitotenv.2021.148864*

Melero–Jiménez IJ, Martín–Clemente E, García–Sánchez MJ, Bañares–España E, Flores–Moya A. 2020. The limit of resistance to salinity in the freshwater cyanobacterium *Microcystis aeruginosa* is modulated by the rate of salinity increase. Ecol Evol. [accessed 2022 Jul 22];10:5045–5055. *https://doi.org/10.1002/ece3.6257*

Melero–Jiménez IJ, Martín–Clemente E, García–Sánchez MJ, Flores–Moya A, Bañares–España E. 2019. Adaptation of the toxic freshwater cyanobacterium *Microcystis aeruginosa* to salinity is achieved by the selection of spontaneous mutants. Phycol Res. [accessed 2022 Jul 22];67:192–201. *https://doi.org/10.1111/pre.12370*

Meneely JP, Elliott CT. 2013. Microcystins: measuring human exposure and the impact on human health. Biomarkers. [accessed 2022 Jul 22];18:639–649. *https://doi.org/10.3109/1354750x.2013.841756*

Metcalf JS, Codd, GA. 2020. Co-occurrence of cyanobacteria and cyanotoxins with other environmental health hazards: impacts and implications. Toxins. [accessed 2022 Jul 22];12:629. *https://doi.org/10.3390/toxins12100629*

Miller MA, Kudela RM, Mekebri A, Crane D, Oates C, Tinker MT, Staedler M, Miller WA, Toy–Choutka S, Dominik C, et al. 2010. Evidence for a novel marine harmful algal bloom: cyanotoxin (microcystin) transfer from land to sea otters. PLoS ONE. [accessed 2022 Jul 22];5(9):e12576. *https://doi.org/10.1371/journal.pone.0012576*

Mioni C, Kudela R, Baxa D, Sullivan M, Hayash K, Smythe UT, White C. 2011. Harmful cyanobacteria blooms and their toxins in Clear Lake and the Sacramento–San Joaquin Delta (California). Rancho Cordova (CA): Central Valley Regional Water Quality Control Board. Surface Water Ambient Monitoring Program (SWAMP) Report #10-058-150. [accessed 2022 Jul 22]. Available from: *https://www.lakecountyca.gov/Assets/Departments/WaterResources/Algae/2011+Cyanobacteria+Report.pdf*

Mitrovic SM, Hardwick L, Dorani F. 2011. Use of flow management to mitigate cyanobacterial blooms in the Lower Darling River, Australia. J Plankt Res. [accessed 2022 Jul 22];33:229–241. *https://doi.org/10.1093/plankt/fbq094*

Moisander PH, McClinton E III, Paerl HW. 2002. Salinity effects on growth, photosynthetic parameters, and nitrogenase activity in estuarine planktonic cyanobacteria. Microb Ecol. [accessed 2022 Jul 22];43:432–442. *https://doi.org/10.1007/s00248-001-1044-2.*

Moisander PH, Paerl HW. 2000. Growth, productivity, and nitrogen fixation potential of *Nodularia* spp. (Cyanophyceae) in water from a subtropical estuary in the United States. J Phycol. [accessed 2022 Jul 22];36:645–658. *https://doi.org/10.1046/j.1529-8817.2000.99227.x*

Moreira C, Azevedo J, Antune A, Vasconcelos V. 2013. Cylindrospermops: occurrence, methods of detection and toxicology. J Appl Microbiol. [accessed 2022 Jul 22];114(3):605–620. *https://doi.org/10.1111/jam.12048.*

Myer MH, Urquhart E, Schaeffer BA, Johnston JM. 2020. Spatio-temporal modeling for forecasting high-risk freshwater cyanobacterial harmful algal blooms in Florida. Front Env Sci. [accessed 2022 Jul 22];8:202. *https://doi.org/10.3389/fenvs.2020.581091*

Najjar RG, Pyke CR, Adams MB, Breitburg D, Hershner C, Kemp M, Howarth R, Mulholland MR, Paolisso M, Secor D, et al. 2010. Potential climate-change impacts on the Chesapeake Bay. Estuarine Coast Shelf Sci. [accessed 2022 Jul 22];86:1–20. *https://doi.org/10.1016/j.ecss.2009.09.026*

[NRC] National Research Council. 2011. A review of the use of science and adaptive management in California's draft Bay Delta Conservation Plan. Washington (DC): The National Academies Press. [accessed 2022 Jul 22]. *https://doi.org/10.17226/13148*

[NRC] National Research Council. 2012. Sustainable water and Eenvironmental management in the California Bay–Delta. Washington, DC: The National Academies Press. [accessed 2022 Jul 22]. *https://doi.org/10.17226/13394*

Nejad ES, Cloern J, Schraga T. 2017. Phytoplankton species composition, abundance and cell size in San Francisco Bay: microscopic analyses of USGS samples collected 1992–2014 [data set]. U.S. Geological Survey data release. [accessed 2022 Jul 22]. *https://doi.org/10.5066/F74F1P6P*

Nixon SW. 1995. Coastal marine eutrophication: a definition, social causes, and future concerns. Ophelia. [accessed 2022 Jul 22];41:199–219. *https://doi.org/10.1080/00785236.1995.10422044*

O'Brien KR, Meyer DL, Waite, AM, Ivey, GN, DP. Hamilton DP. 2004. Disaggregation of *Microcystis aeruginosa* colonies under turbulent mixing: laboratory experiments in a grid-stirred tank. Hydrobiologia. [accessed 2022 Jul 22];519:143–152. *https://doi.org/10.1023/B:HYDR.0000026501.02125.cf*

[OEHHA] Office of Environmental Health Hazard Assessment. 2012. Toxicological summary and suggested action levels to reduce potential adverse health effects of six cyanotoxins. [accessed 2022 Jul 22]. Sacramento (CA): California Environmental Protection Agency, OEHHA. Available from: *https://oehha.ca.gov/media/downloads/fish/document/cyanotoxins053112.pdf*

Oliver RL. 1994. Floating and sinking in gas-vacuolate cyanobacteria. J Phycol. [accessed 2022 Jul 22];30:161–173. *http://doi.org/10.1111/j.0022-3646.1994.00161.x*

Otten TG, Paerl HW, Dreher TW, Kimmerer WJ, Parker AE. 2017. The molecular ecology of *Microcystis* sp. blooms in the San Francisco Estuary Delta. Env Microbiol. [accessed 2022 Jul 22];19:3619–3637. *https://doi.org/10.1111/1462-2920.13860*

Owens S, Ignoffo TR, Frantzich J, Slaughter A, Kimmerer W. 2019. High growth rates of a dominant calanoid copepod in the northern San Francisco Estuary. J Plankt Res. [accessed 2022 Jul 22];41:939–954. *https://doi.org/10.1093/plankt/fbz064*

Paerl HW. 1988. Nuisance phytoplankton blooms in coastal, estuarine, and inland waters. Limnol Oceanogr. [accessed 2022 Jul 22];33:823–847. *https://doi.org/10.4319/lo.1988.33.4part2.0823*

Paerl HW. 1997. Coastal eutrophication and harmful algal blooms: importance of atmospheric deposition and groundwater as "new" nitrogen and other nutrient sources. Limnol Oceanogr. [accessed 2022 Jul 22];42(5part2):1154–1165. *https://doi.org/10.4319/lo.1997.42.5_part_2.1154*

Paerl HW. 2014. Mitigating harmful cyanobacterial blooms in a human- and climatically-impacted world. Life. [accessed 2022 Jul 22];4(4):988–1012. *https://doi.org/10.3390/life4040988*

Paerl HW, Barnard MA. 2020. Mitigating the global expansion of harmful cyanobacterial blooms: moving targets in a human- and climatically-altered world. Harmful Algae. [accessed 2022 Jul 22];96:101845. *https://doi.org/10.1016/j.hal.2020.101845*

Paerl HW, Bland PT, Bowles ND, Haibach ME. 1985. Adaptation to high-intensity, low-wavelength light among surface blooms of the cyanobacterium *Microcystis aeruginosa*. Appl Environ Microbiol. [accessed 2022 Jul 22];49:1046–1052. *https://doi.org/10.1128/aem.49.5.1046-1052.1985*

Paerl HW, Fulton RS. 2006. Ecology of harmful cyanobacteria. In: Granéli E, Turner JT, editors. Ecology of harmful algae. Berlin/Heidelberg (DE): Springer. [accessed 2022 Jul 22]. p. 95–109.

Paerl HW, Gardner WS, Havens KE, Joyner AR, McCarthy MJ, Newell E, Qin B, Scott JT. 2016. Mitigating cyanobacterial harmful algal blooms in aquatic ecosystems impacted by climate change and anthropogenic nutrients. Harmful Algae. [accessed 2022 Jul 22];54:213–222. *https://doi.org/10.1016/j.hal.2015.09.009*

Paerl HW, Hall NS, Calandrino ES. 2011. Controlling harmful cyanobacterial blooms in a world experiencing anthropogenic and climatic-induced change. Sci Tot Env. [accessed 2022 Jul 22];409:1739–1745. *https://doi.org/10.1016/j.scitotenv.2011.02.001*

Paerl HW, Havens KE, Xu H, Zhu G, McCarthy MJ, Newell SE, Scott JT, Hall NS, Otten TG, Qin B. 2020. Mitigating eutrophication and toxic cyanobacterial blooms in large lakes: the evolution of a dual nutrient (N and P) reduction paradigm. Hydrobiologia. [accessed 2022 Jul 22];847(21):4359–4375. *https://doi.org/10.1007/s10750-019-04087-y*

Paerl HW, Huisman J. 2008. Blooms like it hot. Science. [accessed 2022 Jul 22];320(5872):57–58. *https://doi.org/10.1126/science.1155398*

Paerl HW, Otten TG. 2013. Harmful cyanobacterial blooms: causes, consequences, and controls. Microbial Ecol. [accessed 2022 Jul 22];1;65(4):995–1010. *https://doi.org/10.1007/s00248-012-0159-y*

Paerl HW, Otten TG, Kudela R. 2018. Mitigating the expansion of harmful algal blooms across the freshwater-to-marine continuum. Env Sci Tech. [accessed 2022 Jul 22];52(10):5519–5529. *https://doi.org/10.1021/acs.est.7b05950*

Paerl HW, Paul VJ. 2012. Climate change: links to global expansion of harmful cyanobacteria. Water Res. [accessed 2022 Jul 22];46:1349–1363. *https://doi.org/10.1016/j.watres.2011.08.002*

Pal M, Yesankar J, Dwivedi A, Qureshi A. 2020. Biotic control of harmful algal blooms (HABs): a brief review. J Env Manag. [accessed 2022 Jul 22];268:110687. *https://doi.org/10.1016/j.jenvman.2020.110687*

Parker AE, Dugdale RC, Wilkerson FP. 2012. Elevated ammonium concentrations from wastewater discharge depress primary productivity in the Sacramento River and the Northern San Francisco Estuary. Mar Poll Bull. [accessed 2022 Jul 22];64:574–586. *https://doi.org/10.1016/j.marpolbul.2011.12.016*

Peacock MB, Gibble CM, Senn DB, Cloern JE, Kudela RM. 2018. Blurred lines: multiple freshwater and marine algal toxins at the land-sea interface of San Francisco Bay, California. Harmful Algae. [accessed 2022 Jul 22];73:138–147. *https://doi.org/10.1016/j.hal.2018.02.005*

Peacock MB, Kudela RM. 2014. Evidence for active vertical migration by two dinoflagellates experiencing iron, nitrogen, and phosphorus limitation. Limnol Oceanogr. [accessed 2022 Jul 22];59(3):660–673. *https://doi.org/10.4319/lo.2014.59.3.0660*

Peierls BL, Hall, NS, Paerl HW. 2012. Non-monotonic responses of phytoplankton biomass accumulation to hydrologic variability: a comparison of two coastal plain North Carolina estuaries. Estuaries Coasts. [accessed 2022 Jul 22];35:1376–1392. *https://doi.org/10.1007/s12237-012-9547-2*

Phlips EJ, Badylak S, Christman M, Wolny J, Brame J, Garland J, Hall L, Hart J, Landsberg J, Lasi M, et al. 2011. Scales of temporal and spatial variability in the distribution of harmful algae species in the Indian River Lagoon, Florida, USA. Harmful Algae. [accessed 2022 Jul 22];10(3):277–90. *https://doi.org/10.1016/j.hal.2010.11.001*

Plaas HE, Paerl HW. 2020. Toxic cyanobacteria: a growing threat to water and air quality. Env Sci Tech. [accessed 2022 Jul 22];55(1):44–64. *https://doi.org/10.1021/acs.est.0c06653*

Poste AE, Hecky RE, Guildford SJ. 2011. Evaluating microcystin exposure risk through fish consumption. Env Sci Tech. [accessed 2022 Jul 22];45:5806–5811. *https://doi.org/10.1021/es200285c*

Preece EP, Hardy FJ, Moore BC, Bryan M. 2017. A review of microcystin detections in estuarine and marine waters: environmental implications and human health risk. Harmful Algae. [accessed 2022 Jul 22];61:31–45. *https://doi.org/10.1016/j.hal.2016.11.006*

Preece EP, Moore BC, Hardy FJ. 2015. Transfer of microcystin from freshwater lakes to Puget Sound, WA and toxin accumulation in marine mussels (*Mytilus trossulus*). Ecotox Environ Safety. [accessed 2022 Jul 22];122:98–105. *https://doi.org/10.1016/j.ecoenv.2015.07.013*

Puddick J, van Ginkel R, Page CD, Murray JS, Greenhough HE, Bowater J, Selwood AI, Wood SA, Prinsep MR, Truman P, et al. 2021. Acute toxicity of dihydroanatoxin-a from *Microcoleus autumnalis* in comparison to anatoxin-a. Chemosphere. [accessed 2022 Jul 22];263:127937. *https://doi.org/10.1016/j.chemosphere.2020.127937*

Puddick J, Wood SA, Hawes I, Hamilton DP. 2016. Fine-scale cryogenic sampling of planktonic microbial communities: application to toxic cyanobacterial blooms. Limnol Oceanogr: Methods. [accessed 2022 Jul 22];14(9):600–609.

Puschner B, Bautista AC, Wong C. 2017. Debromoaplysiatoxin as the causative agent of dermatitis in a dog after exposure to freshwater in California. Front Vet Sci. [accessed 2022 Jul 22];4:50. *https://doi.org/10.3389/fvets.2017.00050*

Racchetti E, Longhii D, Ribaudo C, Soana E, Bartoli M. 2017. Nitrogen uptake and coupled nitrification-denitrification in riverine sediments with benthic microalgae and rooted macrophytes. Aquat Sci. [accessed 2022 Jul 22];79(3):487–505. *https://doi.org/10.1007/s00027-016-0512-1*

Raimonet M, Cloern JE. 2017. Estuary-ocean connectivity: fast physics, slow biology. Glob Change Biol. [accessed 2022 Jul 22];23:2345–2357. *https://doi:10.1111/gcb.13546*

Ralston DK, Moore SK. 2020. Modeling harmful algal blooms in a changing climate. Harmful Algae. [accessed 2022 Jul 22];91:101729. *https://doi.org/10.1016/j.hal.2019.101729*

Reynolds CS, Oliver RL, Walsby AE. 1987. Cyanobacterial dominance: the role of buoyancy regulation in dynamic lake environments. NZ J Mar Freshwater Res. [accessed 2022 Jul 22];21(3):379–390. *https://doi.org/10.1080/00288330.1987.9516234*

Rines JEB, McFarland MN, Donaghay PL, Sullivan JM. 2010. Thin layers and species-specific characterization of the phytoplankton community in Monterey Bay, California, USA. Cont Shelf Res. [accessed 2022 Jul 22];30(1):66–80. *https://doi.org/10.1016/j.csr.2009.11.001*

Rothenberger MB, Burkholder JM, Brownie C. 2009. Long-term effects of changing land use practices on surface water quality in a coastal river and lagoonal estuary. Environ Mgmt. [accessed 2022 Jul 22];44:505–523. *https://doi.org/10.1007/s00267-009-9330-8*

Saleh D, Domagalski J. 2021. Concentrations, loads, and associated trends of nutrients entering the Sacramento–San Joaquin Delta, California. San Franc Estuary Watershed Sci. [accessed 2022 Jul 22];19(4). *http://doi.org/10.15447/sfews.2021v19iss4art6*

Schaeffer BA, Myer MH. 2020. Resolvable estuaries for satellite derived water quality within the continental United States. Remote Sens Lett. [accessed 2022 Jul 22];11:535–544. *https://doi.org/10.1080/2150704x.2020.1717013*

Schoellhamer DH. 2011. Sudden clearing of estuarine waters upon crossing the threshold from transport to supply regulation of sediment transport as an erodible sediment pool is depleted: San Francisco Bay, 1999. Estuaries Coasts. [accessed 2022 Jul 22];34:885–899. *https://doi.org/10.1007/s12237-011-9382-x*

Shultz D, Campbell L, Kudela RM. 2019. Trends in Dinophysis abundance and diarrhetic shellfish toxin levels in California mussels (*Mytilus californianus*) from Monterey Bay, California. Harmful Algae. [accessed 2022 Jul 22];88:101641. *https://doi.org/10.1016/j.hal.2019.101641*

Shumway SE, Van Egmond HP, Hurst JW, Bean LL. 1995. Management of shellfish resources. Manual on harmful marine microalgae [accessed 2022 Jul 22];33:433.

Smayda TJ. 1997. What is a bloom? A commentary. Limnol Oceanogr. [accessed 2022 Jul 22];42(5part2):1132–1136. *https://doi.org/10.4319/lo.1997.42.5_part_2.1132*

Smith J, Sutula M, Bouma Gregson K, Van Dyke M. 2021a. California State Water Boards' framework and strategy for freshwater harmful algal bloom monitoring: executive synthesis. [accessed 2022 Jul 22]. Costa Mesa (CA): Southern California Coastal Water Research Project. SCCWRP Technical Report #1141. 20 p. Available from: *http://www.sccwrp.org*

Smith J, Sutula M, Bouma–Gregson K, Van Dyke M. 2021b. California State Water Boards' framework and strategy for freshwater harmful algal bloom monitoring: full report with appendices. [accessed 2022 Jul 22]. Costa Mesa (CA): Southern California Coastal Water Research Project. SCCWRP Technical Report #1141.B. 206 p. Available from: *http://www.sccwrp.org/*

Sommer T. 2020. How to respond? An introduction to current Bay–Delta natural resources management options. San Franc Estuary Watershed Sci. [accessed 2022 Jul 22];18(3). *https://doi.org/10.15447//sfews.2020v18iss3art1*

Sommer T, Armor C, Baxter R, Breuer R, Brown L, Chotkowski M, Culberson S, Feyrer F, Gingras M, Herbold B, et al. 2007. The collapse of pelagic fishes in the upper San Francisco Estuary. Fisheries. [accessed 2022 Jul 22];32:270–277. *https://doi.org/10.1577/1548-8446(2007)32[270:TCOPFI]2.0.CO;2*

Spier C, Stringfellow W, Hanlon J, Brunelli M, Estiandan M, Koski T, Kääriä J. 2013. Unprecedented bloom of toxin-producing cyanobacteria in the southern Bay–Delta estuary and its potential negative impact on the aquatic food-web. Report 4.5.1. Ecol Eng. [accessed 2022 Jul 22]. *http://dx.doi.org/10.13140/RG.2.1.3730.3768*

Steffen MM, Belisle BS, Watson SB, Boyer GL, Wilhelm SW. 2014. Status, causes and controls of cyanobacterial blooms in Lake Erie. J Great Lakes Res. [accessed 2022 Jul 22];40(2):215-225. *https://doi.org/10.1016/j.jglr.2013.12.012*

Stewart I, Seawright AA, Shaw GR. 2008. Cyanobacterial poisoning in livestock, wild mammals and birds—an overview. In: Hudnell HK, editor. Cyanobacterial harmful algal blooms: state of the science and research needs. Advances in experimental medicine and biology, volume 619. New York (NY): Springer. [accessed 2022 Jul 22]. *https://doi.org/10.1007/978-0-387-75865-7_28*

Stewart I, Webb PM, Schluter PJ, Fleming LE, Burns JW, Gantar M, Backer LC, Shaw GR. 2006. Epidemiology of recreational exposure to freshwater cyanobacteria—an international prospective cohort study. BMC Public Health. [accessed 2022 Jul 22];6(1):93. *https://doi.org/10.1186/1471-2458-6-93*

Stumpf RP, Johnson LT, Wynne TT, Baker DB. 2016. Forecasting annual cyanobacterial bloom biomass to inform management decisions in Lake Erie. J Great Lakes Res. [accessed 2022 Jul 22];42(6):1174–1183. *https://doi.org/10.1016/j.jglr.2016.08.006*

Sunda WG, Graneli E, Gobler CJ. 2006. Positive feedback and the development and persistent of ecosystem disruptive algal blooms. J Phycol. [accessed 2022 Jul 22];42:963–974. *https://doi.org/10.1111/j.1529-8817.2006.00261.x*

Sutula M, Kudela R, Hagy JD, Harding LW, Senn D, Cloern JE, Bricker S, Berg GM, Beck M. 2017. Novel analyses of long-term data provide a scientific basis for chlorophyll-*a* thresholds in San Francisco Bay. Estuarine Coast Shelf Sci. [accessed 2022 Jul 22];197:107–118. *https://doi.org/10.1016/j.ecss.2017.07.009*

Tango PJ, Batiuk RA. 2013. Deriving Chesapeake Bay water quality standards. J Am Water Resource Assoc. [accessed 2022 Jul 22];49:1007–1024. *https://doi.org/10.1111/jawr.12108*

Tatters AO, Howard MDA, Nagoda C, Fetscher AE, Kudela RM, Caron DA. 2019. Heterogeneity of toxin-producing cyanobacteria and cyanotoxins in coastal watersheds of southern California. Estuaries Coasts. [accessed 2022 Jul 22];42:958–975. *https://doi.org/10.1007/s12237-019-00546-w*

Tatters AO, Smith J, Kudela RM, Hayashi K, Howard MDA, Donovan AR, Loftin KA, Caron DA. 2021. The tide turns: episodic and localized cross-contamination of a California coastline with cyanotoxins. Harmful Algae. [accessed 2022 Jul 22];103:102003. *https://doi.org/10.1016/j.hal.2021.102003*

Thronson CH. 2020. The Delta's blooming problem. [accessed 2022 Jul 22]. Estuary News. Available from: *https://www.sfestuary.org/estuary-news-the-deltas-blooming-problem/*

Tonk L, Bosch K, Visser PM, Huisman J. 2007. Salt tolerance of the harmful cyanobacterium *Microcystis aeruginosa*. Aquat Microb Ecol. [accessed 2022 Jul 22];46(2):117–123. *https://doi.org/10.3354/ame046117*

Trainer VL, Moore SK, Hallegraeff G, Kudela RM, Clement A, Mardones JI, Cochlan WP. 2020. Pelagic harmful algal blooms and climate change: lessons from nature's experiments with extremes. Harmful Algae. [accessed 2022 Jul 22];91:101591. *https://doi.org/10.1016/j.hal.2019.03.009*

Trowbridge PR, Davis JA, Mumley T, Taberski K, Feger N, Valiela L, Ervin J, Arsem N, Olivieri A, Carroll P, et al. 2016. The regional monitoring program for water quality in San Francisco Bay, California, USA: science in support of managing water quality. Reg Studies Mar Sci. [accessed 2022 Jul 22];4:21–33. *http://doi.org/10.1016/j.rsma.2015.10.002*

Urquhart EA, Schaeffer BA, Stumpf RP, Loftin KA, Werdell, PJ. 2017. A method for examining temporal changes in cyanobacterial harmful algal bloom spatial extent using satellite remote sensing. Harmful Algae. [accessed 2022 Jul 22];67:144–152. *https://doi.org/10.1016/j.hal.2017.06.001*

[USEPA] US Environmental Protection Agency. 2015a. Drinking water health advisory for the cyanobacterial microcystin toxins. EPA-820R15100. Washington (DC): US EPA Department of Water. [accessed 2022 Jul 22]. Available from: *https://www.epa.gov/sites/default/files/2017-06/documents/microcystins-report-2015.pdf*

[USEPA] US Environmental Protection Agency. 2015b. Drinking water health advisory for the cyanobacterial microcystin toxins. Washington (DC): US EPA Department of Water. (EPA-820R1501). [accessed 2022 Jul 22]. Available from: *https://www.epa.gov/sites/default/files/2017-06/documents/microcystins-report-2015.pdf*

[USEPA] US Environmental Protection Agency. 2016. Method 546: Determination of total microcystins and nodularins in drinking water and ambient water by adda enzyme-linked immunosorbent assay. [accessed 2022 Jul 22]. Available from: *https://www.epa.gov/sites/default/files/2016-09/documents/method-546-determination-total-microcystins-nodularins-drinking-water-ambient-water-adda-enzyme-linked-immunosorbent-assay.pdf*

[USEPA] US Environmental Protection Agency. 2019. Recommended human health recreational ambient water quality criteria or swimming advisories for microcystins and cylindrospermopsin. Washington (DC): US EPA Department of Water. EPA 822-R-19-001. [accessed 2022 Jul 22]. Available from: *https://www.epa.gov/sites/default/files/2019-05/documents/hh-rec-criteria-habs-document-2019.pdf*

Van Dolah FM. 2000. Marine algal toxins: origins, health effects, and their increased occurrence. Environ Health Persp. [accessed 2022 Jul 22];108(suppl 1):133–141. *https://doi.org/10.1289%2Fehp.00108s1133*

Varga A, Puschner B. 2012. Retrospective study of cattle poisonings in California: recognition, diagnosis, and treatment. Vet Med Res Rep. [accessed 2022 Jul 22];3:111–127 *https://doi.org/10.2147%2FVMRR.S28770*

Vesterkvist PS, Misiorek JO, Spoof LE, Toivola DM, Meriluoto JA. 2012. Comparative cellular toxicity of hydrophilic and hydrophobic microcystins on Caco-2 cells. Toxins. [accessed 2022 Jul 22];4(11):1008–1023. *https://doi.org/10.3390/toxins4111008*

Vico P, Bonilla S, Cremella B, Aubriot L, Iriarte A, Piccini C. 2020. Biogeography of the cyanobacterium *Raphidiopsis* (*Cylindrospermopsis*) *raciborskii*: integrating genomics, phylogenetic and toxicity data. Mol Phylogenetics Evol. [accessed 2022 Jul 22];148:106824. *https://doi.org/10.1016/j.ympev.2020.106824*

Visser PM, Passarge J, Mur LR. 1997. Modelling vertical migration of the cyanobacterium *Microcystis*. Hydrobiologia. [accessed 2022 Jul 22];349:99–109. *https://doi.org/10.1023/A:1003001713560*

Vlamis A, Katikou P. 2014. Ecobiology and geographical distribution of potentially toxic marine dinoflagellates. In: Botana LM, editor. Seafood and freshwater toxins: pharmacology, physiology and detection. 3rd ed. [Accessed 2022 Jul 22]. p 569-625.

Wagner ND, Quach E, Buscho S, Ricciardelli A, Kannan A, Naung SW, Phillip G, Sheppard B, Ferguson L, Allen A, et al. 2021. Nitrogen form, concentration, and micronutrient availability affect microcystin production in cyanobacterial blooms. Harmful Algae. [accessed 2022 Jul 22];103:102002. *https://doi.org/10.1016/j.hal.2021.102002*

Wainger LA, Secor DH, Gurbisz C, Kemp WM, Glibert PM, Houde ED, Richkus J, Barber MC. 2017. Resilience indicators support valuation of estuarine ecosystem restoration under climate change. Ecosyst Health Sust. [accessed 2022 Jul 22];3:e01268. *https://doi.org/10.1002/ehs2.1268*

Wallace BB, Hamilton DP. 1999. The effect of variation in irradiance on buoyancy regulation in *Microcystis aeruginosa*. Limnol Oceanogr. [accessed 2022 Jul 22];44:273–281.

Wang P, Du B, Smith J, Lao W, Wong CS, Zeng EY. 2022. Development and field evaluation of the organic-diffusive gradients in thin-films (o-DGT) passive water sampler for microcystins. Chemosphere [accessed 2022 Jul 22];287:132079. *https://doi.org/10.1016/j.chemosphere.2021.132079*

Wang R, Chen H, Bubenheim D, Moran P, Zhang M. 2021. Modeling nitrogen runoff from Sacramento and San Joaquin River basins to Bay Delta Estuary: current status and ecological implications. J Aquatic Plant Mgmt. [accessed 2022 Jul 22];59:107–111.

Wells ML, Karlson B, Wulff A, Kudela R, Trick C, Asnaghi V, Berdalet E, Cochlan W, Davidson K, De Rijcke M, et al. 2020. Future HAB science: directions and challenges in a changing climate. Harmful Algae. [accessed 2022 Jul 22];91:101632. *https://doi.org/10.1016/j.hal.2019.101632*

Wells ML, Trainer VL, Smayda TJ, Karlson BSO, Trick CG, Kudela RM, Ishikawa A, Bernard S, Wulff A, Anderson DM, et al. 2015. Harmful algal blooms and climate change: learning from the past and present to forecast the future. Harmful Algae. [accessed 2022 Jul 22];49:68–93. *https://doi.org/10.1016/j.hal.2015.07.009*

[WHO] World Health Organization. 2006. Guidelines for drinking-water quality: incorporating first addendum. 3rd ed. 2004: World Health Organization. [accessed 2022 Jul 22]. Available from: *https://apps.who.int/iris/bitstream/handle/10665/43428/9241546964_eng.pdf*

[WHO] World Health Organization. 2020a. Cyanobacterial toxins: anatoxin-a and analogues. Background document for development of WHO Guidelines for drinking-water quality and guidelines for safe recreational water environments. Geneva: WHO. WHO/HEP/ECH/WSH/2020.1. [accessed 2022 Jul 22]. Available from: *https://apps.who.int/iris/bitstream/handle/10665/338060/WHO-HEP-ECH-WSH-2020.1-eng.pdf?sequence=1&isAllowed=y*

[WHO] World Health Organization. 2020b. Cyanobacterial toxins: cylindrospermopsins. Background document for development of WHO guidelines for drinking-water quality and guidelines for safe recreational water environments. Geneva: WHO. WHO/HEP/ECH/WSH/2020.4. [accessed 2022 Jul 22]. Available from: *https://apps.who.int/iris/bitstream/handle/10665/338063/WHO-HEP-ECH-WSH-2020.4-eng.pdf?sequence=1&isAllowed=y*

[WHO] World Health Organization. 2020c. Cyanobacterial toxins: microcystins. Background document for development of WHO guidelines for drinking-water quality and Guidelines for safe recreational water environments. Geneva: WHO. WHO/HEP/ECH/WSH/2020.6. [accessed 2022 Jul 22]. Available from: *https://apps.who.int/iris/bitstream/handle/10665/338066/WHO-HEP-ECH-WSH-2020.6-eng.pdf*

[WHO] World Health Organization. 2020d. Cyanobacterial toxins: saxitoxins. Background document for development of WHO guidelines for drinking-water quality and guidelines for safe recreational water environments. Geneva: WHO. WHO/HEP/ECH/WSH/2020.8. [accessed 2022 Jul 22]. Available from: *https://apps.who.int/iris/bitstream/handle/10665/338069/WHO-HEP-ECH-WSH-2020.8-eng.pdf?sequence=1&isAllowed=y*

Winder M, Carstensen J, Galloway AWE, Jakobsen HH, Cloern JE. 2018. The land–sea interface: a source of high-quality phytoplankton to support secondary production. Limnol Oceanogr. [accessed 2022 Jul 22];62:S258–S271. *https://doi.org/10.1002/lno.10650*

Wright SA, Schoellhamer DH. 2004. Trends in the sediment yield of the Sacramento River, California, 1957–2001. San Franc Estuary Watershed Sci. [accessed 2022 Jul 22];2(2):1–14. *https://doi.org/10.15447/sfews.2004v2iss2art2*

Wolny JL, Tomlinson MC, Schollaert Uz SS, Egerton TA, McKay JR, Meredith A, Reece KS, Scott GP, Stumpf RP. 2020. Current and future remote sensing of harmful algal blooms in the Chesapeake Bay to support the shellfish industry. Front Mar Sci. [accessed 2022 Jul 22];337. *https://doi.org/10.3389/fmars.2020.00337*

Wu H, Wu, X, Yang T, Wang C, Tian C, Xiao B, Lorke A. 2021. Feedback regulation of surface scum formation and persistence by self-shading of *Microcystis* colonies: numerical simulations and laboratory experiments. Water Res. [accessed 2022 Jul 22];194:116908. *https://doi.org/10.1016/j.watres.2021.116908*

Xiao M, Li M, Reynolds CS. 2018. Colony formation in the cyanobacterium E. Biol Rev. [accessed 2022 Jul 22];93:1399–1420. *https://doi.org/10.1111/brv.12401*

Young N, Sharpe RA, Barciela R, Nichols G, Davidson K, Berdalet E, Fleming LE. 2020. Marine harmful algal blooms and human health: a systematic scoping review. Harmful Algae. [accessed 2022 Jul 22];98:101901. *https://doi.org/10.1016/j.hal.2020.101901*

Zehr JP, Kudela RM. 2011. Nitrogen cycle of the open ocean: from genes to ecosystems. Ann Rev Mar Sci. [accessed 2022 Jul 22];3:197–225. *https://doi.org/10.1146/annurev-marine-120709-142819*

Zhou ZX, Yu RC, Zhou MJ. 2017. Resolving the complex relationship between harmful algal blooms and environmental factors in the coastal waters adjacent to the Changjiang River estuary. Harmful Algae. [accessed 2022 Jul 22];1:62:60–7. *https://doi.org/10.1016/j.hal.2016.12.006*

NOTES

Plaice A. 2015. Email communication sent 3/17/15 to RMK about Karlodinium cell count thresholds. Available from: *place@umces.edu*.

Senn D, Kudela RM. Unpublished data collected as part of the San Francisco Estuary Institute Nutrient Management Strategy to assess the role of nutrients in harmful algal toxin distribution found in anchovy within San Francisco Bay. Available from: *davids@sfei.org*.