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Harmful algae and their potential impacts on desalination operations off southern California

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ABSTRACT

Seawater desalination by reverse osmosis (RO) is a reliable method for augmenting drinking water supplies. In recent years, the number and size of these water projects have increased dramatically. As freshwater resources become limited due to global climate change, rising demand, and exhausted local water supplies, seawater desalination will play an important role in the world's future water supply, reaching far beyond its deep roots in the Middle East. Emerging contaminants have been widely discussed with respect to wastewater and freshwater sources, but also must be considered for seawater desalination facilities to ensure the long-term safety and suitability of this emerging water supply. Harmful algal blooms, frequently referred to as 'red tides' due to their vibrant colors, are a concern for desalination plants due to the high biomass of microalgae present in ocean waters during these events, and a variety of substances that some of these algae produce. These compounds range from noxious substances to powerful neurotoxins that constitute significant public health risks if they are not effectively and completely removed by the RO membranes. Algal blooms can cause significant operational issues that result in increased chemical consumption, increased membrane fouling rates, and in extreme cases, a plant to be taken off-line. Early algal bloom detection by desalination facilities is essential so that operational adjustments can be made to ensure that production capacity remains unaffected. This review identifies the toxic substances, their known producers, and our present state of knowledge regarding the causes of toxic episodes, with a special focus on the Southern California Bight.

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1. Introduction

1.1. *General overview of harmful algal blooms: a growing global concern*

Microscopic algae constitute an essential component of all aquatic food webs. Photosynthetic production of organic material by this diverse group of species comprises the primary source of nutrition for all heterotrophic forms of life in much of the world's ocean and freshwater ecosystems. Microalgae can reach high abundances in the plankton during periods of optimal growth and reduced grazing pressure by herbivores. Such localized mass proliferations are known as algal (or phytoplankton) blooms. In addition, a small proportion of microalgal species are capable of producing a number of noxious or toxic compounds that cause a variety of adverse effects on ecosystem structure and function. These substances pose the potential for ecosystem damage, food web disruption and marine animal mortality, and present a significant human health risk through the consumption of contaminated seafood and, in at least one case, direct exposure to water or aerosols containing these toxic compounds. Additionally, the algal biomass and the associated organic load cause significant desalination operational issues, impacting the pretreatment system and possibly forcing the treatment plant to be taken off-line (Petry et al., 2007).

Countless human deaths resulting from the consumption of seafood contaminated with algal toxins have been avoided through rigorous monitoring programs, but sea life has not been so fortunate. Approximately one half of all unusual marine mammal mortality incidents are now attributable to the ingestion of food or prey contaminated by harmful algal blooms (Ramsdell et al., 2005). Losses in revenue due to the direct contamination of seafood products and indirect effects on tourism and other uses of coastal areas have been estimated in the tens of millions of dollars annually in the U.S. states along the Pacific coast (Trainer et al., 2002).

There is now convincing evidence that harmful algal bloom (HAB) events are increasing at local, regional and global scales worldwide (Smayda, 1990; Hallegraeff, 1993, 2003; Anderson et al., 2002; Glibert et al., 2005a) and along the North American west coast in particular (Horner et al., 1997; Trainer et al., 2003). This increased occurrence may be due in part to better detection of HAB episodes in recent years or the global dispersal of toxic algal species via the transport of resting spores in ships' ballast waters (Hallegraeff and Bolch, 1992; Burkholder et al., 2007), but another very likely cause is the increasing impact of anthropogenic activities on coastal ecosystems (Smayda, 1990; Anderson et al., 2002; Glibert et al., 2005b, 2006; Howard et al., 2007; Cochlan et al., 2008; Kudela et al., 2008a). Recent reports reveal extensive and, in some cases, newly emerging occurrences of HABs along the coasts of the U.S. (Fig. 1). These incidents engender a variety of noxious impacts on ecosystems and public health, including direct effects on organisms due to the production of acutely toxic substances, and indirect effects such as reduced availability of dissolved oxygen in the water column resulting from the decomposition of the extensive amounts of organic substances usually produced during such blooms. The dramatic increases in biomass and organic load that accompany these events pose a significant threat to seawater desalination facilities (Gaid and Treal, 2007).

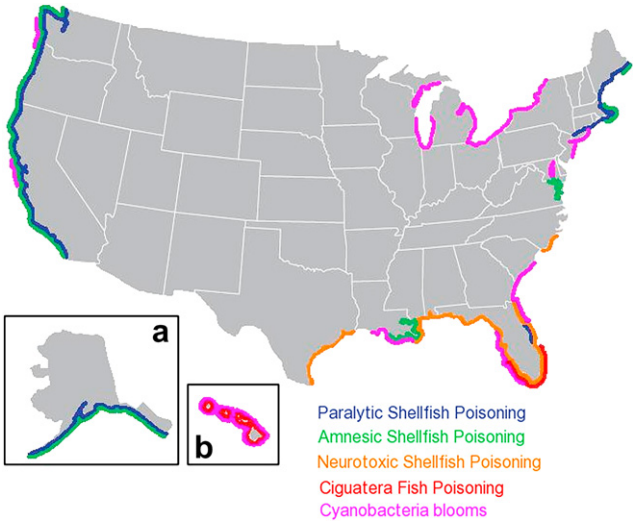


Fig. 1 – Distribution of some well-known regional HAB issues along U.S. shores, including (a) Alaska and (b) Hawaii. Causes and impacts of these poisoning events are defined in Tables 1–3. Summarized from information presented on the Harmful Algae webpage (http://www.whoi.edu/redtide/).

1.2. *Regional HAB issues along U.S. coastlines*

Harmful algae are present throughout U.S. coastal waters, but not all species are of equal concern in all regions (Fig. 1). For example, toxic species of the dinoflagellate genus *Alexandrium* are common over vast stretches of the U.S. coastline, but coastal regions of the northeastern and northwestern U.S. appear to experience particularly high rates of occurrence of toxic 'red tides' caused by these species. The neurotoxins produced by *Alexandrium*, called saxitoxins, cause paralytic shellfish poisoning (PSP) in humans when ingested through contaminated seafood (particularly filter-feeding shellfish). Similarly, several toxic species of the diatom genus *Pseudo-nitzschia* occur along the entire U.S. coastline but significant concentrations of the neurotoxin, domoic acid, produced by these species have historically constituted a health threat primarily in the northeastern and northwestern U.S. (Bates et al., 1989) where it has been documented as the cause of amnesic shellfish poisoning (ASP) in humans. However, high concentrations of domoic acid in the plankton and in diverse planktivorous organisms have been recently documented along the entire Pacific coast of the U.S. (Scholin et al., 2000; Trainer et al., 2002; Schnetzer et al., 2007), as well as in the Gulf of Mexico (Pan et al., 1998). Domoic acid has been attributed to numerous marine animal mortalities along the U.S. west coast. In the Gulf of Mexico, primarily along the west coast of Florida, extensive and recurrent blooms of the dinoflagellate *Karenia brevis* produce a suite of toxins, known as brevetoxins, that can be aerosolized by breaking waves and

induce neurotoxic shellfish poisoning (NSP) in people inhaling the aerosols (see review of Kirkpatrick et al., 2004). The Tampa Bay seawater desalination facility is the only operating seawater desalination treatment plant of significant size in the United States. It is located along the west coast of Florida and is likely to encounter algal blooms that contain brevetoxin.

Less toxic blooms also take place with regional specificity. The pelagophyte *Aureococcus anophagefferens* causes 'brown tides' in coastal waters of Rhode Island, near Long Island (NY) and southward along the mid-Atlantic coast of the U.S. since 1985. No specific toxins have been identified from *A. anophagefferens*, and no human fatalities have been directly attributed to these blooms. Nevertheless, this species appears to be unpalatable or inhibitory to many filter-feeding mollusks and has caused substantial mortality among these populations, including commercially valuable species (Bricelj and Lonsdale, 1997).

Other microalgal species can disrupt food webs or cause reductions in water quality without producing acutely toxic conditions. Among these are the 'colorful' red tides of the dinoflagellate *Lingulodinium polyedrum*, a yessotoxin producer, that have occurred periodically throughout several decades along the south and central Californian coasts (Horner et al., 1997; Gregorio and Pieper, 2000). These blooms have so far been found to be relatively innocuous in these waters but massive accumulations of these cells could have significant impact on desalination plants because of increased turbidity, high suspended solids and organic loading of influent water. Furthermore, accumulations of cells in protected harbors can cause fish mortality by depleting oxygen dissolved in the water, further challenging influent screening and pretreatment systems at desalination plants. Other taxa, such as species of the prymnesiophyte genus *Phaeocystis*, produce substances that can lead to enormous buildups of sea foam along coasts (Armonies, 1989).

1.3. *Desalination, plankton and water quality issues*

Large research programs have developed within different geographic areas throughout the U.S. to address regional HAB issues. These programs are designed to study the species, toxins and environmental causes of HAB outbreaks. These efforts, as well as local, county, state and federal monitoring programs provide basic information for marine resource use and have focused almost exclusively on threats to human health via the consumption of contaminated seafood. Unfortunately, few if any of these programs provide sufficient information on appropriate temporal and spatial resolution for thoroughly assessing the potential impact of HAB events on reverse osmosis desalination operations. Moreover, toxin analyses have primarily examined the presence of these substances in particulate material (plankton or animal tissue, particularly shellfish and finfish), and therefore may be poor predictors for the amount of toxins that might occur in seawater in the dissolved state during algal blooms, which would be most likely to be loaded onto reverse osmosis membranes during desalination.

There are two potential impacts that HABs may have on seawater desalination facilities: (1) algal toxins in ocean water pose a significant treatment challenge for the reverse osmosis system to ensure that these molecules are effectively removed and (2) increased turbidity, total suspended solids and total organic content resulting from algal biomass and growth challenge the entire desalination facility's treatment train. The significance of these issues will depend on the specific algae forming a bloom and the toxin(s) or other substances that they produce, the magnitude and duration of the bloom, and the specific desalination process conducted. For example, multi-effect distillation and multi-flash distillation might be susceptible to (2) but would be much less affected by toxins in the water (1). Desalination using reverse osmosis presumably would be vulnerable to both issues. Therefore, for the latter desalination approach, a thorough understanding of HAB episodes in terms of incidence and seasonality, vertical and horizontal spatial distribution, as well as biological aspects such as algal composition within a geographical region could help optimize the design and operational efficiency of desalination plants employing reverse osmosis.

This paper provides an overview of HABs occurring along the continental U.S. coastline with special emphasis on the southwestern U.S., and provides some insight on the potential impacts that these events may have on the seawater desalination process. In recent years, this geographical area has become a focal point of discussions regarding desalination (Cooley et al., 2006) because of its sizable population and the particularly tenuous nature of the water supply to this region. Although numerous issues involving the desalination process are now being examined (Separation Processes Inc., 2005; Gaid and Treal, 2007; Pankratz, 2008, 2009), very limited information exists on the risks that algal blooms pose to seawater desalination facilities. A review of the major species producing harmful blooms, the substances they produce, and information on the spatial and temporal distributions of blooms are presented along with some conclusions on their potential impacts. This paper also provides some general guidelines on how early detection may help prevent or minimize the impact of HABs on a desalination facility's production capacity or its water quality.

2. Toxin producers and toxin concentrations of the west coast

A variety of toxins including several powerful neurotoxins are produced by microalgae, and a number of these toxins and potentially toxic algal species have been detected on the U.S. west coast (Table 1). The ability to rapidly detect and quantify toxic algae in natural water samples is problematic at this time. Many of these species are difficult to identify using light microscopy. For this reason, new genetic and immunological methods for species identification and enumeration have been appearing rapidly in the literature (Miller and Scholin, 1998; Bowers et al., 2000, 2006; Coyne et al., 2001; Caron et al., 2003; Galluzzi et al., 2004; Anderson et al., 2005; Mikulski et al., 2005, 2008; Ahn et al., 2006; Handy et al., 2006; Moorthi et al., 2006; Iwataki et al., 2007, 2008; Demir et al., 2008; Matsuoka et al., 2008). Moreover, many toxin-producing algal species exhibit variable toxin production in response to environmental conditions, and among different strains of the same species

Table 1 – Planktonic species occurring along the west coast of the U.S. that are potential concerns for reverse osmosis operations.

Microalgae	Toxin(s)	Poisoning Event	References
Diatoms *Pseudo-nitzschia* spp. *P. australis*[b] *P. cuspidata*[b] *P. delicatissima*[b] *P. fraudulenta*[b] *P. multiseries*[b] *P. pungens*[b] *P. pseudodelicatissima*[b] *P. seriata*[a]	Domoic acid (DA)	Amnesic Shellfish Poisoning (ASP) Human effects • Gastro-intestinal symptoms • Neurologic symptoms • Death Ecosystem effects • Marine mammal mortalities • Bird mortalities	Subba Rao et al. (1988), Bates et al. (1989), Martin et al. (1990), Buck et al. (1992), Garrison et al. (1992), Rhodes et al. (1996), Horner et al. (1997), Lundholm et al. (1997), Rhodes et al. (1998), Trainer et al. (2000, 2001), Baugh et al. (2006)
Dinoflagellates *Alexandrium* spp. *A. acatenella*[a] *A. catenella*[b] *A. fundyense*[a] *A. hiranoi*[a] *A. ostenfeldii*[a] *A. tamarense*[a]	Saxitoxins (STXs)	Paralytic Shellfish Poisoning (PSP) Human effects • Gastro-intestinal symptoms • Paralysis • Death Ecosystem effects • Marine mammal mortalities	Sommer and Meyer (1937), Gaines and Taylor (1985), Steidinger (1993), Scholin et al. (1994), Taylor and Horner (1994), Jester (2008)
Dinoflagellates *Lingulodinium polyedrum*[b] *Gonyaulax spinifera*[a] *Protoceratium reticulatum*[a,c]	Yessotoxins (YTXs)	Human and ecosystem effects • None reported	Holmes et al. (1967), Satake et al. (1997, 1999), Draisci et al. (1999a), Paz et al. (2004, 2007), Armstrong and Kudela (2006), Rhodes et al. (2006), Howard et al. (2007)
Dinoflagellates *Dinophysis* spp. *D. acuminata*[a] *D. acuta*[a] *D. caudate* *D. fortii*[a] *D. norvegica*[a] *D. rotundata*[a] *D. tripos*[a] *Prorocentrum* spp. *P. micans* *P. minimum*[a,d]	Okadaic acid (OA) Dinophysistoxins (DTXs) Pectenotoxins (PTXs)	Diarrhetic Shellfish Poisoning (DSP) Human effects • Gastro-intestinal symptoms Ecosystem effects • None reported	Holmes et al. (1967), Yasumoto et al. (1980), Murata et al. (1982), Yasumoto et al., (1985), Cembella (1989), Lee et al. (1989), Horner et al. (1997), Cembella (2003), Miles et al. (2004), Shipe et al. (2008), Sutherland (2008)
Raphidophytes *Chattonella marina*[a] *Fibrocapsa japonica*[a] *Heterosigma akashiwo*[a]	Brevetoxins (PbTxs)	Neurotoxic Shellfish Poisonining (NSP) Human effects • Gastroenteritis • Neurologic symptoms • Respiratory irritation and/or failure Ecosystem effects • Marine mammal mortalities • Fish mortality events	Loeblich and Fine (1977), Hershberger et al. (1997), Gregorio and Connell (2000), Hard et al. (2000), Tyrell et al. (2002), O'Halloran et al. (2006)

a Reported to produce toxin.
b Reported to produce toxin on the west coast of the United States.
c Conflicting reports on toxicity of *P. reticulatum* cultures isolated from California, Washington and Florida.
d Reported to be present on the west coast of Mexico.

even when isolated from the same geographic region (Smith et al., 2001; Trainer et al., 2001; Kudela et al., 2004).

Laboratory experiments have revealed a wide range of physico-chemical factors that increase or decrease toxin production by harmful species of algae, and which appear to be species-specific (see review of Granéli and Flynn, 2006). Reports of factors inducing toxin production have sometimes been conflicting, presumably indicating that multiple factors, or

perhaps generally stressful conditions, may stimulate toxin production. Factors affecting toxin production include: (1) temperature (Ono et al., 2000); (2) light intensity (Ono et al., 2000); (3) salinity (Haque and Onoue, 2002a,b); (4) trace metal availability, especially iron (Ladizinsky and Smith, 2000; Rue and Bruland, 2001; Maldonado et al., 2002; Wells et al., 2005; Sunda, 2006) but also copper (Maldonado et al., 2002) and selenium (Mitrovic et al., 2004, 2005); (5) macronutrient availability including silicate (Pan et al., 1996b; Fehling et al., 2004; Kudela et al., 2004), phosphate (Pan et al., 1996a, 1998; Fehling et al., 2004), nitrogen (Bates et al., 1991; Pan et al., 1998; Kudela et al., 2004) and combinations of nutrient limitation (Anderson et al., 1990; Flynn et al., 1994; John and Flynn, 2000); (6) cellular elemental ratios of nutrients and physiological stress (Granéli and Flynn, 2006; Schnetzer et al., 2007); (7) growth phase (Anderson et al., 1990; Bates et al., 1991; Flynn et al., 1994; Johansson et al., 1996; Maldonado et al., 2002; Mitrovic et al., 2004). The precise combination(s) of environmental factors that select for population growth of particular algal species within diverse natural assemblages, and the specific conditions that induce toxin production, are poorly understood for most harmful algae. This present state of knowledge makes it difficult to predict the timing, duration or spatial extent of the vast majority of HAB events and the toxic events resulting from them.

Our ability to thoroughly characterize HABs is also complicated by the complex array of toxins produced by algae. Marine algal species produce a suite of toxic components (Yasumoto and Murata, 1993), and unidentified toxins undoubtedly remain to be described. Additionally, most toxins are actually composed of families of closely related compounds. Slightly different forms of a toxin can exhibit very different levels of toxicity, or may be characterized differently by some detection methods and analytical approaches (Garthwaite et al., 2001; Lefebvre et al., 2008). Such complexity and variability can sometimes yield vague or contradictory conclusions regarding the exact source of toxicity in a natural sample (Bates et al., 1978; Paz et al., 2004, 2007).

Finally, characterization of HAB events is complicated by inherent difficulties associated with linking specific toxins measured in natural water samples to a specific algal species in a complex, natural phytoplankton assemblage and, as noted above, the presence of toxic species in a water sample does not necessarily indicate the presence of toxins. Despite these shortcomings, there is considerable knowledge of many of the major algal toxins and their producers in U.S. coastal waters that constitute the most important potential concerns for desalination activities because they are the most likely to be encountered in ocean water intakes.

2.1. Domoic acid

2.1.1. Toxin description and activity

Domoic acid (Fig. 2; Table 3) is an amino acid derivative belonging to the kainoid class of compounds containing three carboxyl groups and one secondary amino group (Wright et al., 1990; Jeffery et al., 2004). All four groups are charged at neutral pH, and the carboxyl groups become successively protonated as pH decreases, yielding five possible protonated forms of domoic acid (Quilliam, 2003; Jeffery et al., 2004). There are currently ten known isomers of domoic acid, including the isodomoic acids A through H and the domoic acid 5' diastereomer (Jeffery et al., 2004).

Domoic acid and other members of the kainoid class are glutamate analogues that interfere with neurochemical pathways by binding to glutamate receptors of brain neurons (Wright et al., 1990; Quilliam, 2003). The resulting effect of these neuroexcitants, or excitotoxins, is a continuous stimulation of the neurons, which can lead to rupture and/or eventual formation of lesions (Wright et al., 1990). Depolarized neurons result in short-term memory loss (Clayden et al., 2005), which has led to the common name for the illness related to the consumption of seafood contaminated with domoic acid: amnesic shellfish poisoning (ASP). Symptoms of ASP include gastroenteritis (vomiting, diarrhea, abdominal cramps) that can be experienced in humans within 24 h after ingestion, and neurological symptoms of confusion, memory loss, disorientation, seizures, coma and/or cranial nerve palsies that are typically experienced within 48 h (Perl et al., 1990; Wright et al., 1990). The number of human illnesses resulting from domoic acid poisoning has been few (Horner et al., 1997), likely due to active monitoring of fisheries. However, cultured blue mussels (*Mytilus edilus*) contaminated with domoic acid poisoned 107 people and killed three during the first major documented ASP outbreak in 1987 on Prince Edward Island, Canada (Perl et al., 1990).

ASP poses a serious threat to marine wildlife, and the deaths of thousands of marine mammals and sea birds have been attributed to domoic acid intoxication (Bates et al., 1989; Scholin et al., 2000; Gulland et al., 2002; Caron et al., unpublished data). The first documented poisoning episode of marine animals related to domoic acid on the U.S. west coast was attributed to *Pseudo-nitzschia australis* and occurred in September 1991 off central California (Table 2; Buck et al., 1992; Fritz et al., 1992). High concentrations of domoic acid were also detected in Washington and Oregon in the 1990s (Wekell et al., 1994; Adams et al., 2000; Trainer et al., 2002), and a decade later in coastal waters off southern California (Schnetzer et al., 2007). The frequency and severity of these toxic events appears to be increasing (Trainer et al., 2007).

2.1.2. Producers

The production of domoic acid and its isomers is confined to approximately a dozen chain-forming marine pennate diatom species within the genus *Pseudo-nitzschia* (Bates and Trainer, 2006), a genus containing species that form long chains of cells attached at their ends (Fig. 3a and b). The main toxin producing species that have been documented on the U.S. west coast include: *P. australis*, *P. delicatissima*, *P. fraudulenta*, *P. multiseries*, *P. pungens*, *P. pseudodelicatissima*, *P. seriata* and *P. cuspidata* (Tables 1 and 2). These species are distinguished based on fine morphological features of their silica frustules (Fig. 3a and b). These distinctions are subtle and require careful electron microscopical analysis and elaborate taxonomic training. As a consequence, historical misidentifications are not unusual and debates regarding some species descriptions are still unresolved.

It is surprising that the first reports of ASP on the west coast of the U.S. were not recorded until the 1990s, even though *Pseudo-nitzschia* species have been recorded in surveys of

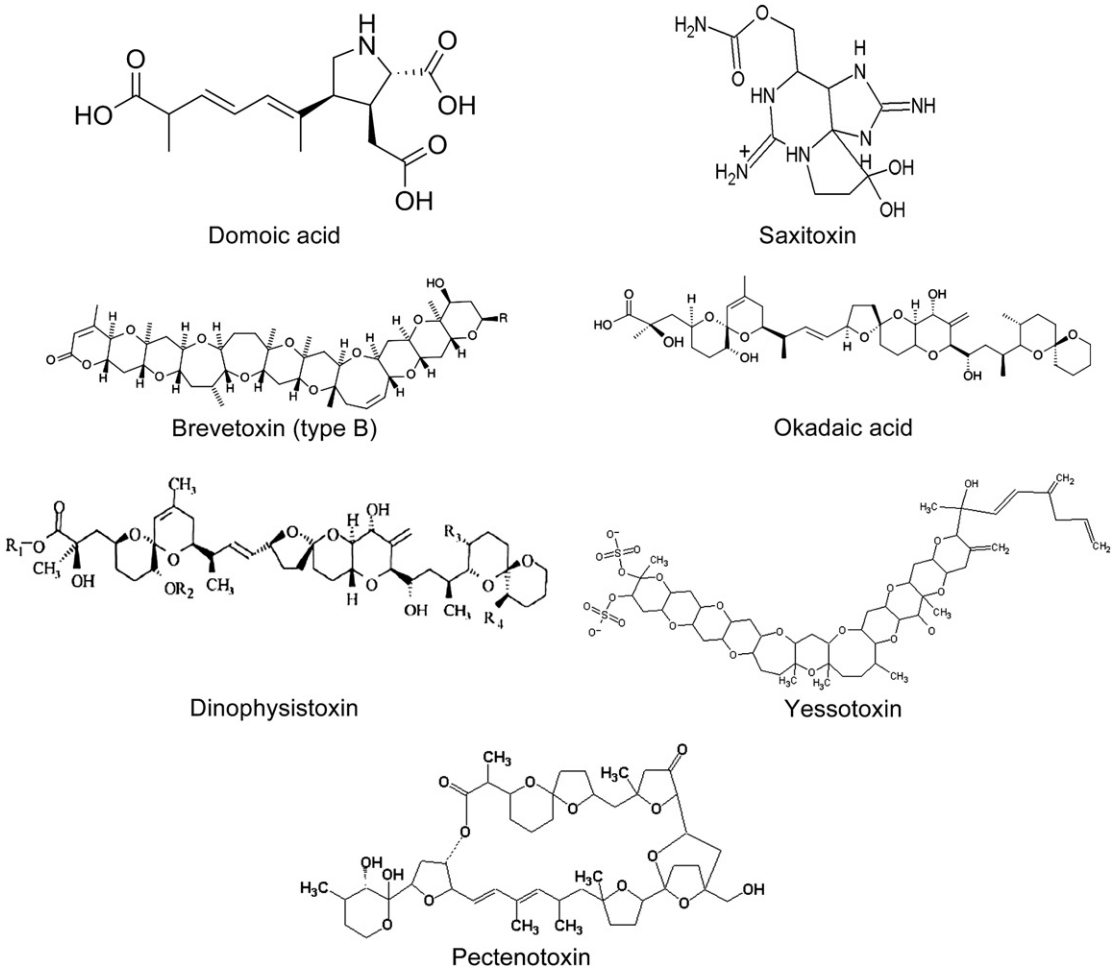


Fig. 2 – Chemical structures of commonly encountered toxins produced by microalgae in U.S. coastal waters.

phytoplankton species in the Southern California Bight since 1917 (Allen, 1922, 1924, 1928, 1936, 1940, 1941; Reid et al., 1970, 1985; Lange et al., 1994; Fryxell et al., 1997; Thomas et al., 2001). Given that these species generally comprise a significant portion of the total diatom assemblage in these waters, it can be surmised that either toxin production has increased in these west coast species, or that poisoning events prior to the 1990s have occurred but have not been attributed to these diatoms. Historical accounts of 'unusual animal mortality events' along the U.S. west coast tend to support the latter hypothesis.

There have been increasing numbers of toxic events recorded along the U.S. west coast (Table 2), notably in Puget Sound (Trainer et al., 2003, 2007), Monterey Bay (Vigilant and Silver, 2007; R. Kudela, unpubl. data), Santa Barbara Channel (Trainer et al., 2000; Anderson et al., 2006; Mengelt, 2006), San Pedro Channel (Busse et al., 2006; Schnetzer et al., 2007), Newport Beach (Busse et al., 2006) and San Diego (Lange et al., 1994; Busse et al., 2006). Most recently, toxic blooms of *Pseudo-nitzschia* in the Long Beach-Los Angeles Harbor and San Pedro Channel have been particularly toxic, with some of the highest domoic acid concentrations recorded for the U.S. west coast (Caron et al., unpublished data). The increased incidence and severity of these toxic episodes off the western U.S. coast parallels the increase in frequency and intensity of harmful algal blooms observed globally (Smayda, 1990; Hallegraeff, 1993, 2003; Anderson et al., 2002; Glibert et al., 2005b).

2.2. *Saxitoxins*

2.2.1. *Toxin description and activity*

Saxitoxin is a complex guanidine-based alkaloid that exists as more than 30 identified analogues in nature (Llewellyn, 2006). It is the most powerful marine toxin currently known and among the most dangerous poisons on Earth, except for some venoms and bacterial toxins (Schantz et al., 1957). Due to its acute toxicity, saxitoxin is currently listed as a chemical weapon in Schedule 1 of the Chemical Weapons Convention (Llewellyn, 2006). Saxitoxins display a rigid tricyclic core (Fig. 2; Table 3) and are very stable in biological and physiological solutions (Rogers and Rapoport, 1980). This nitrogen-rich molecule and its chemical relatives are polar and have a positive charge at pH 7.7 (Shimizu et al., 1981). Consequently, they are soluble in water and alcohols, and insoluble in organic solvents (Schantz et al., 1957).

Saxitoxins are known to disrupt the flow of ions through voltage gate sodium channels (Catterall, 1992; Cestele and Catterall, 2000). It has also been recently discovered that they have the ability to bind to calcium (Su et al., 2004) and

Table 2 – Distribution and concentrations of marine toxins in plankton of confirmed toxin producers in U.S. west coast waters.

Toxin(s)	Location and year	Causative species	Particulate $\mu g\,L^{-1}$ ($nmol\,L^{-1}$)	Cellular $pg\,cell^{-1}$	Dissolved $pg\,mL^{-1}$ ($nmol\,L^{-1}$)	References
Domoic acid	Washington coast and Juan de Fuca Eddy, WA (1997, 1998)	*P. pseudodelicatissima* *Pseudo-nitzschia* spp.	b.d.–2.7 3.6–8.7	b.d.–4.6		Adams et al. (2000), Trainer et al. (2001, 2002)
	Penn Cove, WA (1997)	*P. pungens* *P. multiseries* *P. australis* *P. pseudodelicatissima*	b.d.–0.8			Trainer et al. (1998)
	Washington coast, WA (2001)	*P. australis*	b.d.–0.03			Marchetti et al. (2004)
	Washington coast, WA (2003)	*Pseudo-nitzschia* spp.	(0.4–15)	2×10^{-4}–0.3[a] 0.1–94.4	(b.d.–4.3)[a] (1–5)	Baugh et al. (2006)
	Puget Sound, WA (2005)	*P. pseudodelicatissima* *Pseudo-nitzschia* spp.	b.d.–14			Trainer et al. (2007)
	Central Oregon coast, OR (1998)	*P. australis*	0.5	35		Trainer et al. (2001)
	Pt. Año Nuevo, San Francisco, CA (1998)	*P. pungens* *P. multiseries*	0.1–0.7	0.3–6.3		Trainer et al. (2000)
	Bolinas Bay, San Francisco, CA (2003)	*P. australis*	0.15–9.4			Howard et al. (2007)
	Monterey Bay, CA (1991, 1998)	*P. australis*	b.d.–12.3 0.1–6.7	3–37 7.2–75		Buck et al. (1992), Garrison et al. (1992), Walz et al. (1994), Scholin et al. (2000)
	Monterey Bay, CA (1998)	*P. pseudodelicatissima* *P. multiseries*	0.1–0.4 0.67	0.8–1.2 6		Trainer et al. (2000, 2001)
	Monterey Bay, CA (2000)	*Pseudo-nitzschia* spp. *P. australis*		b.d.–24	b.d.–8491	Bargu et al. (2002, 2008)
	Monterey Bay, CA (2002–2003)	*Pseudo-nitzschia* spp.	24			Vigilant and Silver (2007)
	Morro Bay, CA (1998)	*P. australis*	1.3–7.4	37–78		Trainer et al. (2000, 2001)
	San Luis Obispo, CA (2003–2005)	*P. australis* *P. multiseries*	1.5–7.6	9–38		Mengelt (2006)
	Point Conception, CA (1998)	*P. australis*	2.2–6.3	15–22		Trainer et al. (2000)
	Santa Barbara, CA (1998)	*P. australis* *P. pungens* *P. pseudodelicatissima*	0.5–1.2	0.1–0.9		Trainer et al. (2000)

(*continued on next page*)

Table 2 (*continued*)

Toxin(s)	Location and year	Causative species	Particulate $\mu g\,L^{-1}$ ($nmol\,L^{-1}$)	Cellular $pg\,cell^{-1}$	Dissolved $pg\,mL^{-1}$ ($nmol\,L^{-1}$)	References
	Santa Barbara Channel, CA (2003)	*P. australis*	0.03–1.7	0.14–2.1		Anderson et al. (2006)
	Santa Barbara (Santa Rosa Island and north San Miguel) (2004)	*P. australis* *P. multiseries*	6–12	b.d.–80		Mengelt (2006)
	Southern California Bight, CA (2003, 2004)	*Pseudo-nitzschia* spp. *P. australis* *P. cuspidata*	5.6–12.7	b.d.–117		Schnetzer et al. (2007)
	San Diego and Orange counties, CA (2004)	*P. australis* *P. multiseries*	b.d.–2.33			Busse et al. (2006)
Saxitoxins	Sequim Bay, WA (2004–2007)	*Alexandrium* spp.	0.02–0.5		150–800	Lefebvre et al. (2008)
	Oregon coast, OR (2004)	*Alexandrium* spp.	0.004–0.028			Jester et al. (unpublished data)
	Humboldt Bay, CA (2004)	*A. catenella*		1.6–19[a]		Jester (2008)
	San Mateo County coast, CA (2004)	*A. catenella*		2.1–62.6[a]		Jester (2008)
	Monterey Bay, CA (2004)	*A. catenella*		0.6–31.3[a]		Jester (2008)
	Monterey Bay, CA (2003–2005)	*A. catenella*	b.d.–0.962			Jester et al. (2009b)
	Morro Bay, CA (2004)	*A. catenella*		1.4–16.6[a]		Jester (2008)
Yessotoxin	La Jolla, CA (1993)	*Lingulodinium polyedrum*		0.002–0.02[a] 0–0.05[a]		Armstrong and Kudela (2006) Howard et al. (2008)
Brevetoxins	Indian Inlet, Bald Eagle Creek and Torque Canal, DE (2000)	*Chattonella* cf. *verruculosa*	0.008–<0.2	6		Bourdelais et al. (2002)

b.d.: Below detection limit.
a Toxin concentration from cells in culture.

potassium channels (Wang et al., 2003) and to be a weak inhibitor of neuronal nitric oxide synthase (reviewed in Llewellyn, 2006). These activities directly affect the nervous system, and the consumption of seafood containing saxitoxin can result in serious human illness and death, commonly referred to as paralytic shellfish poisoning (PSP).

Minor symptoms of PSP, such as burning or tingling sensation of the lips and face, dizziness, headache, salivation, intense thirst and perspiration, vomiting, diarrhea and stomach cramps, can be experienced within 30 min after the consumption of contaminated seafood (Llewellyn, 2006). The consumption of a lethal dose can result in death within hours due to muscular paralysis and respiratory difficulty followed by complete respiratory arrest. PSP outbreaks result in more than 2000 illnesses worldwide each year, with a 5–10% mortality rate (Hallegraeff, 2003). PSP toxins also have adverse effects on marine wildlife that can cause mortalities among fish, marine mammal and seabird populations (Geraci et al., 1989; Montoya et al., 1996; Shumway et al., 2003). There are presently no records of unusual animal mortality events along the Californian coast that are attributable to saxitoxin poisoning (Jester et al., 2009b), but occurrence of the toxins in species consumed by humans is sufficient to warrant year-round monitoring.

2.2.2. *Producers*

Saxitoxins are biosynthesized by dinoflagellates in marine ecosystems, most notably species within the genus *Alexandrium* (Fig. 3c), as well as *Gymnodinium catenatum*, *Pyrodinium bahamense* var. *copressum*, and by some cyanobacteria in freshwater ecosystems (Hallegraeff, 2003). Blooms of toxic and noxious dinoflagellates are often referred to as 'red tides' because of the red discoloration of water created by the accessory pigments of the cells. However, toxic levels of saxitoxins can be attained at dinoflagellate abundances that do not significantly discolor the water because of the exceptionally high potency of saxitoxins (Burkholder et al., 2006). This situation exists for *Alexandrium* in that it does not typically reach 'bloom' abundances on the U.S. west coast, and constant toxin monitoring is necessary to identify toxic conditions (Langlois, 2007; IOC HAB Programme, 2008; Jester et al., 2009a).

Alexandrium species and measurable saxitoxin concentrations are common along the U.S. west coast (Table 1), although concentrations reported for this toxin typically have not been as high as noted along the U.S. east coast (Table 2). Thus, few west coast studies have contributed to our understanding the dynamics of *Alexandrium* abundances and saxitoxin production while ongoing research in the Gulf of Maine represents the most comprehensive regional study of this dinoflagellate (Anderson et al., 2005; McGillicuddy et al., 2005). Combined field observations, laboratory studies and modeling efforts have led to a scenario for toxic events along the northeastern coast of the U.S. that involve an interplay between river runoff, resuspension of dinoflagellate cysts from coastal sediments, favorable offshore growth conditions, and winds that generate onshore flow into coastal shellfish areas. A monitoring study of PSP in Puget Sound (WA) from 1993 to 2007 underscores that the timing and location of PSP outbreaks and high *Alexandrium* abundances are highly variable and not easily predicted from local or large-scale climate data (Moore et al., 2009). However, the study points out that periods of warm air and low stream flow may favor saxitoxin accumulation in sentinel mussels (Moore et al., 2009).

2.3. *Brevetoxins*

2.3.1. *Toxin description and activity*

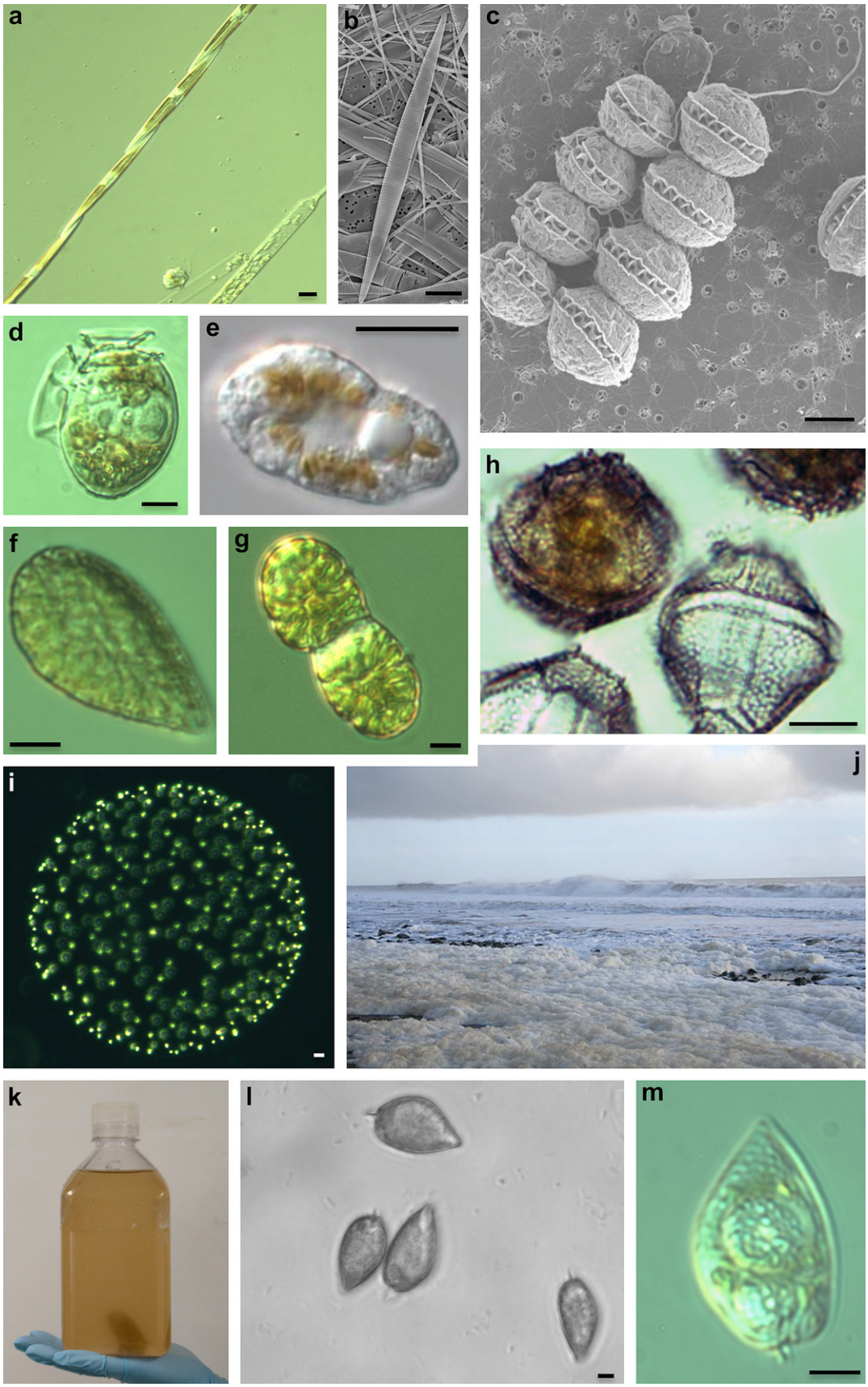
Brevetoxins are polyether, non-polar compounds that depolarize cell membranes by opening voltage gate sodium ion channels and induce enhanced inward flux of ions into cells (Lin et al., 1981; Baden, 1983, 1989; Purkerson et al., 1999). Brevetoxins exist as two structural types and multiple analogs possessing various levels of potency (Baden, 1989; Cembella, 2003; Kirkpatrick et al., 2004). The types differ in their ladder-frame polycyclic ether structural backbones and are designated type A and type B (Fig. 2). The brevetoxin derivatives found in the marine environment (PbTx-2, PbTx-3 and PbTx-9; Table 3) are produced most commonly by dinoflagellate and raphidophyte algae and are of the structural type B (Baden, 1989; Baden et al., 2005).

Brevetoxins bind to site 5 of the voltage-sensitive sodium channel in neurons, causing these channels to remain open and fire repeatedly (Catterall, 1992; Cestele and Catterall, 2000). Brevetoxin poisoning in humans is referred to as neurotoxic shellfish poisoning (NSP), and includes gastrointestinal symptoms of nausea, diarrhea and abdominal pain, neurologic symptoms of paresthesia, and respiratory irritation and/or failure (Kirkpatrick et al., 2004).

The effects of brevetoxins on human health are well documented along the western coast of Florida where severe, nearly annual red tides caused by the dinoflagellate *Karenia brevis* release large amounts of brevetoxins into the air when the fragile cells are broken in breaking waves at the water's edge (Kusek et al., 1999; Kirkpatrick et al., 2004). The aerosolized toxins constitute a significant health risk when they are inhaled and, as a result, *K. brevis* blooms are one of the most intensively studied and best-understood regional HABs. Red tides caused by *K. brevis* have been implicated in marine mammal fatalities, fish kills and human illnesses. Brevetoxins have not been reported from the U.S. west coast, and therefore no known human fatalities or health issues have yet been attributed to brevetoxins from that region.

2.3.2. *Producers*

Several dinoflagellate species and a few raphidophyte species produce a suite of brevetoxins (Baden, 1989). *K. brevis*, the most notorious brevetoxin producer within the Gulf of Mexico, has not been observed on the west coast of the U.S., but several species of raphidophytes that are potential brevetoxin producers have been documented (Tables 1 and 2). *Heterosigma akashiwo* (Fig. 3e), *Chattonella marina* (Fig. 3f) and *Fibrocapsa japonica* have been isolated and cultured from coastal waters off southern California. In general there are few reports of significant blooms of these species on the west coast, although blooms of raphidophytes in San Francisco Bay and Delaware Inland Bays have been observed with cell abundances in excess of 10^8 cells L^{-1} (Herndon et al., 2003; Coyne et al., 2005). In part, this lack of information is a consequence of the fact that raphidophyte species are notoriously difficult to identify using traditional microscopical techniques because they preserve poorly (Hallegraeff and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Hara, 1995; Throndsen, 1997). Recently developed genetic approaches for the identification and quantification of some raphidophytes are beginning to provide much-needed tools for studying the distributions and ecology of these HAB species (Handy et al., 2006; Demir et al., 2008). Despite the difficulties of characterizing these blooms, fish kills have been attributed to raphidophyte blooms on the west coast of the U.S. although these studies have not quantified brevetoxins (Hershberger et al., 1997; Hard et al., 2000).

2.4. *Diarrhetic shellfish toxins*

2.4.1. *Toxin description and activity*

Toxins that cause diarrhetic shellfish poisoning (DSP) include okadaic acid, dinophysistoxins and pectenotoxins (Ramsdell et al., 2005). Okadaic acid is a monocarboxylic acid named for the marine sponge *Halichondria okadai* from which it was first isolated (Tachibana et al., 1981). Okadaic acid can also be found in natural water samples in polar and non-polar esteric forms (Prassopoulou et al., 2009). The first dinophysistoxin described was isolated from the mussel *M. edilus* and was found to be a methyl form of okadaic acid (Murata et al., 1982). Okadaic acid and dinophysistoxins are linear polyethers (Fig. 2) and the mode of action is the inhibition of protein phosphatases (Takai et al., 1987; Bialojan and Takai, 1988; Haystead et al., 1989), enzymes that play a key role in dephosphorylation in many biological processes including cell cycle regulation. The pectenotoxins are lipid soluble and differ structurally from other diarrhetic toxins in that they possess a lactone ring (Fig. 2), and not considered to be protein phosphatase inhibitors, but have a high actin-depolarizing action (Hori et al., 1999). There is speculation that pectenotoxins may not produce diarrhetic effects (Cembella, 2003).

DSP toxins (Table 3) were named for the human symptoms resulting from the ingestion of contaminated shellfish, including inflammation of the intestinal tract, diarrhea, abdominal cramps, vomiting and nausea beginning 30 min to a few hours after ingestion (Hallegraeff, 2003). In addition to the symptoms listed above, okadaic acid is known to be a strong tumor promoter (Suganuma et al., 1988), although the potential health implications of this activity due to the ingestion of contaminated seafood is unknown. There are presently no documented cases of DSP resulting from okadaic acid, dinophysistoxins or pectenotoxins on the U.S. west coast, and thus these toxins are not regularly monitored in the marine environment. DSP toxins have been detected in mussels and water samples from California (Sutherland, 2008), so it is possible that DSP has occurred on the U.S. west coast but has been attributed to other sources of contamination.

2.4.2. *Producers*

Okadaic acid and dinophysistoxins are produced by a few species of the dinoflagellate genus *Prorocentrum* (Cembella, 2003) and most commonly by species of the genus *Dinophysis*. *Dinophysis* species present on the western U.S coast include *D. acuminata*, *D. acuta*, *D. caudata*, *D. fortii*, *D. norvegica*, *D. rotundata*, and *D. tripos* (Table 2). *D. acuminata* and *D. fortii* have been documented in Californian waters for many years (Bigelow and Leslie, 1930). *D. acuminata* produces okadaic acid (Yasumoto et al., 1985), *D. fortii* produces okadaic acid, dinophysistoxins and pectenotoxins (Yasumoto et al., 1980; Murata et al., 1982) while *D. rotundata* and *D. tripos* produce dinophysistoxin-1 (Lee et al., 1989).

Dinophysis species are technically not phytoplankton, but heterotrophic protists that retain chloroplasts acquired from their prey. *Dinophysis* acquires its chloroplasts by preying on ciliates, which in turn prey on cryptophyte algae. This complex trophic relationship has made the culture of these species unsuccessful until recently (Park et al., 2006), and therefore no information on the environmental conditions that influence toxin production by these species presently exists. However, genus members are easily distinguished by light microscopy because they possess a pronounced 'keel' and other unique morphological aspects (Fig. 3d). These species are often encountered in plankton samples. Abundances of 10^3 cells L^{-1} are commonly encountered (Nishitani et al., 2005), and occasionally they may reach abundances in excess of 10^5 cells L^{-1} (Carpenter et al., 1995).

2.5. *Yessotoxin*

2.5.1. *Toxin Description*

Yessotoxin (Fig. 2; Table 3) is a chiral molecule of high polarity due to the presence of two sulfate groups. The molecule consists of fused polyether rings organized into a ladder shaped skeleton (Yasumoto and Murata, 1993; Wright and Cembella, 1998), a structure similar to other ladder-like polyether toxins such as the ciguatera toxin complex (ciguatoxins and maitotoxin), gambieric acids and brevetoxins (Yasumoto and Murata, 1993; Wright and Cembella, 1998). There are nearly 100 analogs of yessotoxin that have been identified to date (Satake et al., 1997, 1999; Ciminiello et al., 1998, 2000, 2001; Daiguji et al., 1998; Miles et al., 2004, 2005a,b, 2006; Paz et al., 2006).

The yessotoxin class was named for the species of scallop, *Patinopecten yessoensis*, in which it was initially detected (Murata et al., 1987). Yessotoxin was originally classified in the DSP-toxin class because it was detected with other DSP toxins, but it appears that it does not induce

Fig. 3 – Major algal toxin producers occurring along the U.S. west coast. (a) *Pseudo-nitzschia australis*, a producer of domoic acid; (b) Scanning electron micrograph of *Pseudo-nitzschia australis*; (c) *Alexandrium catenella*, a producer of saxitoxin; (d) *Dinophysis* sp., a producer of okadaic acid; (e) *Heterosigma akashiwo*, a producer of brevetoxins; (f) *Chattonella marina*, a producer of brevetoxins; (g) *Cochlodinium* sp.; (h) *Lingulodinium polyedrum*, a producer of yessotoxin; (i) *Phaeocystis globosa* colony; (j) foam produced by the prymnesiophyte, *Phaeocystis* accumulating along the shore; (k) Unconcentrated seawater from King Harbor, City of Redondo Beach, with significant discoloration due to an algal bloom; (l) *Prorocentrum* sp., the dominant organism in (k); and (m) higher magnification of *Prorocentrum* sp. from (l). Scale bars = 10 μm. Photo (b) courtesy of Peter Miller, (c) courtesy of Carmelo Tomas, (j) courtesy of Cindi Heslin.

Table 3 – Summary of toxins that can be present in Southern California waters. MW: molecular weight.

Toxin	Properties	Formula	MW	Mode of action	References
Domoic acid (DA)	Hydrosoluble At pH 7: DA^{3-}	$C_{15}H_{21}NO_6$	311.14	Binds to glutamate receptors in the brain disrupting normal neurochemical transmission	Wright et al. (1990), Quilliam (2003)
Saxitoxins (STXs)	Hydrosoluble pH $\leq$7: Stable	$C_{10}H_{17}N_7O_4$	299.3	Bind to site 1 of voltage-sensitive sodium channels and block sodium conductance; bind to calcium and potassium channels	Wong et al. (1971), Wang et al. (2003), Su et al. (2004), Catterall (1992), Cestele and Catterall (2000)
Brevetoxins (PbTxs)	Liposoluble			Bind to site 5 of voltage-sensitive sodium channels, shifting activation to more negative membrane potentials and block channel activation	Lin et al. (1981), Baden (1983, 1989), Purkerson et al. (1999)
Brevetoxin 2 (PbTx 2)		$C_{50}H_{70}O_{14}$	895.1		
Brevetoxin 3 (PbTx 3)		$C_{50}H_{72}O_{15}$	897.1		
Brevetoxin 9 (PbTx 9)		$C_{50}H_{74}O_{14}$	899.1		
Diarrhetic shellfish toxins					
Okadaic acid (OA)		$C_{44}H_{68}O_{13}$	805	Inhibits protein phosphatases, inhibits dephosphorylation of proteins	Tachibana et al. (1981), Murata et al. (1982), Yasumoto et al. (1985), Takai et al. (1987), Bialojan and Takai (1988), Haystead et al. (1989), Hori et al. (1999)
Dinophysistoxins (DTXs)					
Pectenotoxins (PTXs)	Liposoluble			High actin-depolarizaing action	
Yessotoxins (YTXs)	Hydrosoluble	$C_{55}H_{80}O_{21}S_2Na_2$	1187.3	Activation of phosphodiesterase in the presence of external Ca^{2+}; Disruption of the E-cadherin–catenin system in epithelial cells and potentially disrupting its tumour suppressive functions	Murata et al. (1987), Takahashi et al. (1996), Alfonso et al. (2003), Ronzitti et al. (2004)

diarrhetic effects (Ogino et al., 1997). Accordingly, the regulation of the European Commission on marine biotoxins now considers yessotoxins separately from DSP toxins (European Commission, 2002). The great number of yessotoxin analogs complicates toxicity studies, and may explain the sometimes contradictory reports of its mode of action. Studies have shown that lysosomes, the immune system and the thymus (with tumorigenic implications) are the biological targets of yessotoxin (Franchini et al., 2004; Malagoli et al., 2006), while other reports have indicated cardiotoxic effects (Terao et al., 1990; Ogino et al., 1997; Aune et al., 2002). The cardiotoxicity of yessotoxin might be attributed to phosphodiesterase activation in the presence of external calcium ions (Alfonso et al., 2003). Unlike the other marine toxins mentioned above, there have been no reported human health issues or marine mammal deaths associated with yessotoxins.

2.5.2. Producers

There are three known yessotoxin-producing dinoflagellates, *Protoceratium reticulatum*, *Lingulodinium polyedrum*, and *Gonyaulax spinifera*, and they have all been observed in coastal waters off the western U.S. (Table 1). According to phylogenetic analyses of available rRNA gene sequences, the capacity for yessotoxin production appears to be restricted to the order Gonyaulacales (Howard et al., 2009). However, toxin production among strains within each species appears to be highly variable.

Yessotoxin has been detected in *L. polyedrum* isolates cultured from around the globe (Tubaro et al., 1998; Draisci et al., 1999a; Strom et al., 2003; Paz et al., 2004), including isolates from Californian coastal waters (Armstrong and Kudela, 2006). The reported cellular concentrations in the latter cells ranged from below detection to 1.5 pg cell^{-1}, indicating that *L. polyedrum* is significantly less toxic than *P. reticulatum* or *G. spinifera*. Yessotoxin has been recorded in blue mussels at low concentrations along the U.S. west coast (Table 2) during red tides caused by *L. polyedrum*, as well as during non-bloom conditions, but yessotoxin production by this dinoflagellate appears to be less than toxin levels produced by isolates from other geographical regions (see Table 2 in Howard et al., 2008). Expansive and dense blooms of *L. polyedrum* (Fig. 3 h) have been reported in California since 1901, but there have only been anecdotal reports of health problems associated with the red tides caused by *L. polyedrum* (Kudela and Cochlan, 2000).

Yessotoxin production by isolates of *P. reticulatum* has been confirmed (Satake et al., 1997; Boni et al., 2002; Miles et al., 2002; Riobo et al., 2002; Stobo et al., 2003; Samdal et al., 2004; Eiki et al., 2005), but concentrations ranging from below detection to 79 pg cell^{-1} have been reported for isolates from Washington, California and Florida (Paz et al., 2004, 2007). Isolates of *G. spinifera* appear to be the most prolific yessotoxin producers. Concentrations in New Zealand isolates ranged from below detection to 200 pg cell^{-1} (Rhodes et al., 2006), more than 20-fold higher than the per-cell toxicity of *P. reticulatum* and 600-fold higher than *L. polyedrum*. *G. spinifera* does not generally bloom in high densities on the U.S. west coast, but it has been frequently observed at low abundances (Howard et al., 2008; M. Silver, pers. comm.) and has reached bloom concentrations in Tomales Bay, north of San Francisco (G. Langlois pers. comm.). Yessotoxins are monitored in New Zealand, Europe and Japan but they are not routinely measured on the U.S. west coast. Howard et al. (2008) was the first study to confirm yessotoxins in California and Washington coastal waters, albeit at very low concentrations.

2.6. Toxin detection and quantification

A wide variety of methodologies and technologies exist to detect, characterize and quantify the major toxins produced by microalgae. These approaches can be broadly divided into those used to characterize biological activity (toxicity assays) and those used to identify specific chemical structure(s) (immunological, various analytical techniques). Because of the highly variable approaches employed, and in most cases the highly diverse set of compounds comprising a toxin class, the methods provide somewhat different estimates of absolute toxin concentrations or presumed toxicity. It is also noteworthy that the majority of the protocols used to measure algal toxins have focused on the analysis of tissue samples that may be a source of human contamination (e.g. shellfish or finfish tissue) or plankton material filtered from seawater samples. Relatively few studies have examined the concentrations of toxins dissolved in seawater (Table 2). For this reason, limits of detection for toxins in seawater are poorly known for most approaches. However, based on analytical approaches presently available, a practical limit of detection for a few of the major concerns (domoic acid and saxitoxins) is in the range of 0.1 μg per liter of seawater for immunological approaches, but a detection limit of approximately 0.01 μg per liter for dissolved domoic acid in seawater using high-performance liquid chromatography has been reported (Pocklington et al., 1990). Detection limits for toxins in particulate material in the water can be significantly lower because particles can be concentrated by filtration prior to extraction. Knowledge of the concentrations of dissolved toxins would be preferable from the perspective of reverse osmosis desalination operations because dissolved toxins (rather than cell-bound toxins) would most likely impact these membranes.

Domoic acid has been detected and quantified in seawater, plankton, shellfish extract and homogenate, as well as sea bird and mammalian body fluid (e.g. blood, urine, amniotic fluid, cerebral spinal fluid). Analytical approaches for these measurements include commercially available immunological techniques (enzyme-linked immunosorbent assay; ELISA) (Garthwaite et al., 1998, 2001), high pressure liquid chromatography (HPLC) with ultraviolet (UV) diode array detection (DAD) (Quilliam et al., 1989; Quilliam, 2003), receptor binding assay (RBA) (Van Dolah et al., 1994), mouse bioassay (MBA) or liquid chromatography–mass spectrometry (LC–MS). The results obtained by these various approaches are not yet completely compatible or comparable, and therefore comparisons across studies using different analytical methods can be problematic. In general, the choice of an approach is a compromise between cost of analysis (or access to costly equipment), sample throughput, sensitivity and analytical goal (e.g. thorough chemical characterization versus overall

toxicity, or human health risk versus scientific understanding of bloom dynamics).

Saxitoxins can be rapidly detected and quantified with commercial ELISA kits, but the high specificity of these tests precludes the recognition of certain members of the saxitoxin family, especially the neo-saxitoxin (Garthwaite et al., 2001). Saxitoxin detection and quantification is often accomplished by HPLC, RBA, LC–MS, and MBA. Regulatory programs for seafood consumption are still based on 'lethal mouse dosage'. Similarly, detection and quantification of brevetoxin and its derivatives in seawater, shellfish, and mammalian body fluids can be accomplished using commercially available ELISA kits (Naar et al., 2002), by HPLC, RBA (Van Dolah et al., 1994), LC–MS and MBA. A comparative study also quantified brevetoxins by radioimmunoassay (RIA) and a neuroblastoma (N2A) cytotoxicity assay (Twiner et al., 2007).

Quantification of the DSP toxin suite can be underestimated by ELISA because commercial ELISA assays are usually optimized to detect okadaic acid and not the dinophysistoxins (Garthwaite et al., 2001). HPLC with fluorimetric detection (HPLC-FLD) has been commonly used to detect and quantify okadaic acid, its polar and non-polar esters, as well as dinophysistoxins (Lee et al., 1987). The MBA method has also been routinely used for the detection of DSP toxins.

The detection and quantification of yessotoxin is problematic because of the extensive suite of derivatives that may exist. HPLC-FLD analysis (Yasumoto and Takizawa, 1997), MBA, LC-MS (Draisci et al., 1999b; Paz et al., 2006, 2008) and ELISA (Samdal et al., 2004, 2005) have been employed.

2.7. *Other potentially toxic, noxious and nuisance organisms*

Reports of newly occurring HAB species, or recognition of extant issues that have gone previously undocumented, are increasing our awareness of other potentially harmful bloom-forming algae along the U.S. west coast. For example, blooms of an emerging potentially toxic organism, *Cochlodinium* sp. (Fig. 3g) off central California have recently been reported (Curtiss et al., 2008; Iwataki et al., 2008; Kudela et al., 2008b). This species is difficult to identify using light microscopy, and therefore researchers have begun to use gene sequences to provide accurate identification (Iwataki et al., 2007, 2008; Matsuoka et al., 2008). While this organism has only recently reached sufficient abundances to discolor Californian waters, there has already been one reported abalone loss in central California that appears to be associated with a bloom of *Cochlodinium* (R. Kudela pers. comm).

Species of the dinoflagellate *Prorocentrum* (Fig. 3k–m) occasionally bloom in Californian coastal waters where they can attain very high abundances periodically, causing discolorations of the water and nuisance accumulations of algae (Holmes et al., 1967; Shipe et al., 2008). The *Prorocentrum* species known to occur in Californian waters have not yet demonstrated DSP toxin production, but species from elsewhere in the world are known to produce theses toxins (Table 1). Similarly, massive blooms of the dinoflagellate *Akashiwo sanguinea* are common in coastal waters of southern and central California and have recently been the cause of seabird mortality due to surfactant-like proteins (Jessup et al., 2009). Although they may not be overtly toxic, these blooms can cause animal mortalities, deplete oxygen, and result in an increased organic and biomass loading to a seawater desalination facility.

The prymnesiophyte *Phaeocystis globosa* infrequently attains high abundances off the Californian coast (Armonies, 1989). This species produces single cells that are <10 μm in size, but it also forms fluid-filled colonies several millimeters in diameter in which individual cells are embedded in the polysaccharide skin of the colony (Fig. 3i). Single cells of

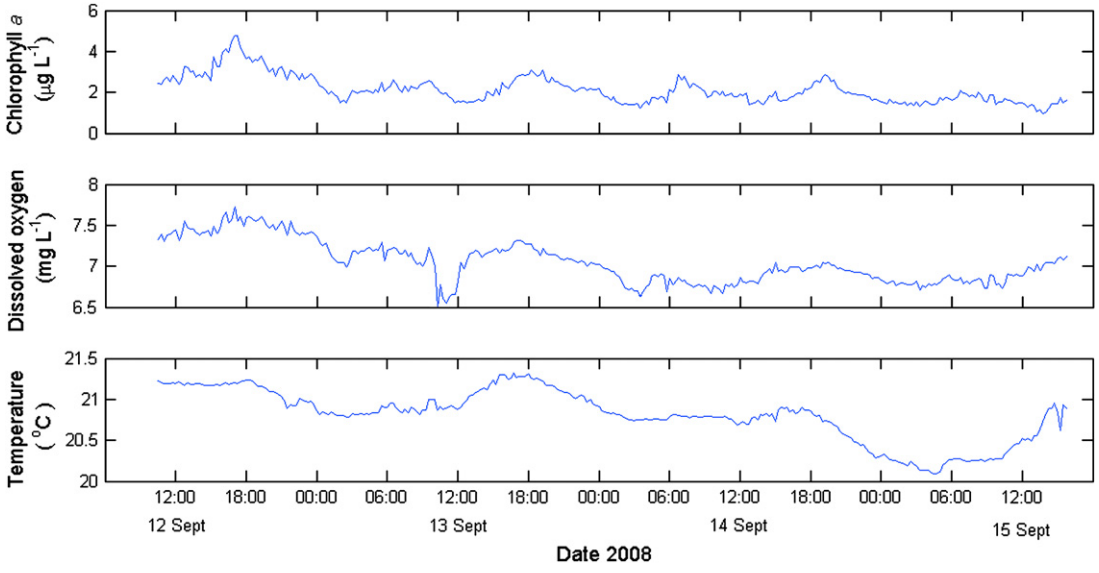

Fig. 4 – Time series of chlorophyll *a* fluorescence, temperature and dissolved oxygen in King Harbor, City of Redondo Beach, CA. Note the short-term temporal fluctuations in these parameters that are a result of tide, wind and biological interactions. These measurements were collected using autonomously recording sensors that provide high resolution observations of chemical and physical properties that might indicate an algal bloom, or environmental factors that might stimulate a bloom.

Phaeocystis are consumed by many zooplankton species but the colonies are typically a poor food source. Selective feeding on single cells appears to favor colony formation and the accumulation of colonies in the water column (Netjstgaard et al., 2007). When released via colony destruction or algal life cycle events, the colony matrix material is easily worked into a 'sea foam' that can form layers many centimeters (even meters) thick at the ocean surface or along the coastline over fairly extensive regions (Fig. 3j).

3. Spatial and temporal patterns of harmful algae

A fundamental aspect of the biology of harmful algal blooms, and of vital importance for desalination operations, is the tendency for rapid and dramatic changes in the spatial and temporal distributions of these species. These changes occur rapidly across a wide range of scales, and pose challenges for documenting and predicting these distributions. Numerous approaches and instruments have been developed to characterize the dynamics of phytoplankton communities. These approaches presently have significant limitations on their abilities to identify species composition of a bloom, but they provide crucial information on the emergence and longevity of bloom events as well as their vertical and geographical extent. This information can help seawater desalination facilities adjust operations to ensure reliable production for the duration of the bloom.

3.1. *Temporal variability*

Significant temporal variations in the abundances of phytoplankton take place on time scales ranging from hours to decades. Short-term temporal variability (hours to a few weeks) can be a consequence of rapid population growth or consumption by herbivores, sinking of senescent populations, diel vertical migration, tidal movement, and aggregation or dispersal by physical processes such as water mass convergences or divergences.

Diel vertical migration of several dinoflagellates has been attributed to geotaxis, phototaxis and nutrient status (Eppley et al., 1968; Blasco, 1978; Cullen and Horrigan, 1981; Levandowsky and Kaneta, 1987). Classic responses involve nighttime sinking out of surface waters to deeper water where nutrient concentrations are greater, and rising into surface waters for photosynthesis during daytime. Shifts in nutrient cell quotas that accompany these migrations may have significant implications for toxin production because cell toxicity can be related to nutrient status of the cells (Anderson et al., 1990; Flynn et al., 1994; John and Flynn, 2000; Flynn, 2002).

Diel-to-weekly variations in phytoplankton abundance can be characterized using self-contained or wirelessly networked sensor packages. Data collected in King Harbor of the City of Redondo Beach, CA (Fig. 4) demonstrate the efficacy of these instruments for providing high temporal resolution of chlorophyll *a* fluorescence (which approximates phytoplankton biomass) and pertinent environmental factors (e.g. dissolved oxygen and temperature) that provide insight into the factors controlling the pattern. These data reveal a 4-fold variation in phytoplankton standing stock over a two-day period. In addition, changes in water quality criteria were easily and rapidly identified (e.g. decrease in dissolved oxygen concentration near noontime on September 13). Daily variations in the latter parameter can be extreme at night during algal blooms. High resolution, short-term monitoring approaches allow rapid detection of sentinel parameters, and in turn provide information for the development of predictive models of HAB events.

Chlorophyll *a* fluorescence sensors provide valuable information on the short-term temporal patterns of total algal biomass, but these instruments cannot identify noxious algal species within a phytoplankton assemblage. More sophisticated instruments now coming online offer that possibility. For example, the Environmental Sample Processor (http://www.mbari.org/ESP) is an *in situ* instrument capable of performing real-time identifications of HAB and other microbial species, as well as toxin analyses such as domoic acid using on-board molecular analyses (Greenfield et al., 2006). Similarly, handheld devices now exist for the detection of some HAB species and toxins in the field (Casper et al., 2007). These rapid and highly specific analyses are becoming valuable tools for quick determinations of toxin presence resulting from algal blooms. These more costly instruments can be used to improve the information available on bloom composition once the cheaper and more readily available sensors identify an emerging bloom event.

Seasonal variability of HAB species and their toxins along the Californian coast can be gleaned from the records of the Marine Biotoxin Monitoring Program (MBMP) of the California Department of Public Health (CDPH), a program started after a major domoic acid outbreak in the fall of 1991. At present, the annual effort involves the analysis of approximately 300 shellfish samples for domoic acid and >1000 samples for PSP toxins from all fifteen Californian coastal counties (CDPH, 2007). Shellfish toxin information provides a reasonable representation of toxins in the upper water column over seasonal and annual scales such as demonstrated in studies on toxic dinoflagellates (Montojo et al., 2006; Jester et al., 2009a; Moore et al., 2009). Detailed information on the sampling effort is provided in the MBMP annual reports (Langlois, 2007).

Distributions of domoic acid and PSP toxins along the Californian coast during the period 2002–2007 based on the MBMP data reveal both seasonal and geographical trends (Figs. 5 and 6). Monthly averages for the 6-year period (histograms) and maximal concentrations (triangles and lines) showed detectable concentrations of PSP toxins and domoic acid in shellfish in nearly all months for all Californian coastal counties (Fig. 5). Domoic acid concentrations showed pronounced seasonality, with very high peaks during spring and much smaller peaks during fall. This temporal trend is concordant with previous observations along the southern Californian coast (Lange et al., 1994; Walz et al., 1994; Schnetzer et al., 2007; Shipe et al., 2008). Fall domoic acid peaks correspond to minor blooms of toxic *Pseudo-nitzschia* occasionally noted on the west coast of the U.S. (Bolin and Abbott, 1960; Buck et al., 1992; Walz et al., 1994; Trainer et al., 2002; Schnetzer et al., 2007).

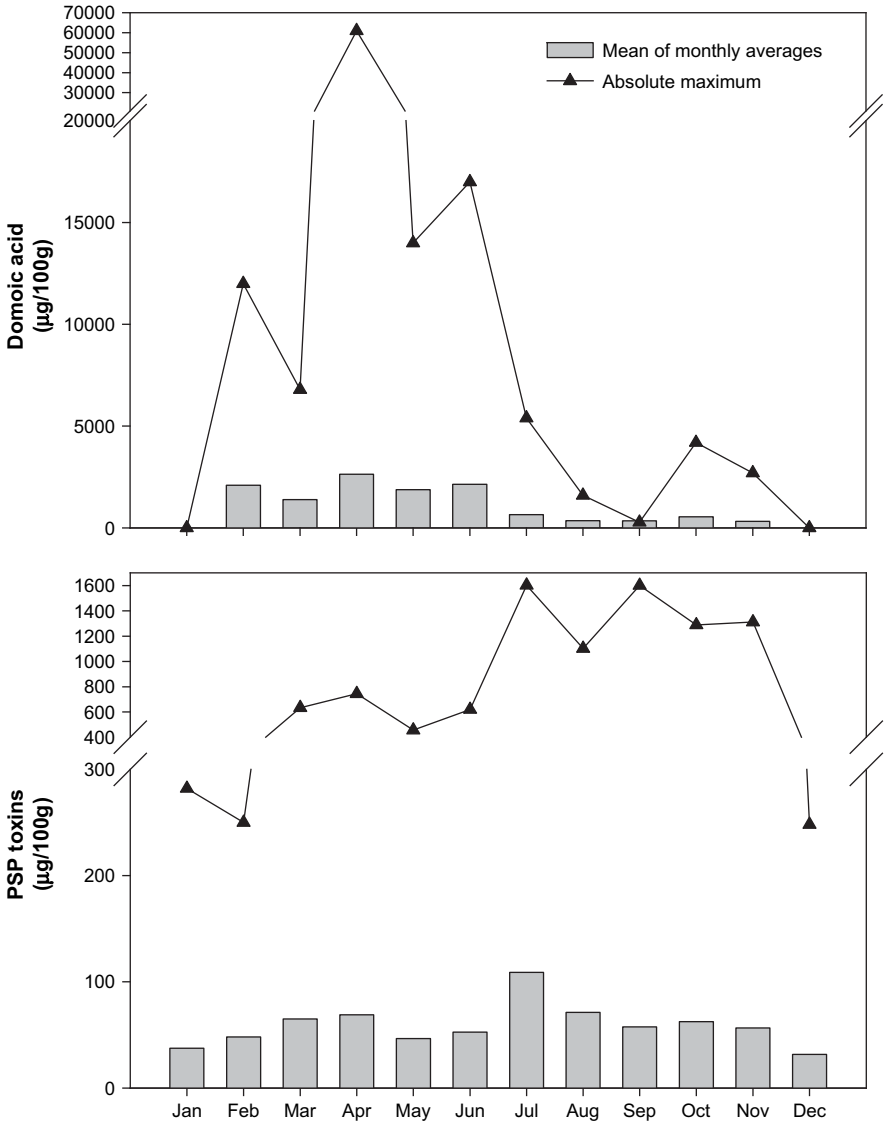

Fig. 5 – Seasonal variability of domoic acid (upper panel) and PSP toxin (lower panel) concentrations along the Californian coast. Monthly averages of data collected from 2002 and 2007 summarized for fifteen Californian coastal counties are shown as histograms. Also shown are the maximal values recorded during each month over the entire study period (triangles and solid lines). Toxin concentrations were derived from shellfish tissue. Data from CDPH (Langlois, 2007).

Seasonal peaks in domoic acid along the U.S. west coast differ along a latitudinal gradient. Highest domoic acid concentrations in Washington have been observed during the fall (Trainer, 2002; Office of Shellfish and Water Protection, 2008; Moore et al., 2009) compared to spring blooms that are common in southern California (Schnetzer et al., 2007). This north-south trend in seasonality is also evident on a smaller latitudinal scale. Blooms along the Californian coast have been more frequently observed later in the year in northern counties (Humboldt versus Santa Barbara counties in Fig. 6). The time lag between bloom periods observed from south to north may be related to the timing of the California Current System (CCS) upwelling maximum, which brings nutrients into surface waters and promotes phytoplankton growth. The CCS upwelling occurs in early spring in southern California, in June off Washington, and throughout summer in northern California and Oregon (Reid et al., 1956; Landry, 1989).

Monthly averages as well as maximal PSP toxin concentrations showed less pronounced seasonal variability than domoic acid during the period 2002–2007, although the highest concentrations were recorded from July to September (Fig. 5). This seasonal pattern of maximal PSP is in agreement with the last 25 years of monitoring results (Langlois, 2007). It is also consistent with the 1927–1989 observations on the Californian coast indicating that most significant concentrations of the toxin take place between May and October (Price et al., 1991). The highest PSP toxin concentrations in shellfish have also been observed in the summer and the fall periods off the Washington and Oregon coasts (Trainer et al., 2002; Determan, 2003).

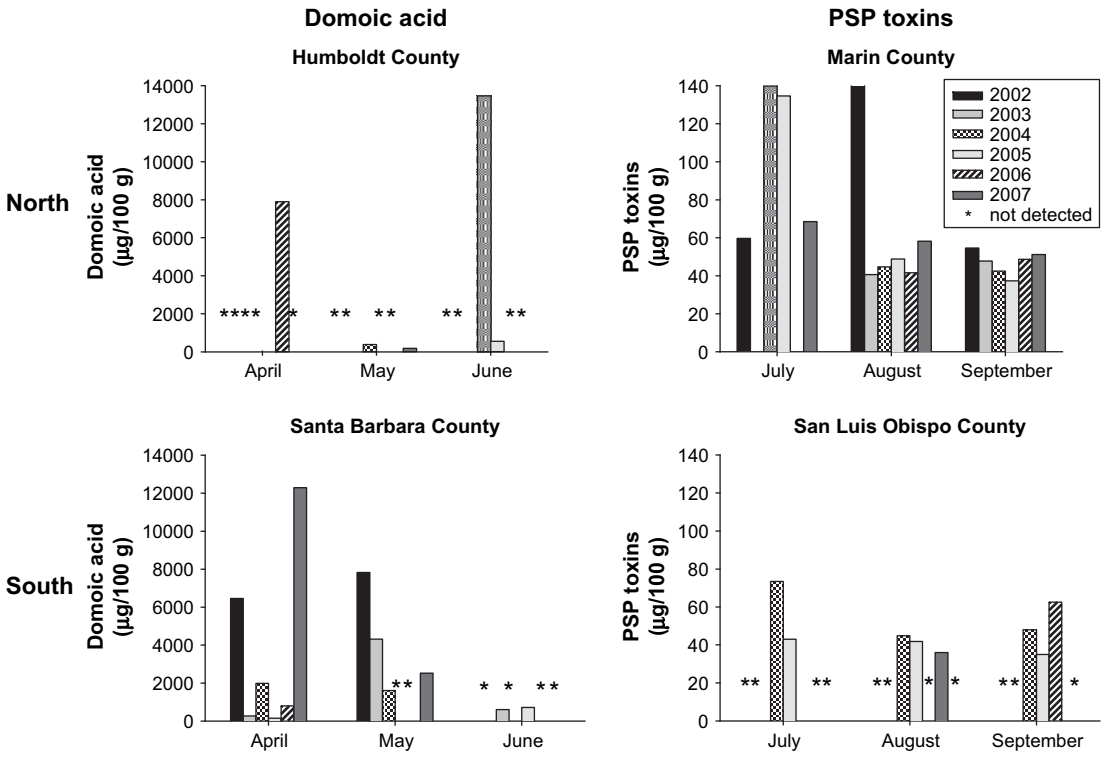

Fig. 6 – Interannual variation in toxin concentration in four Californian coastal counties from 2002 to 2007 during three months exhibiting high concentrations of domoic acid (April, May, June) and saxitoxin (July, August, September). Humboldt and Marin counties are located north of Santa Barbara and San Luis Obispo counties. Toxin concentrations were measured from shellfish tissues. Data from CDPH (Langlois, 2007).

Diel to seasonal temporal patterns in ASP and PSP outbreaks off the U.S. west coast are augmented by large interannual variability in the intensity and frequency of HABs. Interannual variations in HABs presumably are related, at least in part, to changes in atmospheric and hydrographic features modulated by the 3–7 year cycles of El Niño-Southern Oscillations (ENSOs) (Price et al., 1991; Horner et al., 1997), but the exact relationship between HABs and these climatic events is not clear because there are still relatively few observations spanning these temporal regimes. Warming off California during El Niño episodes reduces seasonal upwelling, enhances physical stratification in the CCS and lowers the nutricline in the water column (the depth at which nutrient concentrations increase rapidly). It is clear that these changes in water stability and nutrient availability have significant impacts on plankton productivity and community structure, but the specific responses of the phytoplankton communities vis-à-vis HAB events are not yet predictable (Barber and Chavez, 1983).

ASP outbreaks occurred during El Niño episodes of 1991 and 1997–98 along the coasts of Oregon, Washington and California (Table 2; Moore et al., 2008), but the specific factors contributing to toxic *Pseudo-nitzschia* blooms during these events could not be clearly identified (Horner and Postel, 1993; Trainer et al., 2000). Interannual variability in ASP and PSP concentrations in coastal waters along the Californian coast was evident and substantial in the MBMP dataset during the period 2002–2007 (Fig. 6). ASP and PSP outbreaks were frequent, but they varied in intensity and the timing of peak concentrations between the years within a single geographical location, and between southern and northern Californian counties in the same year and season. Notably, the magnitude of this variability is on the same order of magnitude as the variability observed on short-term (daily) or seasonal time scales.

Little is known regarding the longer time scale fluctuations in HABs along the U.S. west coast. Multi-decadal fluctuations in ocean temperature are known to provoke shifts in the biological regime (Chavez et al., 2003), and it is anticipated that climatic shifts would affect the timing, intensity or frequency of phytoplankton blooms. Long-term regime shifts may affect the occurrence of blooms of *L. polyedrum*, a producer of yessotoxin, along the coast of southern California (Tables 1 and 2). The Pacific Ocean was cooler in the years preceding 1976, and red tides dominated by *L. polyedrum* commonly developed along the Southern California Bight during fall (Gregorio and Pieper, 2000). During a recent warm regime (1976-mid 1990s), red tides occurred during winter and spring and persisted until summer in the region of the Los Angeles River mouth (Gregorio and Pieper, 2000). Recent massive blooms of *L. polyedrum* during fall may indicate a return to the pre-1976 conditions (Moorthi et al., 2006).

The generality surmised from data depicting short- to long-term temporal variability in phytotoxin dynamics is that variability can be high at all scales. Given our present state of understanding regarding the specific combination of forcing factors that give rise to this high variability, it is difficult to

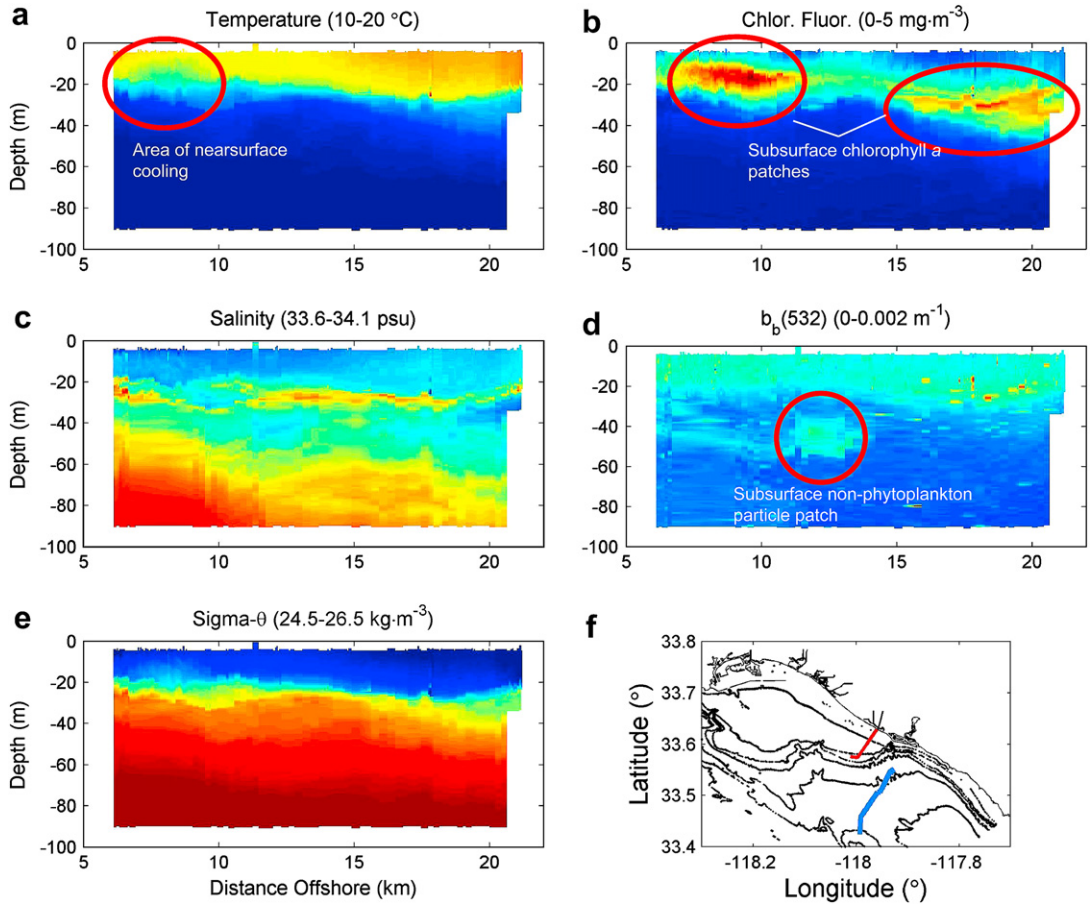

Fig. 7 – Two-dimensional presentations of chemical and physical data collected by an autonomous vehicle (Webb Slocum glider, Teledyne Webb Research, East Falmouth, MA) along a nearshore–offshore transect at Newport Beach, CA shown in (f) (indicated on map by the blue line). Contour plots are shown for (a) temperature, (b) chlorophyll *a* fluorescence, (c) salinity, (d) backscattered light and (e) water density. The Webb Slocum glider is an autonomous vehicle commonly employed in coastal ecosystems. This buoyancy-driven underwater vehicle generates horizontal motion by ascending and descending with pitched wings (Schofield et al., 2007). A rudder directs heading while buoyancy is controlled by pumping seawater into and out of the nose of the vehicle. This long-lived, low-power glider achieves horizontal velocities of approximately 25–30 cm s^{-1} with vertical velocities of 10–15 cm s^{-1}.

accurately predict the timing and magnitude of toxic blooms. For these reasons, monitoring at multiple temporal scales is necessary to adequately characterize plankton dynamics.

3.2. *Spatial variability*

Spatial variability of HABs is considerable at multiple scales, analogous (and strongly related) to the temporal variability described above. Blooms can be highly localized (10s of meters) or expansive (100s of kilometers), and distributions vertically within the water column are heterogeneous over scales of centimeters to meters. The geographical extent and heterogeneous nature of the U.S. west coast results from differences in local hydrography that are manifested in small- and large-scale differences in spatial patterns of toxic blooms.

Regional-scale variations in HAB distributions are illustrated by MBMP data during 2002–2007 for ASP and PSP concentrations observed in counties along northern and central California (Fig. 6). ASP events (frequency and level of toxicity) were generally lower in northerly Humboldt County during this period relative to Santa Barbara County nearly 1000 km to the south. More recently, high domoic acid concentrations have been observed within the Southern California Bight, presumably indicating a continuing southward movement of the *Pseudo-nitzschia* spring blooms (Langlois, 2007; Schnetzer et al., 2007). On the other hand, Marin County (north of San Francisco) exhibited higher monthly PSP values during most months than San Luis Obispo County located nearly 500 km to the south. This general latitudinal trend in PSP events is consistent with findings that the three southernmost counties (Los Angeles, Orange and San Diego) generally experience low concentrations of PSP relative to northern California (Price et al., 1991; Langlois, 2007). Regional-scale and geographical differences in ASP and PSP events have also been reported along the coastline of Oregon (Trainer et al., 2002).

Small-scale spatial variability (horizontally and vertically) can also be dramatic. HAB events can be highly restricted

geographically (e.g. relegated to a protected embayment). Even within spatially extensive blooms, phytoplankton biomass is often highly discontinuous over very small spatial scales because of differences in local circulation, and wind or wave forcing. Considerable spatial variability in the abundance of *P. australis* within Monterey Bay has been observed, a phenomenon that has been attributed to advective forces (Buck et al., 1992). Discontinuities within a water column, such as thermoclines, haloclines, nutriclines and light absorption often leads to the establishment of subsurface microlayers where phytoplankton biomass can be many-fold elevated relative to algal standing stocks only centimeters above or below (Dekshenieks et al., 2001; Rines et al., 2002).

Characterization of phytoplankton spatial distributions include approaches ranging from shore-based sampling, to the use of oceanographic ships, to remote sensing of large-scale patterns using satellite imagery. Vertical profiling of phytoplankton assemblages can be accomplished using over-the-side, ship-based instrument packages and more recently autonomous vehicles equipped with a variety of sensor packages. The use of autonomous vehicles to provide synoptic measurements of phytoplankton biomass from chlorophyll *a* fluorescence and pertinent chemical/physical parameters is state-of-the-art for obtaining two-dimensional cross-sections or three-dimensional patterns in the water column (Fig. 7).

Autonomous vehicles provide time- and depth-stamped measurements of a variety of parameters that can be optimized for a specific mission. Such an instrument deployed off Newport Beach, CA (blue line in Fig. 7f) during May–June 2008 yielded detailed patterns of chemical/biological parameters that provided information on the extent and vertical distribution of phytoplankton biomass (Fig. 7a–e). Evidence of upwelling in this spring deployment was apparent from the upward-pointing isotherms (temperature) and isopycnals (density) on the shoreward end of the transect (left sides of Fig. 7a, e). This was particularly evident in the temperature plot 5–10 km from shore between the surface and 20 m (Fig. 7a, red circle). Nutrients (not shown) were generally depleted in surface waters, and increased with decreasing temperature. This physico-chemical structure is concordant with a significant subsurface maximum in chlorophyll *a* concentration, indicative of a response of the phytoplankton assemblage to elevated nutrient concentrations at this depth (red circle on left in Fig. 7b), as previously observed (Jones et al., 2002). The cross-sectional picture provided by the autonomous vehicle indicated that the phytoplankton community had a patchy structure on the scale of meters (vertically) and kilometers (horizontally). Two major horizontal patches were observed at 6–12 km and at 15–21 km from shore (red circles in Fig. 7b).

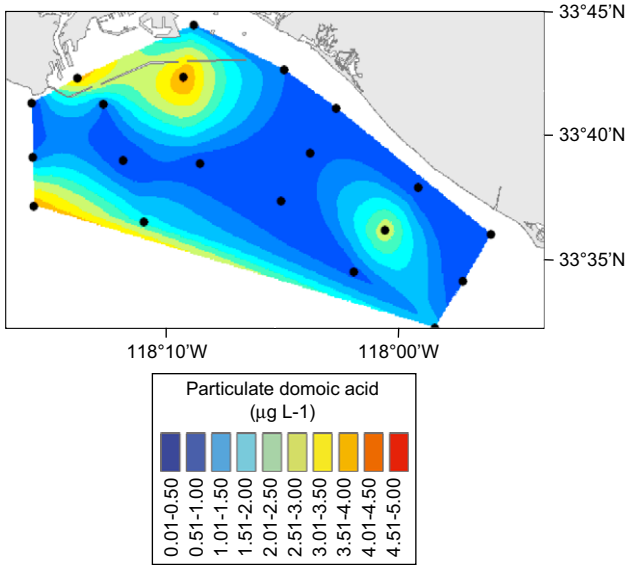

Fig. 8 – Spatial variability in domoic acid concentrations contained in the total particulate material (algal cells, other microbes and detritus) in surface waters within the San Pedro Bay area including the Long Beach-Los Angeles harbor area. Data were collected at 20 sampling stations (locations indicated by filled circles).

Measurements in addition to chlorophyll *a* fluorescence can add information on the observed general patterns such as particle size distributions derived from the optical backscatter spectrum. The backscatter profile obtained at a wavelength of 532 nm indicated a patch of particles that were not of algal nature (Fig. 7d, red circle). The wavelength-dependent slope of the backscatter, which is dependent on the particle size distribution, can also be mapped to indicate the size-class of phytoplankton particles that dominate the chlorophyll *a* maxima. This information is particularly important for a seawater desalination facility, where the incoming particle size distribution is known to impact the source water filterability.

Autonomous vehicles allow nearly synoptic measurements of the spatial distribution of phytoplankton and ancillary parameters. Many of these instruments operate for significant periods of time (weeks) and thereby supply temporal as well as spatial coverage. Knowledge of the temporal evolution and spatial organization of coastal marine systems enables a better understanding of the linkages between physical processes and the biological responses that contribute to the formation of algal blooms. Moreover, data from these instruments can be telemetered to the laboratory in near-real time and used to direct costly efforts such as shipboard sampling, or plan operations for land-based activities and measurements.

Broad-scale, horizontal distributions can also be acquired via shipboard sampling programs (Fig. 8). Shipboard work is time and labor intensive, but onboard sample processing enables more sophisticated analyses than autonomous vehicles are presently capable of providing. Moreover, ships and other manned platforms permit time series studies at a single study site. Shipboard sampling conducted in April 2008 in the Long Beach-Los Angeles harbor area and the adjacent San Pedro Channel demonstrated considerable spatial variability in the distribution of HAB species and their toxins within this relatively small area (approximately 500 km^2). Results indicated a patchy distribution of domoic acid in particulate material (i.e. within phytoplankton cells) collected near the surface with highest concentrations in the vicinity of the harbor breakwater and at several offshore locations (Fig. 8). Intermediate regions exhibited toxin concentrations that were more than an order of magnitude less than these maxima.

3.3. *Environmental driving factors*

Characterizing the factors that lead to the stimulation of harmful algae and the production of toxins by these algae has been an area of very active research for decades. These studies involve field observations to document the spatiotemporal extent of blooms and toxin concentrations in plankton and marine life, and laboratory experiments aimed at understanding the key environmental factors leading to HAB events and toxin production. The overall results gleaned from many years of work group into three basic categories: (1) factors and conditions leading to phytoplankton blooms in general, (2) factors leading specifically to the growth of HAB species, and (3) factors leading to toxin production.

Numerous factors have been implicated as contributors to the observed global expansion of HABs (Smayda, 1990; Hallegraeff, 1993, 2003; Anderson et al., 2002; Glibert et al., 2005b). Phytoplankton blooms occur naturally as a consequence of the vertical mixing of deep, nutrient-rich waters into lighted surface waters. This process occurs seasonally in temperate environments due to winter storm events, and due to coastal upwelling events caused by appropriate regional wind conditions. There is no a priori reason why these 'natural' sources of nutrients cannot lead to HAB events, but the global increase in frequency and severity of HABs implies that human activities may be an underlying reason for this escalation.

Eutrophication of coastal ecosystems is a growing global concern that has clear consequences for blooms of nearshore algal populations (Anderson et al., 2008; Heisler et al., 2008; Howarth, 2008). Nutrient enrichment has been implicated in harmful blooms occurring in some protected bays, but the linkage between nutrient discharges mediated by human activities and many HAB events is still unconfirmed. For example, field studies have shown that coastal upwelling of nitrate-rich waters can be a driving factor leading to toxigenic *Pseudo-nitzschia* blooms along the U.S. west coast (Horner et al., 2000; Scholin et al., 2000; Anderson et al., 2006) but the specific role of river/coastal runoff in domoic acid production is unclear (Scholin et al., 2000; Schnetzer et al., 2007). The importance of nutrient discharge into coastal waters is, of course, dependent on the amount of nutrients available for phytoplankton growth from natural sources but the latter term is poorly defined in most situations. Constructing nutrient budgets for coastal ecosystems is an area ripe for future work. In the meantime, it has been speculated that anthropogenic nutrient sources, such as elevated nutrient concentrations in river discharge, coastal runoff from agricultural land, and sewage discharge may significantly increase the total amounts of nutrients available for the growth of coastal phytoplankton (Scholin et al., 2000; Glibert et al., 2005a,b, 2006; Howard et al., 2007; Kudela et al., 2008a).

There now exists a basic understanding of the general conditions that favor the growth of phytoplankton per se (Allen et al., 2008). Despite this basic understanding, there is still only limited information on the specific conditions that selectively stimulate the growth of harmful algal bloom-forming species of phytoplankton. As a result, mathematical models that attempt to predict HAB events tend to be more correlative than deterministic (i.e., they identify the conditions that *may* promote a HAB, rather than the conditions that *will* promote a bloom). One generality is that rarely can one identify a 'silver bullet', a single parameter or set of circumstances that provide an accurate prediction of the occurrence of a particular HAB species. The environmental circumstances leading to the dominance of a HAB population over all other species of algae in a given locale are composed of a complex set of physical, chemical and biological conditions with poorly known variances, and these conditions appear to be species-specific.

Biological factors contributing to the success or demise of individual HAB taxa include allelopathy among competing phytoplankton, mixotrophy by HAB species, and the deterrence of potential consumers via the production of noxious or toxic compounds (Strom et al., 2003; Burkholder et al., 2008; Buskey, 2008; Flynn, 2008; Smayda, 2008). These biological interactions presuppose a period of stable environmental conditions in order that the scenarios of allelopathy, grazer deterrence or phagotrophic activity of HAB species can play themselves out. This requirement may explain why the formation of a stable water mass appears to play a role in the development of some HAB events (Scholin et al., 2000).

Models predicting the population growth of potentially toxic algae are necessary for understanding bloom dynamics, but these models must also integrate information on toxin induction. Many toxins do not appear to be constitutively produced by algae, but are induced by a variety of specific environmental conditions that are not completely understood. Silica, phosphorus, nitrogen and trace metal limitations, and nutrient or elemental ratios (in addition to or instead of absolute concentrations) have all been implicated in toxin induction (Pan et al., 1996a, 1998; Rue and Bruland, 2001; Fehling et al., 2004; Wells et al., 2005; Granéli and Flynn, 2006; Schnetzer et al., 2007). Again, species-specific differences (and perhaps strain-specific differences) may exist in the factors promoting toxin production. Physical aspects such as temperature and light intensity may stimulate toxin production by some harmful algae (Ono et al., 2000).

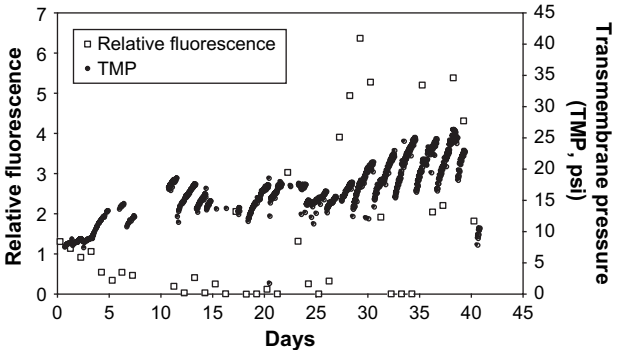

Fig. 9 – Correlation between chlorophyll *a* fluorescence (open squares, in relative fluorescence units) and transmembrane pressure (filled circles) in a pilot-scale reverse osmosis desalination system. Increased loading of phytoplankton biomass resulted in greater transmembrane pressure.

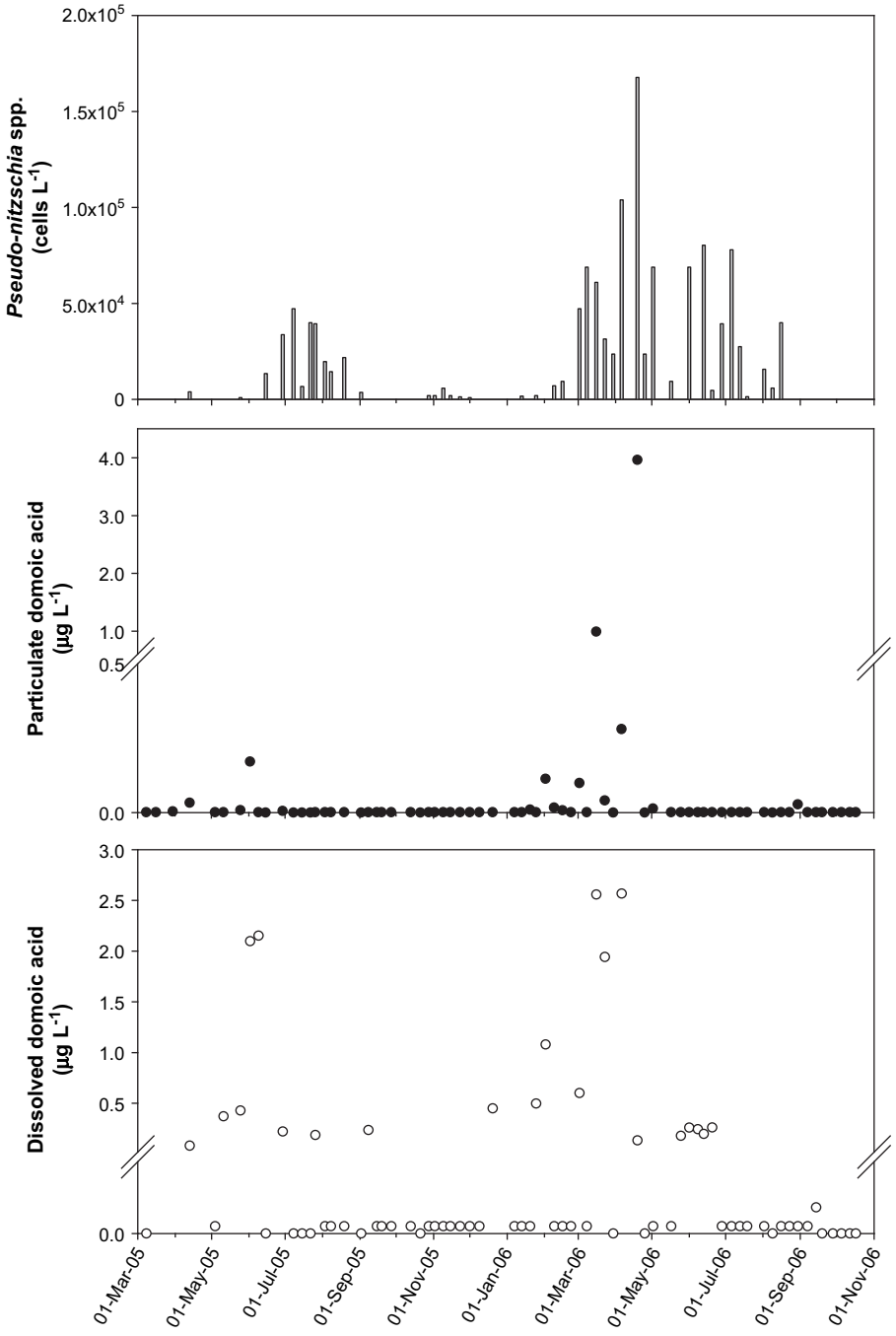

Fig. 10 – Time series of abundances of *Pseudo-nitzschia* spp. cells (top), domoic acid concentrations in particulate material (middle) and dissolved in seawater (bottom) at a coastal monitoring site in El Segundo, CA.

4. Desalination operations and HAB events

A thorough understanding of the specific factors and conditions giving rise to harmful algal blooms and toxin production in coastal waters will require a great deal of additional research before accurate models for predicting these toxic events will be readily available. Until then, appropriate monitoring strategies to detect imminent bloom events and the ability to track the evolution of an active bloom, coupled with an understanding of the potential toxins being produced, the toxin chemistry, and their rejection by seawater reverse osmosis membranes, provide a seawater desalination facility with the best strategy for making operational adjustments to ensure that the treatment plant capacity or product water quality remains unaffected.

4.1. *Concern with harmful algal blooms and their toxin production*

As seawater desalination has continued to become more cost-effective and less energy intensive (Al-Sahlawi, 1999), many

communities are planning or implementing seawater desalination facilities (Al-Sahlawi, 1999; Burbano et al., 2007). The selected pretreatment procedures and the process engineering that determines the ultimate facility design is entirely dependent upon the source water quality (e.g. suspended solids, turbidity, organic material content, algal cell content, etc.) and its variations, particularly for facilities incorporating open intakes (Bonnelye et al., 2004b; Gaid and Treal, 2007). When the seawater desalination process is performed by reverse osmosis membranes, the selection and proper operation of a pretreatment system is paramount to the success of the downstream desalination process (Tenzer et al., 1999; Bonnelye et al., 2004b; Separation Processes Inc., 2005; Gaid and Treal, 2007; Petry et al., 2007).

Algal blooms are known to have significant negative impacts on reverse osmosis desalination facilities. A variety of pretreatment trains have been considered to address the difficult source water quality associated with algal blooms, where the organic and biomass load increase dramatically (Adin and Klein-Banay, 1986; Al Arrayedhy, 1987; Hasan Al-Sheikh, 1997; Watson, 1997; Abdul Azis et al., 2000; Bonnelye et al., 2004a,b; Burbano et al., 2007; Gaid and Treal, 2007; Petry et al., 2007; Peleka and Matis, 2008). Recently, microfiltration and ultrafiltration membrane pretreatment has been identified as a component of a preferred pretreatment train due to the consistent, high quality water produced by membrane filtration, especially when compared to conventional processes (Wilf and Schierach, 2001). However, one significant drawback to implementation of these modern pretreatment technologies is that they are as susceptible, or possibly more so, to significant algal blooms (Bonnelye et al., 2004a).

An early warning system can provide information to a seawater desalination facility so that functional changes can be made to efficiently maintain operations even as source water quality deteriorates. Turbidity sensors offer a rapid measurement of the total amount of suspended particles in the intake water for making these decisions. On *in situ* instruments such as the Slocum glider, backscatter measurements yield this type of information (Fig. 7d). Measurements of chlorophyll *a* fluorescence can augment this surveillance approach by estimating the degree to which algal bimoass contributes to the total load of suspended particles. Responses to deteriorating water quality may include chemical additions, use of additional pretreatment equipment, or additional staff preparations (e.g. maintenance activities, guaranteeing all membranes and filters are clean in preparation for an event) to continuously deliver a high quality feedwater to the reverse osmosis system that will produce the desalinated drinking water.

4.2. *Addressing spatiotemporal variability in HAB abundance and early detection*

As detailed above, it is essential that a desalination facility incorporate a means of rapid algal bloom detection so that, when necessary, proper process changes can be made to maintain the production capacity. Sensors for detecting an eminent algal bloom can be located at the desalination facility to inform personnel regarding changes in water quality that are directly observed on the source water. Fig. 9 presents the transmembrane pressure (TMP) of a microfiltration system that serves as pretreatment to a pilot-scale reverse osmosis desalination system along with the levels of chlorophyll *a* fluorescence observed in the feedwater. It is clear from this figure that increased membrane fouling rates (e.g. faster daily rise in the TMP) were associated with increasing chlorophyll *a* fluorescence (i.e. increased algal biomass) in the source water.

It is well known that higher concentrations of algae cause increased membrane fouling rates in microfiltration systems that are frequently incorporated, or considered, in today's desalination facilities (Gijsbertsen-Abrahamse et al., 2006; Lee and Walker, 2006; Reiss et al., 2006). A more complete approach might include a monitoring system located offshore that measures some of the primary factors influencing algal blooms, such as nutrient monitoring in near-real time using new *in situ* sensor technology (Glibert et al., 2008). Such information would be useful to both the desalination facility and HAB researchers who are continually improving their understanding of the causative factors that produce HABs and their associated toxins. Using the information provided by offshore sensors, the desalination facility personnel could note trends and shifts in driving factors that generate algal blooms and make any chemical orders or perform maintenance procedures that have significant lead times. The same offshore sensor might also incorporate real-time monitors of sentinel parameters for changes in algal biomass, such as turbidity and chlorophyll *a*, allowing the facility to prepare for changes in chemical additions and redundant equipment service.

Monitoring of basic chemical parameters of seawater (e.g. chlorophyll *a* concentrations) will provide valuable information for facility operations, but this activity is not sufficient to fully assess potentially toxic conditions that might arise from algae that do not require high standing stocks to constitute a significant toxic threat, such as *Alexandrium* species. Species-specific approaches, such as automated *in situ* instruments or laboratory-based methods, as well as chemical/immunological analyses that identify and quantify specific algal toxins are necessary to more thoroughly characterize the potential hazards posed by HAB species. The consistent removal of these potentially toxic substances through the reverse osmosis process is both a function of size (e.g. molecular weight) and charge (e.g. zeta potential) (Amy et al., 2005). Depending on the size and charge of the contaminant of concern, the rejection, or removal, by the reverse osmosis process will differ. It is important that we continue to broaden our knowledge on potentially toxic substances excreted by algal stock and their associated blooms.

The approach for obtaining this information would be best complemented with knowledge of the species that are present regionally, the potential problems they pose (e.g. specific toxins and the amounts of soluble microbial products and extracellular polymeric substances excreted), the spatial extent of HAB episodes, and their seasonality. The seasonality of *Pseudo-nitzschia* spp. and domoic acid near the intake of a pilot desalination plant in El Segundo, CA, exemplifies the usefulness of routine monitoring for identifying potentially toxic conditions in coastal waters adjacent to a plant (Fig. 10). Abundances of *Pseudo-nitzschia* spp. and concentrations of domoic acid contained in the

algal cells or dissolved in the intake water exhibited a springtime peak. Knowledge of the seasonality of this toxic bloom-forming species allows intensive sampling of coastal waters during spring when toxic events are more common, improving the overall effectiveness of the monitoring effort and making it more cost effective.

Historical and real-time information on the spatial distribution of HABs can provide information vital for optimizing design and performance of desalination operations. Local/regional hydrography, and resultant algal blooms can differ dramatically. When constructing a new intake pipeline, the selection of its location (e.g. depth and distance from shore) can be greatly enhanced through the use of offshore monitoring devices and efforts to take into account the presence of any local accumulations of algal biomass due to currents, water mass convergences/divergences or internal waves, and also subsurface maxima in algal abundance. Properly locating offshore monitoring can provide significant information that will allow optimal location of a new intake pipeline or identification of issues that might affect an existing one, thereby significantly reducing the organic and suspended solid loads present in the feedwater during algal bloom events. These considerations will ease pretreatment operations, reduce the cost of water production, and help improve the facility's longevity.

5. Conclusions

5.1. *Potential impacts, unresolved issues and research prospectus*

The presence of harmful algae in coastal waters that might be employed in reverse osmosis desalination pose potential problems for these operations that have been known to even cause desalination facilities to temporarily cease production (Tenzer et al., 1999; Pankratz, 2008). As the number of seawater desalination facilities continues to grow with lower costs and increasing demand, it is essential that these operating facilities develop the tools necessary to allow process changes and ensure capacity objectives continue to be met. Regardless of the pretreatment configuration, changes in source water quality require adjustments and these changes need to carefully coordinate to ensure that the reverse osmosis membranes are not irreversibly fouled or damaged in the process.

Benchmark work is required to establish the effectiveness of the seawater reverse osmosis process in dealing with HAB toxins and other phytoplankton-derived substances. Even if advanced pretreatment technologies such as microfiltration are implemented upstream of the reverse osmosis process, passage of transparent extracellular material produced by the algal bloom (Alldredge et al., 1993; Hong et al., 1997) may affect reverse osmosis membrane performance. Additionally, the physical durability of phytoplankton varies greatly and the pretreatment process might disrupt cells and create significantly higher concentrations of dissolved organic substances, including toxins, than were originally present in the source water. For example, dissolved domoic acid has been observed in the seawater passing through the prefiltration process (Fig. 10, bottom panel) but it is unclear if these values are higher due to cell breakage as a result of the prefiltration process. Therefore, it is important that the international desalination community carefully characterize these potential contaminants and their removal to improve treatment approaches in seawater desalination.

To our knowledge, there are no published reports on the effectiveness of reverse osmosis for removing dissolved algal toxins from seawater. Some of these toxin molecules (e.g. domoic acid) are near the theoretical molecular size of molecules rejected by reverse osmosis membranes, but experimental studies are required to validate the effective of this process on toxin removal. In addition, more information will be needed to understand the potential impact of discharged brine and pretreatment backwash water resulting from the reverse osmosis desalination process on the ecology of coastal ecosystems. The use of ferric sulfate or ferric chloride as a pretreatment coagulant would concentrate toxic algae and their associated toxins if they are present in the intake water. Similarly, the discharge of brine resulting from the reverse osmosis process would contain elevated concentrations of dissolved algal toxins relative to unfiltered seawater. The degree of concentration of these toxins would not be expected to be large, but the significance of these processes will depend on the starting concentrations in the raw intake or prefiltered water and the degree of concentration due to treatment. There is presently no information on algal toxins in these discharges.

HABs on the U.S. west coast exhibit significant generalities across geographical and temporal scales (e.g. many of the same species occur throughout the region), but the details of bloom dynamics differ with geographic location, depth and season (and perhaps on interannual and decadal scales). The high degree of variability associated with these events makes constant monitoring of HABs in intake water for desalination a vital issue. Regional HAB programs and regulatory agencies along the U.S. west coast presently provide useful information for some known potential problems (e.g. ASP and PSP toxins) for end users that need information on coastal water quality. Awareness (and augmentation) of this information could improve planning and safe operation of desalination facilities. Monitoring of newly emerging HAB concerns (e.g. *Cochlodinium* spp.), or HABs and toxins that are presently poorly characterized (e.g. NSP, DSP, and yessotoxin poisoning) should also be implemented in the future to allow evaluation of their potential impacts on desalination processes.

New technologies for toxin detection and quantification, and *in situ* monitoring of biological and chemical parameters are rapidly improving our ability to monitor coastal ecosystems and identify potentially problematic situations involving HABs. Advances in *in situ* observing technologies (sensor networks, autonomous sensor-equipped vehicles) provide the capability for obtaining unprecedented resolution in the spatial and temporal distributions of chemical and physical parameters, and some biologically important features (Sukhatme et al., 2007). New approaches and instruments for toxin detection help identify contaminated seafood products, and constitute sentinels for the threats of HABs to marine animal populations. Future uses of coastal waters for desalination will also benefit from, and contribute to, these activities.

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