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Stephen S. Bates, Daniel G. Beach, Luc A. Comeau, Nicola Haigh, Nancy I. Lewis, Andrea Locke, Jennifer L. Martin, Pearse McCarron, Cynthia H. McKenzie, Christine Michel, Christopher O. Miles, Michel Poulin, Michael A. Quilliam, Wade A. Rourke, Michael G. Scarratt, Michel Starr, and Terri Wells

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2020

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Canadian Technical Report of Fisheries and Aquatic Sciences

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Canadian Technical Report of Fisheries and Aquatic Sciences 3384

2020

MARINE HARMFUL ALGAL BLOOMS AND PHYCOTOXINS OF CONCERN TO
CANADA

by

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ABSTRACT

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In Canada, reports of marine harmful algal blooms (HABs) have increased over the past few decades. HABs are caused by the growth of certain phytoplankton that produce phycotoxins or otherwise cause harm. Phycotoxins are problematic to human health, and their cumulative effects are stressors to marine ecosystems by causing the mortality of marine fish, birds and mammals, including species designated at risk. Paralytic Shellfish Poisoning (caused by saxitoxin group toxins produced by *Alexandrium* spp.) has been problematic for years on both the east and west Canadian coasts. Amnesic Shellfish Poisoning (caused by domoic acid produced by *Pseudo-nitzschia* spp.) was identified for the first time worldwide following consumption of blue mussels from eastern Canada in 1987. Domoic acid has since also been found on the west coast. Diarrhetic Shellfish Poisoning (caused by okadaic acid group toxins produced by *Dinophysis* spp. and *Prorocentrum* spp.) was first recognized as a hazard in eastern Canadian waters in 1990, and these phycotoxins have since been found on both the east and west Canadian coasts. Other phycotoxins that may cause harm to human health in Canada include pectenotoxins, yessotoxins, azaspiracids, and cyclic imine group toxins (spirolide toxins, pinnatoxins, and gymnodimines). Multiple harmful algal species have been associated with fish-killing blooms on both east and west Canadian coasts. The range of exotic toxic/harmful algae is expanding in Canadian waters due in part to introductions from ships' ballast water and climate change. The detection of domoic acid and the discovery of several toxigenic diatoms and dinoflagellates in the Canadian Arctic is of increasing concern because of the limited knowledge of HABs in this region. Canada's experience in dealing with toxic events resulted in research and monitoring programs designed to understand HABs and to assist the fishing and aquaculture industries. In spite of decreases in research and phytoplankton monitoring efforts, consumers of molluscan shellfish are still protected by phycotoxin monitoring, which is conducted by the Canadian Food Inspection Agency. Novel phycotoxins and toxic algae will continue to be discovered. Continued vigilance and the maintenance of an effective capacity to manage developing problems via strategic research programs is essential. This document reviews Canadian marine HABs and phycotoxins up to late 2018, and provides a foundation for any future research in this area.

RÉSUMÉ

Bates, S.S., Beach, D.G., Comeau, L.A., Haigh, N., Lewis, N.I., Locke, A., Martin, J.L., McCarron, P., McKenzie, C.H., Michel, C., Miles, C.O., Poulin, M., Quilliam, M.A., Rourke, W.A., Scarratt, M.G., Starr, M., and Wells, T. 2020. Marine harmful algal blooms and phycotoxins of concern to Canada. Can. Tech. Rep. Fish. Aquat. Sci. 3384: x + 322 p.

Au Canada, les observations d'efflorescences algales nuisibles marines (dorénavant identifiées par l'acronyme anglophone HABs) ont augmenté au cours des dernières décennies. Les HABs sont causées par la croissance de certaines espèces de phytoplancton producteur de phycotoxines ou deviennent simplement une nuisance. Les phycotoxines représentent un enjeu pour la santé humaine et leurs effets cumulatifs agissent comme agents stressants pour les écosystèmes marins en provoquant la mortalité de poissons, oiseaux et mammifères marins, incluant les espèces désignées en péril. L'intoxication paralysante par les mollusques (IPM), causée par les toxines du groupe des saxitoxines produites par *Alexandrium* spp., est depuis longtemps un problème le long des côtes est et ouest canadiennes. L'intoxication amnésique par les mollusques (AIM), causée par l'acide domoïque produit par *Pseudo-nitzschia* spp., a été rapportée pour la première fois dans le monde suite à la consommation de moules bleues provenant de l'est du Canada en 1987. Depuis, l'acide domoïque fut aussi détecté sur la côte ouest. L'intoxication diarrhéique par les mollusques (IDM), causée par les toxines du groupe de l'acide okadaïque produites par *Dinophysis* spp. et *Prorocentrum* spp., a été reconnue pour la première fois comme étant un problème dans les eaux de l'est du Canada en 1990, et ces toxines ont depuis été trouvées le long des côtes est et ouest canadiennes. D'autres phycotoxines pouvant être nuisibles à la santé humaine au Canada incluent les pecténotoxines, yessotoxines, azaspiracides, ainsi que les toxines du groupe des imines cycliques (toxines spirolides, pinnatoxines et gymnodimines). Plusieurs espèces d'algues nuisibles ont été associées à des mortalités massives de poissons le long des côtes est et ouest canadiennes. L'étendue d'algues exotiques toxiques/nuisibles augmente dans les eaux canadiennes en raison des introductions par les eaux de ballast des navires ainsi qu'aux changements climatiques. La détection d'acide domoïque et la découverte de plusieurs diatomées et dinoflagellés toxigènes dans l'Arctique canadien sont d'intérêt croissant en raison d'une connaissance limitée sur les HABs de cette région. L'expérience du Canada à faire face aux événements d'algues toxiques a résulté à l'établissement de programmes de monitoring et de recherches destinés à comprendre les HABs ainsi que porter assistance aux industries des pêches et de l'aquaculture. Malgré une diminution des efforts de recherche et de surveillance du phytoplancton, les consommateurs de mollusques sont toujours protégés par un monitoring des phycotoxines, qui est mené par l'Agence canadienne d'inspection des aliments. Cependant, de nouvelles algues toxiques et leurs phycotoxines continueront d'être découvertes. Une vigilance soutenue et le maintien d'une capacité efficace de gestion rapide des problèmes futurs via l'utilisation de programmes de recherche stratégiques sont essentiels. Ce document se veut une revue des événements canadiens d'algues marines nuisibles et des phycotoxines jusqu'à la fin 2018 et fournit une base pour toute recherche future dans ce domaine.

1.0 Introduction

1.1 Marine Harmful Algal Blooms (HABs) and Phycotoxins

Canada has a long history of experiencing what have been commonly known as “red tides”, on both its east and west coasts. The term “harmful algal bloom” (HAB) is now used by scientists because “red tides” are neither tides, nor are they always red, and some are indeed visually undetectable (Anderson 1994; Anderson et al. 2012b; Berdalet et al. 2016; Shumway et al. 2018). The term “harmful algal event” (HAE) is also sometimes used when referring to the impacts of blooms that are of lower cell concentration (Reguera et al. 2016; McKenzie et al. 2020a,b). HABs are made up of phytoplankton (microscopic single-celled algae). Typically, phytoplankton are beneficial because they form the base of the food web and produce about half of the oxygen we breathe (Field et al. 1998; Petsch 2003). However, at least 90 species (~2% of the ~5000 phytoplankton species) also produce potent toxic compounds called phycotoxins (to reflect their algal origin), also referred to more generally as algal toxins or biotoxins (Sournia 1995; Richardson 1997; Landsberg 2002); this number continues to increase. Allen (2018) compiled a list of the >200 species of marine harmful, or potentially harmful, microalgae (worldwide), including those that produce phycotoxins. Lassus et al. (2016) list 174 taxa.

During the past decades, novel marine phycotoxins and toxigenic phytoplankton have been discovered in Canadian marine waters. The chronology of the first detection of phycotoxins in Canadian waters is shown in [Table 1](#). Some of these are known to be problematic elsewhere in the world, but others have appeared here for the first time. Indeed, the number of reports of toxic and harmful algae is also increasing worldwide (Anderson 1989; Hallegraeff 1993; Anderson et al. 2008, 2012b; Lassus et al. 2016). This is likely explained by a combination of factors, including improved awareness, expanded monitoring efforts, increased aquaculture operations, climate change (including unusual climatic conditions), novel shipping routes and increased volumes of ship ballast water, and anthropogenic eutrophication, although the exact causes are still a matter of debate (Smayda 1990, 2007; Mudie et al. 2002; Glibert et al. 2005b; Anderson et al. 2012b, 2015; Lassus et al. 2016; Glibert 2020). Nevertheless, there is now a scientific consensus that the quantity and composition of nutrient inputs are important factors (Glibert and Burkholder 2006, 2018; Heisler et al. 2008; Howarth 2008; Glibert et al. 2010). Anderson (2014) stated that “the global expansion of HAB phenomena is real, due in part to our ability to better define the boundaries of the problem.” Furthermore, he concluded that “the global problem of HABs is serious and much larger than we thought.”

Whatever the cause, the presence of these marine phycotoxins and toxic algae can have a great impact on human health, health costs, economic costs, marine mammals, seabirds, wild finfish, the salmon aquaculture industry, aquaculture and wild molluscan shellfish industries, food webs, and ecosystems ([Table 2](#)). Measures are therefore required to understand and cope with the problems.

Table 1. First detection of phycotoxins in Canadian waters. First illnesses are indicated in the text for Paralytic Shellfish Poisoning (PSP; caused by saxitoxin group toxins), and Diarrhetic Shellfish Poisoning (DSP; caused by okadaic acid group toxins). ND = not detected.

Phycotoxin	West Coast (BC)		East Coast		
	First report	Reference	First report	Province ¹	Reference
Saxitoxin	1942	Quayle 1966	1939	NB	Gibbard et al. 1939
Domoic acid	1992	Forbes and Chiang 1994	1987	PE	Subba Rao et al. 1988a
Okadaic acid group toxins	2011	McIntyre and Kosatsky 2013	1989	PE	Holmes 1991
Spirolide toxins	2011	CFIA and NRC data	1991	NS	Hu et al. 1995c
Pectenotoxins	2003	CFIA and NRC data	2000	QC	CFIA and NRC data
Gymnodimines	2003	CFIA and NRC data	2004	NS	CFIA and NRC data
Yessotoxins	2004	CFIA and NRC data	2004	NL	Finch et al. 2005
Azaspiracids	–	ND	2005	NS	Garnett et al. 2006
Pinnatoxins	2011	CFIA and NRC data	2008	NB	McCarron et al. 2012

¹ NL = Newfoundland and Labrador; NS = Nova Scotia; NB = New Brunswick; PE = Prince Edward Island; QC = Quebec; BC = British Columbia

Table 2. Impacts of marine phycotoxins in Canada and internationally.

Impacts on	Reference
Human health	Quayle 1969; Prakash et al. 1971; Landsberg et al. 2005; Ibrahim 2007; Zaccaroni and Scaravelli 2008; Lefebvre and Robertson 2010; Picot et al. 2011; Landsberg et al. 2014; Singh et al. 2015; Berdalet et al. 2016; Grattan et al. 2016a,b, 2018a,b; Pulido 2016
Health costs	Cembella and Todd 1993; Todd 1995; Wessells et al. 1995; Van Dolah et al. 2001; Hoagland and Scatasta 2006
Economic costs	Wessells et al. 1995; Forbes and Martin 1996; Hoagland and Scatasta 2006; McKenzie 2006; Haigh and Esenkulova 2014; Sanseverino et al. 2016; Adams et al. 2018; Niehörster and Murnane 2018; Ritzman et al. 2018; Moore et al. 2019
Marine mammals	Scholin et al. 2000; Durbin et al. 2002; Van Dolah et al. 2003; Doucette et al. 2006; Fire et al. 2009, 2010; Starr et al. 2017; Bates et al. 2018; Broadwater et al. 2018

Impacts on	Reference
Seabirds	Fritz et al. 1992; Work et al. 1993; Shumway et al. 2003; Jessup et al. 2009; Scarratt et al. 2014; Starr et al. 2014; Bates et al. 2018; Gobble and Hoover 2018; Soliño et al. 2019; Van Hemert et al. 2020
Wild finfish	White 1977, 1980, 1981a,b, 1984; Gosselin et al. 1989; White et al. 1989; Haya et al. 1990; Martin and Wildish 1990; Rensel 1991, 1992, 1993, 1995, 2007; Robineau et al. 1991a,b, 1993; Rensel and Whyte 2003; Bates et al. 2018; Mincarelli et al. 2018; Brown et al. 2020
Salmon aquaculture industry	Black 1991; Black et al. 1991; Whyte et al. 2001a,b; Cembella et al. 2002; Chang et al. 2005, 2006, 2007b; Martin et al. 2006a,c, 2007b; BurrIDGE et al. 2010; Haigh and Esenkulova 2012, 2014; Haigh et al. 2014; Brown et al. 2020
Cultured and wild molluscan shellfish industries	Shumway 1990; Jellett 1993; Bates 1997, 2004, 2006; Whyte 1997; Martin et al. 2001a; Haya et al. 2004; Trainer et al. 2008; Haigh and Esenkulova 2014; Brown et al. 2020
Food webs/ecosystems	Liu et al. 2007; Lefebvre et al. 2012; Trainer et al. 2012; Roncalli et al. 2016; Bates et al. 2018; Burkholder et al. 2018; Mincarelli et al. 2018

Originally, the major marine phycotoxins were categorized according to the symptoms they generate. In Canadian waters ([Table 3](#)), these include: Paralytic Shellfish Poisoning (PSP) toxins, produced by certain dinoflagellates, which cause muscle paralysis and potentially death; Amnesic Shellfish Poisoning (ASP) toxin (domoic acid), produced by certain pennate diatoms, which causes short-term memory loss and potentially death; and Diarrhetic Shellfish Poisoning (DSP) toxins, produced by certain dinoflagellates, which cause gastrointestinal illness (mostly diarrhea) without neurologic manifestations. In March 2004, however, the FAO/WHO/IOC organized a Joint FAO/WHO/IOC *ad hoc* Expert Consultation on Biotoxins in Molluscan Bivalves in Dublin to assess the risks of a number of biotoxins present in bivalve molluscs. The Expert Consultation categorized the biotoxins into distinct groups based on chemical structure, rather than on the symptoms they generate (Toyofuku 2006; [Advance pre-publication version with full authors](#)), and this organization will be used in this review:

- Saxitoxin (STX) group
- Domoic acid (DA) group
- Okadaic acid (OA) group
- Pectenotoxin (PTX) group
- Yessotoxin (YTX) group
- Azaspiracid (AZA) group
- Cyclic imine group:
 - Spirolides (SPXs)
 - Pinnatoxins (PnTXs)
 - Gymnodimines (GYMs)

Table 3. Potentially toxic and toxic phytoplankton that may affect human health, and the phycotoxins potentially produced, in Canadian waters. Closure levels are given per unit wet weight of the whole animal, unless indicated otherwise (Health Canada 2016; CSSP 2019). Note that the unit given in websites is mg kg⁻¹, although the equivalent unit, µg g⁻¹, is also used in this review. Some of the responsible phytoplankton have also been reported in the Canadian Arctic, as detailed in Sections [2.4.5](#), [3.4](#), [4.6](#), [10.2](#), and in [Appendix 1](#).

Name of poisoning	Phycotoxin group	Closure level	Responsible organisms	Provinces where phycotoxins detected
Paralytic Shellfish Poisoning (PSP)	Saxitoxin (STX) and derivatives (e.g. gonyautoxin, neosaxitoxin)	80 µg 100 g ⁻¹ STX _{eq} in edible tissue	<i>Alexandrium acatenella</i> <i>A. catenella</i> <i>A. ostenfeldii</i>	BC NL, PE, NS, NB, QC, BC NS, NB ¹ , QC ¹ , BC ¹
Amnesic Shellfish Poisoning (ASP)	Domoic acid (DA)	20 µg g ⁻¹ in edible tissue	<i>Pseudo-nitzschia australis</i> <i>P. multiseriis</i> <i>P. pseudodelicatissima</i> <i>P. seriata</i>	NB, BC NL ¹ , PE, NB, NS, BC ¹ NL ¹ , PE, NB, NS, BC ¹ NL ¹ , PE, NB, NS
Diarrhetic Shellfish Poisoning (DSP)	Dinophysistoxin (DTX): Sum of okadaic acid (OA), DTX1, DTX2, OA esters, DTX1 esters, DTX2 esters	0.2 µg g ⁻¹ in edible tissue ² ; 1 µg g ⁻¹ in digestive tissue (under review)	<i>D. acuminata</i> ¹ <i>D. acuta</i> ¹ <i>D. fortii</i> ¹ <i>D. norvegica</i> ¹ <i>Prorocentrum lima</i>	NL, PE, NB, NS, QC, BC NL, PE, NB, NS, QC, BC BC NL, PE, NB, NS, QC, BC NL, PE, NB, NS, QC
None	Spirolide toxins (SPXs)	Not yet established	<i>Alexandrium ostenfeldii</i>	NL, PE, NB, NS, QC, BC
None	Pinnatoxins (PnTXs)	Not yet established	<i>Vulcanodinium rugosum</i> ³	NL, PE, NB, NS, QC, BC
None	Gymnodimines (GYMs)	Not yet established	<i>Alexandrium ostenfeldii</i> ⁴ <i>Karenia selliformis</i> ³	NL, NS, QC, BC
None	Pectenotoxins (PTXs): Sum of PTX1, PTX2, PTX3, PTX4, PTX6, PTX11	0.2 µg g ⁻¹ in edible tissue; 1 µg g ⁻¹ in digestive tissue	<i>Dinophysis acuminata</i>	NL, PE, NB, NS, QC, BC

Name of poisoning	Phycotoxin group	Closure level	Responsible organisms	Provinces where phycotoxins detected
None	Yessotoxin group (YTXs)	Not yet established ⁵	<i>Gonyaulax spinifera</i> ⁶ <i>Protoceratium reticulatum</i>	NB ¹ , NS ¹ , BC ¹ NL, PE, NB, NS, BC
None	Azaspiracid group (AZAs): AZA1–AZA26	Not yet established ⁷	<i>Azadinium poporum</i> ³ , <i>A. spinosum</i> ³	NL, NS, QC

¹ Organism responsible for producing the phycotoxin not proven in Canadian waters

² When Canada exports product to the European Union (EU) and the U.S., it respects the regulatory limit of 0.16 µg g⁻¹ in edible tissue. The EU limit for OA also includes PTX1 and PTX2.

³ Organism not detected in Canadian waters

⁴ Not a proven producer of GYMs in Canadian waters

⁵ When Canada exports product to the EU, it respects the EU regulatory limit of 3.75 µg YTX_{eq} g⁻¹ in the whole animal (European Commission 2013)

⁶ Although present, not a proven toxin producer in Canadian waters

⁷ When Canada exports product to the EU, it respects the EU regulatory limit of 0.16 µg g⁻¹ (for AZA1, AZA2, AZA3)

These phycotoxins are produced by a variety of phytoplankton that can form HABs (Shumway et al. 2018). The conditions that cause the HAB species to proliferate, and the reasons why the phytoplankton produce these toxins, are still a matter of intense research (Glibert and Burkholder 2018). However, the phycotoxins can become concentrated in the stomach and flesh of various molluscan bivalves (e.g. mussels, oysters, clams, scallops) by their normal filter-feeding activity. When these bivalves are consumed by other marine organisms or humans, the phycotoxins can cause illness or even death (Basti et al. 2018; Farabegoli et al. 2018). Some phycotoxins can accumulate throughout the food web during a HAB, moving from the phytoplankton to zooplankton, fish, marine birds, and mammals. Some phycotoxins can also have deleterious effects on bivalves and otherwise cause ecosystem disruptions: delayed escape responses to predators (Hégaret et al. 2012); suppressed immunocompetence (Hégaret and Wikfors 2005; Hégaret et al. 2012); disruption of the reproduction and early stage development of aquatic organisms (Vasconcelos et al. 2010; Jensen et al. 2015; Roncalli et al. 2016; D’Agostino et al. 2017); genotoxicity in hemocytes and DNA damage to gill cells of mussels (Prego-Faraldo et al. 2013, 2016); growth inhibition of co-occurring phytoplankton via allelopathy (Xu et al. 2015); detrimental affects on respiratory capabilities, feeding behaviour, and reproductive condition of whale populations (Durbin et al. 2002); and alteration of various metabolic processes (Pousse et al. 2019; Ventoso et al. 2019). HAB-related mass pathologies and mortalities of bivalve molluscs have also been reported worldwide (Shumway 1990; Landsberg 2002; Banno et al. 2018).

Early reviews of phycotoxins (mostly STX group toxins) present in Canadian waters, excluding the Canadian Arctic regions, were written by Medcof et al. (1947), Edwards (1956), Quayle (1966), Prakash et al. (1971), and Todd (1997). Other phycotoxins have since been discovered in Canadian waters (Tables 1, 3), either in molluscan shellfish, dinoflagellate species, or seawater, and include:

- Pectenotoxins (Anderson et al. 2001; McCarron et al. 2014a)
- Yessotoxins (Finch et al. 2005; McCarron et al. 2014a)
- Azaspiracids (Garnett et al. 2006; Twiner et al. 2008)
- Spirolides (Hu et al. 1995c, 2001; Cembella et al. 1998, 1999, 2000, 2001a,b; McCarron et al. 2014a)
- Pinnatoxins (McCarron et al. 2012)
- Gymnodimines (McCarron et al. 2014a)

These compounds are not known to have caused human illnesses in Canada, but their presence must be monitored (Quilliam 2003a), and their potential effects will therefore be considered in this review.

Some phytoplankton produce toxins that have not yet been characterized, including some ichthyotoxins that kill fish (Table 4). Other phytoplankton do not produce any phycotoxins, but cause harm to fish by mechanical damage, or to pelagic and benthic organisms by depleting the oxygen content of water due to the decay of exceptionally dense blooms of toxic or nontoxic algae (Table 4). Those phytoplankton that cause harm in Canadian waters have been termed “Ecological and Biological Significant Species”, based on their ecological effects, geographic distribution, and frequency of occurrence (Scarratt et al. 2006).

Table 4. Additional problematic phycotoxins and toxic/harmful algae in Canadian waters, excluding the Canadian Arctic regions (see Sections [2.4.5](#), [3.4](#), [4.6](#), [10.2](#), and [Appendix 1](#)).

Phycotoxin	Responsible organism	Location	Selected references
Dinoflagellates			
Goniodomin A	<i>Alexandrium pseudogonyaulax</i>	NB, QC, BC	Martin and LeGresley 2008; Martin et al. 2009; Klein et al. 2010
Unknown	<i>Margalefidinium polykrikoides</i> (formerly <i>Cochlodinium polykrikoides</i>)	BC	Whyte et al. 2001a,b; Haigh 2007; Iwataki et al. 2008
Unknown	<i>Margalefidinium fulvescens</i>	BC	Haigh 2007; Haigh and Esenkulova 2012
None	<i>Hematodinium</i> spp.	NL, BC	Bower et al. 1994; Taylor and Khan 1995; Taylor et al. 2002; Bower 2013
None	<i>Noctiluca scintillans</i>	BC	Quayle 1969; Haigh and Taylor 1990; Chittenden et al. 2018
Cytotoxins ¹ Haemolytic toxins ¹	<i>Gyrodinium aureolum</i>	NB (Bay of Fundy, Gulf of St. Lawrence), BC	Martin and Wildish 1990; Blasco et al. 1994; Martin et al. 1995
Diatoms			
None	<i>Chaetoceros convolutus</i> , <i>C. concavicornis</i>	NL, PE, NB, QC, BC	Bell 1961; Haigh and Taylor 1990; Albright et al. 1992, 1993; Harrison et al. 1993; Taylor 1993; Bérard-Therriault et al. 1999; Haigh 2007, 2017; Martin et al. 2014c; Haigh and Esenkulova 2012
None	<i>Chaetoceros debilis</i>	NL, NB (Bay of Fundy), QC	Wildish et al. 1991; Bérard-Therriault et al. 1999; Penney et al. 2001
None	<i>Corethron pennatum</i>	NB (Bay of Fundy), BC	Speare et al. 1989; Haigh et al. 1992; Martin et al. 2009
Unknown	<i>Leptocylindrus minimus</i>	NB (Bay of Fundy), QC, BC	Forbes and Waters 1993a,b; Martin et al. 1995, 2006a, 2009, 2010b, 2014b,c; Bérard-Therriault et al. 1999;

Phycotoxin	Responsible organism	Location	Selected references
			Haigh et al. 2004a,b,c,d,e; Martin and LeGresley 2014
Raphidophyte			
Unknown	<i>Heterosigma akashiwo</i>	QC, BC	Haigh and Taylor 1990; Bérard-Therriault et al. 1999; Kennedy and Kreiberg 1999; Cassis 2007; Haigh 2007; Haigh and Esenkulova 2014; Haigh 2017
Protozoan ciliate			
None	<i>Mesodinium rubrum</i>	NL, NS, NB (Bay of Fundy), BC	Parsons and Blackburn 1968; Quayle 1969, 1988; Taylor et al. 1971; White et al. 1977; Takahashi and Hoskins 1978; Grice et al. 1980; Martin and Wildish 1990; Watson-Wright et al. 1993a; Parrish et al. 1995; Carver et al. 1996; Bérard- Therriault et al. 1999; Kyewalyanga et al. 2002; Martin et al. 2007b, 2009
Prymnesiophytes			
Digalactosyl glycerol	<i>Chrysochromulina polylepis</i> , <i>C. birgeri</i>	NS, QC, BC	Taylor et al. 1991; Taylor 1993; Forbes and Waters 1993b; Levasseur et al. 1994; Taylor and Haigh 1996; Bérard-Therriault et al. 1999
Unknown	<i>Chrysochromulina</i> cf. <i>hirta</i> (= <i>Haptolina</i> cf. <i>hirta</i>), <i>C.</i> <i>ericina</i> (= <i>Haptolina ericina</i>)	BC	Haigh 2017
Chrysomonadales			
Unknown	<i>Mallomonopsis elliptica</i> or <i>Mallomonas acaroides</i>	NS	C. Carver unpubl. data
Silicoflagellates (Dictyochophytes)			
Unknown	<i>Octactis speculum</i>	NL, NB (Bay of Fundy), QC, BC	Brunel 1962; Haigh and Taylor 1990; Taylor et al. 1991; Bugden et al. 1992; Taylor et al. 1994; Bérard-

Phycotoxin	Responsible organism	Location	Selected references
			Therriault et al. 1999; Haigh and Esenkulova 2012, 2014; Martin et al. 2014c; Haigh et al. 2018
Unknown	<i>Dictyocha fibula</i>	NB (Bay of Fundy), QC, BC	Bérard-Therriault et al. 1999; Martin et al. 2014c; Haigh et al. 2018; Van Guelpen and Pohle 2019
Unknown	<i>Pseudochattonella verruculosa</i>	BC	Haigh et al. 2014; Haigh 2017; Jones 2017
Cyanobacteria			
Microcystin-LR	Source unknown in Canadian waters	PE, NB, BC	Andersen et al. 1993; Boland et al. 1993; Chen et al. 1993

¹ Only documented in *Gyrodinium aureolum* from Norway, Scotland, and Ireland (Dahl and Tangen 1993)

The safety of bivalve molluscan shellfish consumed by Canadians is protected by the Canadian Shellfish Sanitation Program ([CSSP](#)), which is jointly administered by the Canadian Food Inspection Agency ([CFIA](#)), Environment and Climate Change Canada ([ECCC](#)), and Fisheries and Oceans Canada ([DFO](#)). When the CFIA determines that the concentration of phycotoxins in monitored shellfish approaches the respective regulatory limit, at or above which harm may be caused to human health ([Table 3](#)), the harvesting of the seafood product is immediately stopped by DFO (Conservation and Protection Branch). Information about which harvesting areas are closed due to unacceptable levels of phycotoxins is given for each of the DFO Regions in Canada. Current closure information for the Pacific, Quebec, Gulf, Maritimes, and Newfoundland and Labrador Regions can be found [here](#). Note that not all closures due to phycotoxins are currently captured with this system. A map of Canada showing which bivalve shellfish harvesting areas are open or closed in real time is available [here](#). Note that harvesting closures listed at the above two websites can also be due to sanitary reasons (including unsatisfactory levels of fecal coliform bacteria, viruses or chemicals in water and/or shellfish), as determined by ECCC. The [BC Toxicology News Monthly Bulletin](#) summarizes toxicological events, including those involving the major phycotoxins, in BC.

It is essential that physicians become aware of the potential for shellfish poisonings to occur in both commercial and self-harvested products in Canada (McIntyre and Kosatsky 2013; Bouchouar et al. 2014; Schroeder et al. 2015; Wooten and Parsh 2017) and as a result of consumption by travellers (Ansdell 2019). Furthermore, physicians should be capable of recognizing the symptoms of each of the phycotoxin poisons and of implementing any treatments, if available. This information is included herein. As previously reviewed (Lelong et al. 2012; La Barre et al. 2014; Bates et al. 2018), phycotoxins have a high potency towards marine birds (pelicans, cormorants, shearwaters, murres) and mammals (dolphins, seals, sea lions, whales), and terrestrial birds (chickens) and mammals (cats, dogs, cows, humans), including laboratory animals (mice, rats, monkeys). Nevertheless, evidence

does not support that they have a primary role in defending against predators (Lincoln et al. 2001; Cembella 2003; Anderson et al. 2012b). However, there are some exceptions with respect to disrupting predator-prey interactions (Mitra and Flynn 2006; Lundholm et al. 2018).

General overviews of marine phycotoxins and HABs, and their risks, are available (Anderson 1994; Zingone and Enevoldsen 2000; Landsberg 2002; Sellner et al. 2003; Glibert et al. 2005a, 2018; Landsberg et al. 2005, 2014; Backer and McGillicuddy 2006; Picot et al. 2011; Anderson et al. 2012b; Daneshian et al. 2013; Berdalet et al. 2016; Visciano et al. 2016; Farabegoli et al. 2018; Shumway et al. 2018). International organizations have also summarized various aspects of phycotoxins, including toxicology and risk assessment (FAO 2004; FAO/IOC/WHO 2005; Toyofuku 2006; EFSA 2008a,b,c, 2009a,b,c, 2010; Lawrence et al. 2011; Peteva et al. 2016) and HAB research (Global Ecology and Oceanography of Harmful Algal Blooms; GEOHAB, now called [GlobalHAB](#); Berdalet et al. 2017; Glibert et al. 2018). Canadian scientists have played an important role in developing analytical techniques and assessing phycotoxins for these international organizations. Several websites compile relevant information ([ISSHA](#), [IOC](#), [WHOI](#)). Quayle (1969) and Prakash et al. (1971) provide a comprehensive historical coverage of HABs in Canada for the period when it was believed that producers of STX group toxins (causing PSP) were the main concern. Hargraves and Maranda (2002) describe the toxic or harmful phytoplankton from the northeast coast of North America, including the southeast coast of NS. A portion of the present review was published earlier from the perspective of the Canadian shellfish aquaculture industry (Bates 1997).

1.2 Scope of this review

Given the increase in the number and types of toxic or potentially toxic phytoplankton species and phycotoxins in Canadian waters, it is timely to write a review whose goals are to: 1) summarize the marine harmful algae and phycotoxins that are known to occur in Canadian waters; 2) discuss additional toxins and organisms that may be of concern; 3) outline monitoring programs that address (or have addressed) phycotoxins and HAB species; 4) indicate gaps in our knowledge about phycotoxins and HABs in Canada; and 5) suggest future research directions. A brief overview of the history of the DFO's Phycotoxins Working Group (PWG) (Section [1.3](#)), along with other committees and workshops on HABs, will provide context of HAB research in Canada.

This review discusses marine phycotoxins, the phytoplankton that produce them, and their temporal distribution along the Atlantic, Pacific and Arctic coasts of Canada, documented until the end of 2018. It was submitted for peer review and authorization on July 15, 2019, and final authorization was received on June 17, 2020; new relevant literature was added in 2020. The focus is on those phycotoxins and phytoplankton that can cause harm to human health, to the marine ecosystem, or to the wild shellfish harvest, as well as to the aquaculture and fishing industries in Canada. Expressly excluded are discussions on the physiological ecology of these phycotoxin-producing organisms, discussed elsewhere (Anderson et al. 1998; Bates and Trainer 2006; Trainer et al. 2008, 2012; Lelong et al. 2012; Bates et al. 2018). Also excluded are brevetoxins (causing Neurotoxic Shellfish Poisoning; NSP) (Landsberg 2002; Steidinger and Meave del Costillo 2018) and Ciguatera Fish Poisoning (CFP) toxins (Todd 1985, 1995, 1997; Todd and Holmes 1993; Todd et al. 1993; Pilon et al. 2000; Friedman et al. 2017; Steidinger and Meave del Costillo 2018). These toxins are found in more southern, warmer waters, although they can cause problems when Canadian tourists consume contaminated fish abroad, or when tropical fish are consumed in Canada. Todd (1997) lists reports of CFP in Canada from 1983 to 1997. Subsequently, on June 17, 1998, cases of CFP were

reported by a physician, when two families had eaten imported barracuda purchased in Montreal (Pilon et al. 2000). In April 2015, 16 of 19 crew members of a foreign ship carrying potash into Saint John (NB) were admitted to the Saint John Regional Hospital with food poisoning. It was traced to the consumption of fish captured in “international waters”, and was thought to be CFP (Muecke et al. 2015). Thus, it is imperative that Canadian physicians become aware of the symptoms and treatments of illnesses caused by phycotoxins, even if not from Canadian waters. Researchers at NRC (Halifax, NS) are involved in research with ciguatoxins (e.g. Robertson et al. 2018).

Non-algal toxins such as histamine and tetramine will also be excluded from this review. An excess level of histamine and associated biogenic amines, usually in spoiled marine fish, has caused numerous poisoning cases in Canada (Todd 1997). Known as scombroid poisoning, because of its origin in scombroid fish such as mackerel (*Scomber* spp.) and tuna (*Thunnus* spp.), and or histamine fish poisoning, this syndrome is the most common form of seafood poisoning worldwide (Hungerford 2010). Symptoms include oral numbness, headache, dizziness, anxiety, palpitations, rapid and weak pulse (low blood pressure), thirst, hives and rash, flushing and facial swelling, and difficulty in swallowing. Nausea, vomiting, abdominal cramps and diarrhea may also be experienced.

Tetramine (tetramethylammonium ion; not to be confused with a [toxic rodenticide](#)) is responsible for numerous incidents of human intoxication in Japan, Europe and Canada (Watson-Wright 1992b; Todd 1997). It is an autonomic ganglionic blocking agent that can cause a number of symptoms, including nausea, vomiting, anorexia, weakness, fatigue, faintness, dizziness, photophobia, impaired visual accommodation, and dryness of the mouth. The first incidence of tetramine poisoning in Canada was reported in the Bay of Fundy area of NS (Watson-Wright 1992b), and the second in Labrador (Zhao et al. 1997b). It is caused by the consumption of contaminated whelks (sea snails), which include species of the genus *Neptunea*: *N. arthritica* and *N. intersculpta* in Japan, *N. antiqua* in Europe, and *N. decemcostata* and *N. despecta tornata* in eastern Canada (Anthoni et al. 1989; Watson-Wright et al. 1992b; Kenchington and Lundy 1996; Zhao et al. 1997b). These snails are often taken as a by-catch in trawling for scallops. The toxin is produced by, and located mainly in, the salivary (or “buccal”) glands of the whelks (Todd 1997), which if removed prior to consumption can prevent the poisoning. The NRC (Halifax, NS) developed an analytical technique for tetramine, using capillary zone electrophoresis-tandem mass spectrometry (Zhao et al. 1997b). It quantified this toxin in a sample of *N. decemcostata* from NS and from *N. despecta tornata* from Labrador. The latter was cooked material taken from a meal of a family in Labrador suffering from a severe case of tetramine poisoning (Zhao et al. 1997b).

Tetrodotoxin (TTX) is a potent neurotoxin (Knutsen et al. 2017), also from warm southern waters, found in pufferfish. Although not yet detected in Canadian molluscan shellfish, TTX is present in European (Turner et al. 2015a; Vlamis et al. 2015) and New Zealand (Biessy et al. 2018) bivalves, and is now monitored routinely in at least the Netherlands (Gerssen and Gago-Martinez 2019) and the United Kingdom (Turner et al. 2017). Its biological source remains uncertain, but TTX is thought to be of bacterial origin (references in Jal and Khora 2015; Biessy et al. 2018). Nevertheless, it has a potential link to dinoflagellates (Kodama et al. 1996; Vlamis et al. 2015) and is a potential emerging phycotoxin, so it is included in this review (Section [10.3.2](#)).

Freshwater cyanobacterial toxins are not reviewed here, although they may be found in marine shellfish (Gibble et al. 2016; Turner et al. 2018) and mammals (Miller et al. 2010). Cyanobacteria (blue-green algae) are not actually algae, but are often lumped in with other HAB species because

they can produce cyanotoxins. They are known as cyanobacterial HABs, or cyanoHABs (Paerl et al. 2016; Levey 2017). The cyanotoxins they produce (microcystins, anatoxins, cylindrospermopsins, STX group toxins) are of concern to human and animal health because of their presence in lakes and drinking water in Canada (Health Canada 2002, 2017). A special session on the ecology of freshwater cyanobacteria was held in 2007, at the 10th Canadian Workshop on Harmful Algae (CWhA), when “marine” was first removed from the workshop name ([Table 5](#)). The NRC (Halifax, NS) develops analytical methods and reference materials for cyanotoxins (e.g. Hollingdale et al. 2015; Mallia et al. 2018; Miles et al. 2018a; [Wikipedia](#)). Cyanotoxins in marine coastal waters are discussed in [Section 10.3.1](#), and selected information on freshwater cyanobacteria in Canada is found in [Appendix 2](#).

1.3 DFO working groups and workshops on phycotoxins and harmful algae

The Phycotoxins Working Group (PWG) was established by DFO in 1988 (Whyte 1997), as the result of human poisonings by a novel toxin (domoic acid; DA) in late 1987 in eastern PE ([Section 3.2.1](#)). The 1987 event prompted the Treasury Board of Canada to establish additional funding for phycotoxins research on both coasts. The PWG had representatives from all of the DFO Regions (Newfoundland and Labrador, Maritimes, Gulf, Quebec, Central and Arctic, and Pacific). The role of the PWG was to provide advice on the coordination, planning and prioritizing of national DFO research on phycotoxins and other harmful aspects of algal blooms, and the organisms producing them (including environmental factors, bloom forecasting, biological pathways of the toxins, their ultimate fate, and effects on the food web). This was accomplished by: 1) analyzing HAB problems and issues; 2) maintaining liaison with other Canadian and international organizations with related interests; and 3) communicating inter-disciplinary HAB information. An earlier, unpublished, version of this review was written by the PWG in late 1994, with the goal of informing management at DFO Headquarters in Ottawa of the status and ongoing risks of marine HABs and phycotoxins.

Another PWG communication mechanism was to hold periodic workshops to: 1) promote the exchange of new scientific information on harmful algae and their effects; 2) foster the development of cooperative and collaborative scientific programs; and 3) encourage new research initiatives. Nine Canadian Workshops on Harmful Marine Algae (CWHMA) have been organized, and eight of their proceedings have been published ([Table 5](#)). The 10th Canadian Workshop on Harmful Algae eliminated “marine” in the title because it included a special session on freshwater cyanobacteria.

Table 5. Canadian Workshops on Harmful (Marine) Algae, and their proceedings. All workshops, except for the 10th, had “marine” in the title.

Number	Date	Location	Reference
1	Sep 27–28, 1989	Moncton, NB	Bates and Worms 1989
2	Oct 2–4, 1990	Dartmouth, NS	Gordon 1991
3	May 12–14, 1992	Mont-Joli, QC	Therriault and Levasseur 1992
4	May 3–5, 1994	Sidney, BC	Forbes 1994
5	Sep 11–13, 1996	St. John’s, NL	Penney 1996
6	May 27–29, 1998	St. Andrews, NB	Martin and Haya 1999

Number	Date	Location	Reference
7	May 23–25, 2001	Nanaimo, BC	Whyte 2001
8	May 28–30, 2003	Moncton, NB	Bates 2003
9	Jul 6–8, 2005	St. John's, NL	Not published
10	May 9–11, 2007	Mont-Joli, QC	Not published

As a consequence of changing priorities and funding, scientific research on HABs has decreased (Bates and Keizer 1996). However, phytoplankton monitoring programs in the Bay of Fundy (Section [9.2.2.3](#)) and in the Quebec Region of the Gulf of St. Lawrence (Section [9.2.2.4](#)) remained.

The PWG was disbanded in May 2007, after a final meeting of PWG members at the 10th CWWA in Mont-Joli. HAB events are mostly related to human health issues and thus are primarily the responsibility of the CFIA, which continues to conduct regular monitoring for biotoxins related to food safety. The role of DFO is to administer shellfish harvesting closures, based on the shellfish biotoxin data collected by the CFIA.

In the decade following the disbanding of the PWG, DFO researchers continued to carry out regional studies to address specific issues. For example, following the mass mortalities of farmed Atlantic salmon in 2003, the project “Phytoplankton early warning approaches for salmon in the Bay of Fundy” was funded under the DFO Aquaculture Collaborative Research Agreement Program ([ACRDP](#)). This project involved: 1) training and communication of information on HABs between DFO Science and the southwest NB salmon aquaculture industry; 2) retrospective analyses of abundance patterns of harmful algal species; 3) enhanced phytoplankton, environmental and fish sampling at fish farms; 4) determining the threshold concentrations of phytoplankton that were harmful to farmed salmon; 5) studying water circulation; 6) testing the usefulness of a real-time moored light sensor to detect phytoplankton blooms; 7) evaluating the usefulness of SeaWiFS and MODIS satellite imagery as a tool for detecting offshore phytoplankton blooms in the Bay of Fundy; and 8) continuing a communication system to allow fish farm and DFO personnel to communicate data on phytoplankton counts (Chang et al. 2007b).

Additional HAB work at DFO's St. Andrews Biological Station (SABS) and at the Bedford Institute of Oceanography (BIO) during later years was funded through various sources, often as a smaller component of a large project. Funding sources included: Integrated Multi-trophic Aquaculture ([IMTA](#)), Canadian Network of Centres of Excellence for Aquaculture (AquaNet), Canadian Integrated Multi-trophic Aquaculture Network ([CIMTAN](#)), Program for Regulatory Research ([PARR](#)), and a collaborative project with the Northeastern Regional Association of Coastal Ocean Observing Systems ([NERACOOS](#)). The HAB portions of IMTA, AquaNet, CIMTAN and PARR concentrated primarily on the toxic dinoflagellate *Alexandrium catenella* (Section [2.2](#)), and resulting STX group toxins in shellfish, with a focus on permitting a safe, viable blue mussel aquaculture industry in the Bay of Fundy. The collaborative NERACOOS funding was to investigate the potential for detecting *A. catenella* using remote sensing data.

Another example of the involvement of DFO HAB expertise occurred following a multispecies mass mortality of fish, birds and mammals caused by phycotoxins in the St. Lawrence Estuary in

2008 (Starr et al. 2017; Section [2.4.3.1](#)). DFO researchers later provided advice during investigations into: the possible involvement of toxic phytoplankton in a mass kill of herring along the west coast of NS in December 2016; a shellfish harvesting closure due to DA in parts of the Bay of Fundy in autumn 2016 (Section [3.2.4](#)); and the mortality of right whales in 2017 (Section [10.4](#)).

Interest in HABs was renewed at DFO after the 2015 widespread bloom of the DA-producing diatom *Pseudo-nitzschia*, which affected the entire west coast of North America, including BC (Section [3.3](#)). Consequently, a DFO workshop was held at the Institute of Ocean Sciences (IOS, Sidney, BC) during July 11–13, 2017 (ICES 2018; McKenzie and Martin 2018). Its 16 participants included DFO scientists and emeritus researchers (some of whom were former members of the PWG) and representatives from outside of DFO and Canada. The objectives were to: 1) review HAB-related activities and issues in Canada and internationally; 2) develop a list of prioritized HAB-related research questions that DFO could consider advancing in the near term, as well as the resources and partners required to implement them; and 3) discuss the future and continuing activities of the newly formed DFO HAB Working Group. One recommendation from this workshop was to produce a formal CSAS (Canadian Science Advisory Secretariat) research document that would review and assess the status of HAB knowledge in Canada, identify knowledge gaps, and highlight areas of particular concern for current and future impacts of HABs on ecosystems and resources. Workshops were held during September 25–27, 2018 (BIO, Dartmouth, NS) and March 12–14, 2019 (Victoria, BC; DFO 2020) to prepare for the CSAS research document. One goal was to examine harmful algal events (HAEs) reported in the ICES-IOC Harmful Algal Event Database ([HAEDAT](#)) since 1988 (McKenzie et al. 2020a,b). The present review provides a more comprehensive summary of marine HABs and phycotoxins in Canadian waters.

1.4 Information sources

Information for the present review was taken mostly from peer-reviewed publications, but international, DFO and provincial reports, as well as conference proceedings, Canadian government websites, and personal communications, have also been incorporated. For example, DFO reports summarize workshops on: Pacific coast research on toxic marine algae (Forbes 1991a); harmful algae research in the DFO Maritimes Region (Bates and Keizer 1996); developments for a Canadian GEOHAB program (Martin 2002); and criteria for identifying toxic or harmful phytoplankton species as ecologically significant species (Scarratt et al. 2006).

This review also contains summaries of HAB events in Canadian waters, as given in the yearly reports of the International Council for the Exploration of the Sea–International Oceanographic Commission Working Group on Harmful Algal Bloom Dynamics ([ICES-IOC WGHABD](#)), since 1988. PWG members have attended this Working Group, as well as the IOC International Panel on Harmful Algal Blooms ([IPHAB](#)). This review includes links to HAEs reported in HAEDAT, which was initiated in 1988. HAEDAT contains information on HAEs from member countries belonging to [ICES](#) (bordering the North Atlantic Ocean Canada) (Bresnan et al. 2018) and [PICES](#) (North Pacific Marine Science Organization); Canada has members in both organizations. Several years after HAEDAT was established, a decision was made to divide each country into various reporting areas, with each area assigned a country code. For example, Canada was divided into 40 areas, although the entire coast of BC contains only seven areas. Peña et al. (2004) tested the utility of report forms designed for HAEDAT events on the west coast of Canada.

HAEDAT includes those events that result in management actions, such as closures of shellfish harvesting areas, or negative environmental impacts, such as fish kills and mortalities of marine mammals. In HAEDAT, HAEs are defined as either a phycotoxin accumulation in seafood exceeding the regulatory level that is considered safe for human consumption; water discoloration caused by a high concentration of toxin-producing/harmful algae; or any event where humans, animals or any other organisms are negatively affected by algae (Bresnan et al. 2018). The Canadian information stored in HAEDAT is a 30-year record of the temporal and spatial distribution of HAEs, including data provided by DFO phytoplankton monitoring and research programs, the CFIA biotoxin analyses, and the BC Harmful Algae Monitoring Program ([HAMP](#); see Section [9.2.1](#)). Three decades of Canadian marine HAEs were recently reviewed and evaluated, using information from the HAEDAT database (McKenzie et al. 2018).

HAEDAT does not contain complete datasets, but rather a summary of HAEs and the contact for primary HAE data (ICES 2011). Any events reported in the early years should be treated with caution because of deficiencies in the HAEDAT reporting system during that time. Although the database evolved and improved, there are nevertheless limitations (see [HAEDAT](#) for further disclaimers):

- Not all early records are complete due to difficulties at the time in obtaining the information; communication was by mail, telex, fax, or phone in the years prior to personal computers, internet and e-mail.
- Submitted country reports were written using a typewriter, and although headings were assigned, no forms were used.
- Some representatives did not always complete and submit reports.
- HAEDAT questionnaires evolved over several years and once they were set up after the advent of personal computers, they were first formatted in DOS, then in Word documents, and finally in spreadsheets, whose event data had to be entered manually.
- Many submissions were incomplete, missing information on the exact location of the event, particular phycotoxin involved, maximum phycotoxin concentration, causative phytoplankton species, and shellfish species affected.
- The early shellfish toxicity records (including those of the CFIA) were archived as hard copies and were often not accessible, so were never entered into the system.
- Reports were often not submitted if a country did not have a representative attending the meeting during a particular year.

For these reasons, the ICES-IOC WGHABD chose to treat the years 1988–1997 with caution and removed those data from their analyses and publications. Given the above limitations, HAEDAT information reported here should likewise be used with some caution.

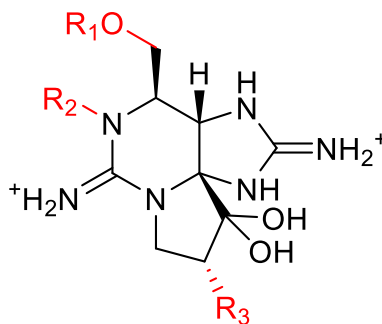
Information from the CFIA biotoxin monitoring program is also included in this review. Although these biotoxin data can be used as a proxy of temporal and spatial trends of HABs, one should be aware that they, also, have limitations (Hamer et al. 2012). The dataset was not collected for research purposes, but rather to protect the consumer of shellfish and therefore human health. A major limitation is the presence of gaps in sampling effort. Once a sample from a harvesting site shows phycotoxin levels equal to or greater than the Canadian maximum levels provided by [Health Canada](#), the area is closed to harvesting. Shellfish sampling for phycotoxins may also be decreased during that closure period, for a minimum of 14 days, during which three acceptable results must be found in order to revoke the closure. Because of this, the maximum levels of phycotoxins may not be quantified. As well, there is an inability to document the exact time that the phycotoxins first

appeared in the bivalve species, and the timing of their uptake/depuration. Using information from the above sources, the phycotoxins, the organisms that produce them, and their distribution on the east and west coasts of Canada, are discussed below.

2.0 Saxitoxin Group Toxins (Paralytic Shellfish Poisoning; PSP)

2.1 Description of saxitoxin group toxins and their biological effects

Toxins within the hydrophilic saxitoxin (STX) group are responsible for numerous cases of Paralytic Shellfish Poisoning (PSP) worldwide (Hall et al. 1990; Mons et al. 1998; Suarez-Isla 2016). There are at least 57 analogues of STX ([Structure 1](#)) of differing toxicity, including gonyautoxins (GTXs), neosaxitoxin (NEO), and *N*-sulfocarbamoyl and decarbamoyl toxins (Wiese et al. 2010). In addition, the low toxicity STX analogue LWTX-1 was found in cyanobacterial mats of *Lyngbya wollei* growing in two fluvial lakes of the St. Lawrence River (Hudon et al. 2016).



Structure 1. The structure of saxitoxins. Saxitoxins are based on a purine structure, with a variety of substituents. Saxitoxin is the principal analogue, with $R_1 = \text{NH}_2\text{C=O}$, $R_2 = R_3 = \text{H}$.

The wide variety of derivatives, and their varying specific toxicities, greatly complicates the chemical analysis of these toxins and the expression of total toxicity, calculated by integrating the potency of all toxins present and expressing it as μg of STX equivalents (STX_{eq}) per 100 g wet weight (ww) of tissue. Furthermore, less toxic forms of the toxin may be converted to more potent ones during chemical extraction, prolonged storage, and also by molluscan shellfish themselves (Cembella et al. 1994; Bricelj and Shumway 1998; Kodama and Sato 2008; Li et al. 2018a). This tends to confound studies of toxin transfer through the food web, but results in a more conservative estimate of toxicity in food for human consumption because of the inclusion of some less toxic forms. Cooking or freezing does not destroy the toxins, although cooking or otherwise heating may decrease the level of toxins (Gill et al. 1985; Mons et al. 1998; Indrasena and Gill 1999; Reboreda et al. 2010).

In Canada, and other parts of the world, closures of shellfish harvesting are initiated when samples reach or exceed the maximum level of $0.8 \text{ mg STX}_{\text{eq}} \text{ kg}^{-1} \text{ ww}$ of tissue (Health Canada 2016) ([Table 3](#)). This is equivalent to $80 \mu\text{g STX}_{\text{eq}} 100 \text{ g}^{-1}$ and the latter units will be used in this review because they are more consistent with the older literature. The European Food Safety Authority (EFSA) has proposed that this action level be lowered to $7.5 \mu\text{g STX}_{\text{eq}} 100 \text{ g}^{-1}$ (EFSA 2009a). The $80 \mu\text{g STX}_{\text{eq}} 100 \text{ g}^{-1}$ action level was based on the AOAC mouse bioassay (AOAC 1990), which has a limit of

detection of about 40 µg STX_{eq} 100 g⁻¹. The mouse bioassay was established in the late 1940s and proved very effective for over 50 years in protecting the public from PSP. However, the method can suffer from interferences leading to false positives (McCulloch et al. 1989) and was viewed as a very cruel assay by animal rights groups.

Originally, STX group toxins were detected and quantified using the mouse bioassay, which was further developed by Gibbard and Naubert (1948) and Stephenson et al. (1955) at the Department of National Health and Welfare. Currently, STX group toxins are analyzed by chemical methods (Dell'Aversano et al. 2005; van de Riet et al. 2009, 2011; Turner et al. 2015b; Thomas et al. 2017) and *in vitro* (Cembella et al. 2003) or various *in vivo* assays (Fernández et al. 2003), as reviewed in Cusick and Sayler (2013) and Ben-Gigirey et al. (2016). There is also a Canadian rapid test for STX group toxins ([Scotia Rapid Testing Ltd.](#), Chester Basin, NS) based on lateral flow immunochromatography (Laycock et al. 2000; Jellett et al. 2002). Other biochemical and functional assays used in Scotland for PSP group toxins and other phycotoxins are described by McLeod et al. (2015). The AOAC mouse bioassay (AOAC 1990) has been replaced by the AOAC 2011.02 post-column oxidation (PCOX) high performance liquid chromatography (HPLC) method (van de Riet et al. 2009, 2011), and is carried out by the CFIA for regulatory purposes in Canada. The advantages of using this method by the CFIA are described by Rourke and Murphy (2014). A pre-column oxidation HPLC method, AOAC 2005.06, has been implemented as the reference method for STX group toxins in the EU (O'Neill and Turner 2015). LC-MS methods have been developed and are in use in Canada and elsewhere (Dell'Aversano et al. 2005; Turner et al. 2015b; Thomas et al. 2017). The Biotoxin Metrology group of the National Research Council of Canada (NRC), in Halifax (NS), produces certified calibration solutions and matrix reference materials for the determination of STX group toxins by chemical analysis methods ([NRC Certified Reference Materials; Wikipedia](#); Thomas et al. 2002, 2016, 2017; Burton et al. 2005; Quilliam 2006).

PSP in humans is caused by ingesting shellfish containing STX group toxins. Symptoms of PSP are neurological and normally appear within a few minutes and up to 10 hours after eating toxic shellfish. They include tingling and numbness of the mouth, lips and fingers, headache, dizziness, abdominal pain and muscular weakness, leading to respiratory paralysis and death by suffocation in the most severe cases (Acres and Gray 1978). There is no antidote to PSP ([WHOI website](#)); seriously affected persons must be hospitalized and placed under artificial respiration until they recover. Other information is available about this toxin group, including their chemical structure, properties, analysis, distribution, source organisms, uptake and depuration by molluscan shellfish, toxicity, clinical characteristics, mechanisms of action, genetics, molecular targets, and ecological functions (Mons et al. 1998; FAO 2004; Cusick and Sayler 2013; Durán-Riveroll and Cembella 2017; Doucette et al. 2018; Durán-Riveroll et al. 2018). More information about PSP is found at the [CFIA website](#) (CFIA 2014a). Edwards (1956) describes the etiology and epidemiology of PSP on the Pacific and Atlantic coasts of Canada up to the late 1940s.

In Canadian marine waters, STX group toxins are produced by several species of the dinoflagellate genus *Alexandrium*, as discussed in the sections below. Anderson et al. (2012a) reviewed the biology of this genus. *Alexandrium* cells are filtered out of the seawater during normal feeding by molluscan (mainly bivalve) shellfish, which then concentrate the toxins. Harm occurs when humans or marine species consume the toxic molluscs. As detailed below, wild and cultured fish mortalities have also been linked to this toxin group, either by transfer through the food chain or directly through exposure to *Alexandrium* cells. It should be pointed out that some strains of

some *Alexandrium* species also produce unidentified allelopathic extracellular toxins, not related to STX group toxins, which caused loss of motility and cell lysis of the heterotrophic dinoflagellates *Oblea rotunda* and *Oxyrrhis marina* (Tillmann and John 2002).

Alexandrium species produce resting cysts (hypnozygotes) that are an important part of the life cycle and act as “seeds” for the next bloom (Prakash 1967; Wall 1975; Dale 1977; Anderson 1998; Anderson et al. 2012a). Resting cysts also contain STX group toxins (Dale et al. 1978; White and Lewis 1982). These cysts fall to the sediment, where they provide a useful record of previous blooms of *Alexandrium*. They are well preserved for long periods, are easy to identify and culture (to assess viability), and they do not have to be collected during a bloom as is required of phytoplankton surveys (e.g. Schwinghamer et al. 1991). Knowledge of their presence, as well as their absence, is therefore important. However, it must be made clear that cyst numbers in sediments are not an indicator of vegetative cell density or bloom concentrations in the following years. Cyst densities are consistently high in the Bay of Fundy and only provide the seed for a bloom. Bloom concentrations tend to be more related to short-term weather patterns (Martin and Wildish 1994; Martin et al. 2014a).

The abundance of fossil cysts has also been used to determine the history of toxic dinoflagellate blooms in Canadian waters (Rochon, as reported in ICES 2000; Mudie et al. 2002). Cyst production was 10 times greater during the early Holocene (10,000–10,500 years ago) than at present. The number of toxin-producing species and cyst production has risen during the past ~2000 years on both the east and west coasts, but has never been as high as during the past 50 years.

2.2 *Alexandrium* species in Canadian waters and changes in nomenclature

The organisms identified as being responsible for STX group toxin production in Canada are dinoflagellate species belonging to the genus *Alexandrium*. The taxonomy of these species has undergone revision from time to time based on both morphological and genetic differences. These toxin-producing species, which were originally in the genus *Gonyaulax* (Lebour 1925), were later transferred to *Protogonyaulax* (Taylor 1979) and are currently included in the genus *Alexandrium* (Balech 1985). In the literature, toxin production is typically associated with three species (*Alexandrium catenella*, *A. fundyense*, and *A. tamarense*) distinguished by minor morphological features in their thecate plates, including the presence or absence of a pore on the first apical plate and the ability to form chains. These species, which occur globally, were believed to have different geographic preferences (Taylor and Harrison 2002), with *A. catenella* occurring on the Pacific coast and *A. fundyense* and *A. tamarense* on the Atlantic coast. Together, they have been termed the *A. tamarense* complex and have been shown to be linked with toxic blooms, causing PSP, around the world (John et al. 2014).

Their designation as true biological species has been disputed because it was not supported by early studies of toxin content, toxin composition, bioluminescence, nor genetics (Anderson et al. 1994; Scholin et al. 1995). Furthermore, these species are able to interbreed (Anderson et al. 1994) and morphological intermediates have been found in the field (Taylor 1984, 1985; Cembella and Taylor 1985, 1986). Phylogenetic analyses of nuclear-encoded ribosomal DNA (rDNA) sequences support the occurrence of five distinct ribotype groups in the *A. tamarense* complex, identified as Groups I–V (Lilly et al. 2007). Confusion arose because these groups do not correspond to the original morphologically defined species, and there have thus been changes in the nomenclature of some *Alexandrium* species. This is further clarified by Litaker et al. (2018). John et al. (2014) assigned

names to each group and all toxin-producing *Alexandrium* isolates from Canada were assigned to Group I, which they called *A. fundyense*, based primarily on molecular data. Although these groups are genetically distinct, they do not correspond to the original morphologically defined species and thus morphological characteristics can no longer be used to distinguish them (Litaker et al. 2018). A dispute over the name of this group was resolved when the Nomenclature Committee for Algae voted that the species name “*catenella*” has priority over “*fundyense*” (Prud’homme van Reine 2017). The currently accepted name for Group I, *A. catenella*, is therefore used for all Canadian *Alexandrium* species, on both coasts, throughout this review. When necessary, the former name is also indicated for clarity.

Now that the classification of *A. catenella* Group I has been settled, the quantification of *A. catenella* via *sxtA*, a gene involved in STX group toxin synthesis, has been evaluated in comparison to traditional cell counts, quantitative polymerase chain reaction (qPCR) assay targeted to a species-specific region of rDNA, and an established fluorescent *in situ* hybridization (FISH) microscopy method (Murray et al. 2019).

STX group toxin production has been confirmed for another species, *A. ostenfeldii*, found in Canadian waters (Qiu et al. 2018). *Alexandrium ostenfeldii* and *A. catenella* are difficult to identify morphologically using light microscopy, but the development of rRNA probes (John et al. 2003; Anderson et al. 2005a) has made it possible to distinguish them in field populations. Finally, although it has not been shown to produce STX group toxins, *Alexandrium pseudogonyaulax* (formerly *Goniodoma pseudogonyaulax*) has also been observed in Canadian waters (Taylor and Haigh 1993).

2.3 Pacific coast

Symptoms consistent with poisoning by STX group toxins on the Pacific coast of Canada ([Fig. 1A](#); note that in this, and all subsequent maps, the absence of shading along a coastline does not imply an absence of phycotoxins, as the area may not be monitored) are reported as far back as 1793, when a member of Captain George Vancouver’s crew died from eating contaminated mussels in what was subsequently called Poison Cove, in BC (reported in Edwards 1956; Quayle 1969; Taylor 1993; Cembella and Todd 1993; Prakash et al. 1971; McIntyre et al. 2013; Durán-Riveroll et al. 2018).

Since monitoring began in 1942, there has not been a year in which toxicity has not occurred and some areas are toxic every year (Quayle and Bernard 1966; Taylor and Harrison 2002). Deaths caused by this toxin group have subsequently occurred in 1942 (three, in Barkley Sound, which prompted the establishment of the monitoring program in 1942; Quayle 1966); 1965 (one, in Theodosia Inlet); and 1980 (one, in southern Queen Charlotte Sound) (Prakash and Taylor 1966; Quayle 1969; Chiang 1988; Taylor and Harrison 2002). Illnesses occurred in 1957 (near Courtenay, oysters); 1963 (near Namu, butter clams); 1970 (Viner Sound, butter clams); 1972 (Barkley Sound, Manila clams); 1975 (Work Channel, butter clams); 1980 (Gilford Island, butter clams); 1981 (Church House, butter clams); 1982 (Work Channel, butter clams); and 1985 (Simoom Sound, butter clams) (Quayle 1988). People continue to be affected by PSP; for example, after eating butter clams harvested on Dundas Island (BC) in November 2018 ([CBC News](#)).

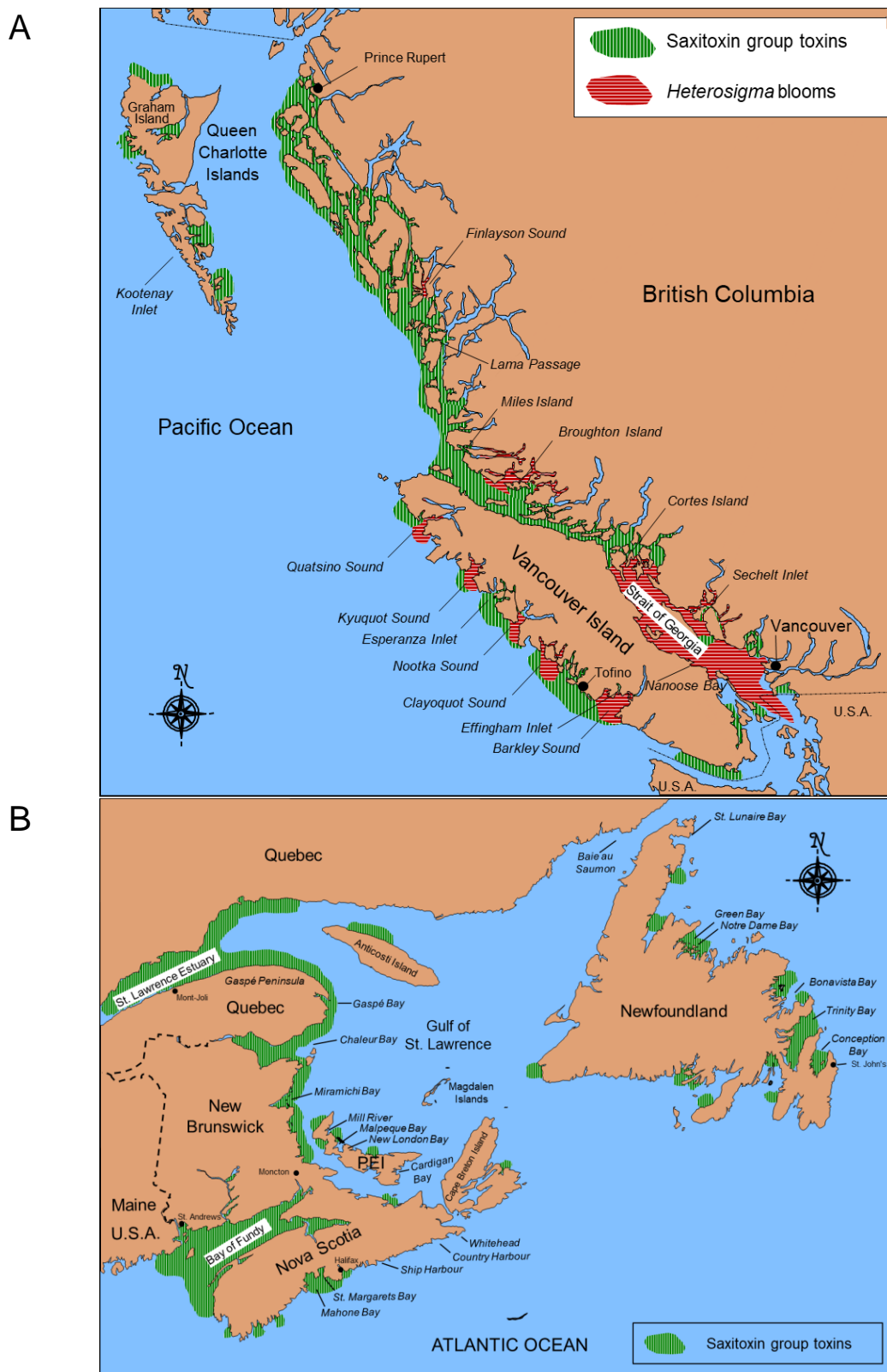


Figure 1. Location of saxitoxin group toxins (green shaded areas) at levels exceeding the regulatory action level ($80 \mu\text{g STX}_{\text{eq}} 100 \text{ g}^{-1}$). A) British Columbia coast. Also shown are the locations of fish-killing *Heterosigma* blooms (red shaded areas). B) Atlantic Canada coast.

The presence of STX group toxins along virtually the entire 27,000 km coast of BC poses a continuing problem (Taylor and Horner 1994). Because the toxins are ubiquitous, and blooms occur unpredictably in time, certain coastal areas remain closed indefinitely to shellfish harvesting, especially those which are too remote for monitoring (Quayle 1988; Jamieson and Lessard 2000). However, additional harvesting sites have been opened because of new industry-funded monitoring programs (Section 9.2.1). Quayle and Bernard (1966) compiled STX group toxin data in BC (in “mouse units”) from the beginning of the monitoring program (1942) to 1965. They noted that it was important “to gather together these 20 years of data under one cover in a useful form before they are irretrievably lost”. Records of STX group toxins were published as “British Columbia toxicity records” (DFO 1970, 1972). A “Summary of paralytic shellfish toxicity records in the Pacific Region” was compiled by DFO (DFO 1982–1996), and was then succeeded by the annual series “Summary of marine toxin records in the Pacific Region”, reported first by DFO (DFO 1992–1996) and then by the CFIA (CFIA 1997–2001), when the DFO Fish Inspection Branch staff became part of the newly formed CFIA in 1997. Such reports are no longer published. Jamieson and Lessard (2000) list shellfish harvesting closures due to phycotoxins for 1997, which included the entire North Coast and McIntyre Bay. Information about PSP in BC is found at the [BC Centre for Disease Control](#) website. Current shellfish harvesting closures for the DFO Pacific Region are posted at the [DFO bivalve shellfish contamination closures website](#).

Even though all toxicity values were below the regulatory limit, Yan et al. (2004) reported a range of 15–20 $\mu\text{g STX}_{\text{eq}} 100 \text{ g}^{-1}$ in mussels (*Mytilus trossulus*) from English Bay and Burrard Inlet, in Vancouver Harbour, an area where shellfish harvesting is normally prohibited. A hierarchical cluster analysis of STX measurements from 49 shellfish monitoring sites along the BC coast during 2002–2012 showed that locations within the Strait of Georgia exhibited more seasonally dependent toxicity, whereas those on the west coast of Vancouver Island had more variable temperatures and circulation patterns, resulting in yearlong toxicity (Finnis et al. 2017). Toxicity events in both geographic areas were most strongly correlated with sea surface salinity and freshwater discharge. The results could be used to optimize monitoring of STX group toxins (Section 9.0).

2.3.1 Species contaminated with STX group toxins in BC

Molluscan and crustacean shellfish species that have accumulated STX group toxins leading to harvesting closures in BC are shown in [Table 6](#). Interestingly, the purple-hinge rock scallop (*Crassadoma gigantea*) can accumulate these toxins in its adductor muscle, where concentrations up to 130 $\mu\text{g STX}_{\text{eq}} 100 \text{ g}^{-1}$ have been observed (Beitler 1991), which is greater than the regulatory limit; this adds to the increasing evidence that the adductor muscle of some scallop species can contain high levels of STX group toxins (Shumway and Cembella 1992, 1993). The butter clam (*Saxidomus giganteus*) can potentially remain toxic for more than a year (Quayle 1969). Nevertheless, the DFO Integrated Fisheries Management Plan for 2000 (DFO 2000) indicated that there was a limited sport and commercial fishery for butter clams, with the warning that they “chronically retain high levels of paralytic shellfish poison (PSP). Openings will be based on the results of biotoxin (PSP) monitoring”. Taylor and Harrison (2002) reported that its harvesting is permanently prohibited along the entire coast. The 2019–2021 management plan (DFO 2019) indicates that the current most important target species has shifted to the Manila clam (*Venerupis philipinarum*), and that landings of butter clams have been low in recent years. Nevertheless, harvesting may still occur for Food Social and Ceremonial purposes, provided that they are tested for phycotoxins.

Table 6. Shellfish species known to become contaminated with saxitoxin (STX) group toxins in coastal waters of BC. The toxin levels ($\mu\text{g STX}_{\text{eq}} 100 \text{ g}^{-1}$ in edible tissue) are the highest values found for the reference listed. Levels are for the whole animal, unless indicated otherwise. The closure level is $80 \mu\text{g STX}_{\text{eq}} 100 \text{ g}^{-1}$ in edible tissue. When two references are given for the same species, this is to indicate the possible range in values.

Organism			
Common name	Scientific name	PSP level	Reference
Molluscs			
Bay mussel	<i>Mytilus trossulus</i>	11,000	Chiang 1988
Blue mussel	<i>Mytilus edulis</i>	30,000	Chiang 1988
Butter clam	<i>Saxidomus giganteus</i>	10,000	DFO 1992
		2000 ¹	Smith and Kitts 1994
Geoduck clam	<i>Panopea generosa</i>	2200 ²	Bricelj and Shumway 1998
Horse clam	<i>Tresus capax</i>	3520	Prakash and Taylor 1966
Humboldt squid	<i>Dosidicus gigas</i>	483 ³	Braid et al. 2012
Lewis’ moonsnail	<i>Polynices lewisii</i>	176	Quayle 1969
Manila clam	<i>Venerupis japonica</i>	4000	Chiang 1988
Native littleneck clam	<i>Protothaca staminea</i>	2200	Chiang 1988
Nuttall’s cockle	<i>Clinocardium nuttalli</i>	1920	Prakash and Taylor 1966
		6700	DFO 1992
Pacific oyster	<i>Crassostrea gigas</i>	1900	Chiang 1988
Pink scallop	<i>Chlamys rubida</i>	930	Lauzier and Parker 1999
Purple-hinge rock scallop	<i>Crassadoma gigantea</i>	1200	Beitler 1991
		130 ⁴	Beitler 1991
Razor clam	<i>Siliqua patula</i>	880	DFO 1992
Ringed lucina clam	<i>Lucinoma annulata</i>	8640	Quayle 1969
Rock scallop	<i>Hinnites multirugosa</i>	240	Quayle 1969
California mussel	<i>Mytilus californianus</i>	10,000	DFO 1994
		8800	CFIA 1997
Soft-shell clam	<i>Mya arenaria</i>	864	Prakash and Taylor 1966
Spiny scallop	<i>Chlamys hastata</i>	5900	Bricelj and Shumway 1998
Weathervane scallop	<i>Patinopecten caurinus</i>	110	DFO 1995

Organism			
Common name	Scientific name	PSP level	Reference
Crustaceans			
Barnacles	<i>Balanus cariosus</i>	51	Quayle 1969
Dungeness crab	<i>Metacarcinus magister</i> (= <i>Cancer magister</i>)	580 ⁵	DFO 1992
Shore crab	<i>Hemigrapsus oregonensis</i>	14,000	Barber et al. 1988a

¹ Foot tissue

² Visceral mass

³ Digestive gland

⁴ Adductor muscle

⁵ Hepatopancreas

California mussels (*Mytilus californianus*) are used in the shellfish monitoring program (Section [9.2.1](#)) and can reach levels up to 10,000 µg STX_{eq} 100 g⁻¹ (DFO 1994). Barnacles (*Balanus cariosus*) also accumulate STX group toxins (Quayle 1969), as do crabs. The shore crab (*Hemigrapsus oregonensis*) accumulated up to 14,000 µg STX_{eq} 100 g⁻¹ during a toxic bloom in Okeover Arm, southern BC (Barber et al. 1988a). Interestingly, this crab, and *H. nudus*, show a seasonal variability in resistance to STX, traced to the presence of a high molecular weight protein (Barber et al. 1988b), which is also found in toxic butter clams (*Saxidomus giganteus*) from BC (Smith and Kitts 1994). Human poisonings were reported in 1965, following consumption of contaminated cockles (*Clinocardium nuttallii*), mussels (*Mytilus edulis*), oysters (*Crassostrea gigas*), and soft-shell clams (*Mya arenaria*) from Malaspina Inlet, northeastern Strait of Georgia (Prakash and Taylor 1966; Lassus et al. 2016). Razor clams (*Siliqua patula*) have rarely shown levels above the regulatory limit, except in 1992, when they reached 880 µg STX_{eq} 100 g⁻¹ (DFO 1992). These toxins have also been found in the hepatopancreas of Dungeness crabs (*Metacarcinus magister*, previously called *Cancer magister*), only rarely above the maximum level, but reaching 580 µg STX_{eq} 100 g⁻¹ at Elbow Bank, on the west coast of Vancouver Island (DFO 1992). STX group toxins are commonly found in the digestive tracts of Dungeness crabs in neighbouring Washington (WA) State and Alaska. In Alaska, highest concentrations were found in the Kodiak management area, with a record value of 1992 µg STX_{eq} 100 g⁻¹. When product exceeds 70 µg STX_{eq} 100 g⁻¹ in the viscera, processors are required to eviscerate the crabs before further processing (K. Ballentine, State of Alaska, Department of Environmental Conservation, Juneau, Alaska, 1995 pers. comm.).

Matter (1994) reviewed the accumulation of STX group toxins in predatory snails and other invertebrates in waters of WA State, adjoining BC. Toxicity in these shellfish has significant implications for management of non-traditional invertebrates, both subsistence harvesting by aboriginal peoples and recreational harvesting. These harvesting methods are changing as a result of the shifting ethnic composition of harvesters, resulting in the exploitation of a broader range of shellfish species than in the past. Species that tested positive for STX group toxins in Puget Sound (WA State) include: barnacles (*Balanus* spp.), dogwinkles (*Nucella lamellosa* and *N. lima*), hermit crabs (*Pagurus* spp.), kelp crabs (*Pugettia producta*), moon snails (*Polinices lewisii*), Pacific

falsejingles (*Prododesmus cepio*), periwinkles (*Littorina sitkana*), red rock crabs (*Cancer productus*), shore crabs (*Hemigrapsus nudus*, *H. oregonensis*), and tritons (*Fusitron* spp.) (Matter 1994). All of these species also occur in BC waters.

It is recognized that bivalve, gastropod and crustacean shellfish are not the only vectors for STX group toxins (Shumway 1995). For example, cephalopod molluscs also accumulate these toxins by consuming contaminated fish prey. The Humboldt squid (*Dosidicus gigas*), found stranded on the west coast of Vancouver Island (Chesterman Beach, Tofino) in the fall of 2009, contained up to 483 $\mu\text{g STX}_{\text{eq}} 100 \text{ g}^{-1}$ in the digestive gland (Braid et al. 2012); these same animals also contained DA (Section 3.3). This is the first report of STX group toxins in this squid species. The toxin level in the mantle tissue, the part typically consumed by humans, was below the regulatory limit. However, this species is a potential toxin vector for top predators such as tuna, sharks, swordfish, fur seals, and whales.

Although the report series entitled “Summary of marine toxin records in the Pacific Region” is no longer published, the Canadian representative on the ICES-IOC Working Group on Harmful Algal Bloom Dynamics (WGHABD) provides a summary of toxin events in Canada, including their location in HAEDAT maps. For example, it was reported that “2011 was considered to be a ‘normal’ year with respect to shellfish closures as a result of elevated levels of STX group toxins. The highest level observed was 1334 $\mu\text{g STX}_{\text{eq}} 100 \text{ g}^{-1}$, on June 6, in *Mytilus edulis* in the lower Strait of Georgia, with shellfish harvesting areas closed to harvesting from 6 May to 24 October” (ICES 2012). During 2012, the highest shellfish toxicity (5200 $\mu\text{g STX}_{\text{eq}} 100 \text{ g}^{-1}$) was detected at Effingham Inlet, on the Lower west coast of Vancouver Island, in *M. edulis*, on July 17 (ICES 2013).

2.3.2 Species producing STX group toxins in BC

Historically, three species in the dinoflagellate genus *Alexandrium* have been identified as being responsible for STX group toxin production in BC: *A. catenella*, *A. acatenella* and *A. tamarense* (Prakash and Taylor 1966; Gaines and Taylor 1986; Taylor and Harrison 2002). As discussed above (Section 2.2), *A. catenella* and *A. tamarense* are now considered to be the same Group I species, *A. catenella*. The morphologically similar *A. acatenella* is still considered to be a separate species (AlgaeBase; WoRMS). However, there have not yet been genetic studies on this species to confirm its validity. Prakash and Taylor (1966) reported that a human fatality in Theodosia Inlet was coincident with a bloom of toxic *A. acatenella*, although there are no reports of its ability to produce STX group toxins.

In the older literature, *Alexandrium* was called *Protogonyaulax* and *Gonyaulax* (see Dale 1977; Dale et al. 1978; Taylor 1979; Cembella and Taylor 1985, 1986; Cembella et al. 1988a,b; Cembella and Therriault 1989; Gosselin et al. 1989). These species, it has been argued, have different geographic preferences along the coast (Taylor and Harrison 2002). Earlier, there was debate about morphological and genetic similarities between *A. catenella* and *A. tamarense*, which were considered to be in the *A. tamarense* species complex (e.g. Cembella 1986; Anderson et al. 1994; Scholin et al. 1995).

Alexandrium ostenfeldii has only been reported from English Bay, Vancouver (Taylor and Harrison 2002) and Saanich Inlet, Vancouver Island (Kremp et al. 2014) (Section 6.1). However, this species may be missed in phytoplankton samples as it is similar morphologically to *A. catenella*.

Production of STX group toxins by *A. ostenfeldii* from BC has not been confirmed. However, the CFIA has regularly detected SPXs in shellfish harvested in BC (Section [6.1](#)).

2.4 Atlantic coast

The Atlantic coast of Canada ([Fig. 1B](#)) has also been seriously affected by annual STX group toxin events over the years (Murphy 1936; Medcof et al. 1947; Gibbard and Naubert 1948; Prakash et al. 1971; Martin and Richard 1996; Hamer et al. 2012). The first reports of toxicity go back over a century (Ganong 1889; Stafford 1912).

2.4.1 Bay of Fundy

Accounts of sicknesses and death following the consumption of molluscan shellfish date back to 1609, so Indigenous populations were already aware of the problem of PSP in the Bay of Fundy (Prakash et al. 1971). During summer months, they would prefer to eat the bark from trees rather than consume shellfish. Since then, there have been many unrecorded cases of human and animal poisonings in coastal communities. From this local knowledge, fishers learned which types of shellfish were generally safer to eat (Gibbard and Naubert 1948; White 1987; Hamer et al. 2012). Murphy (1936) published the first clinical report of five cases of PSP and two deaths, in July 1936, following the consumption of mussels from St. Mary's Bay, in the Bay of Fundy (southwest NS). As a result of these deaths, and because commercial shellfish processors asked permission to can mussels to meet heavy wartime demands at that time (Prakash et al. 1971), monitoring of molluscan shellfish toxicity began in 1943 (Gibbard and Naubert 1948; van de Riet et al. 2006). This is the longest continuous time series for such monitoring in the world (Martin and Richard 1996).

Although toxicity has been highly variable over the years, the highest toxicity value observed in wild blue mussels (*M. edulis*) in the Bay of Fundy dataset from 1944 to the present (28,000 $\mu\text{g STX}_{\text{eq}}$ 100 g^{-1}) was in September 1944 (Martin and Richard 1996). Consequently, harvesting of these mussels was prohibited, and processors were not given permission to can and sell them. Although Medcof's private notes indicate that he recommended that the mussel industry be suspended in the Bay of Fundy because of this high toxicity, he felt that more study was required and that the issue of harvesting wild mussels be revisited at a later date (J.L. Martin unpubl. data). Unfortunately, although additional research showed that there were periods when blue mussels were safe for consumption (Medcof et al. 1947), the regulatory decision continues to be in effect and the Bay of Fundy remains closed to the harvesting of wild blue mussels today. Cooke Aquaculture Inc., through the Integrated Multi-trophic Aquaculture (IMTA) project, was nevertheless able to get permission to harvest farmed mussels in the past several years (Reid et al. 2011). They are, however, responsible for following regulatory procedures by collecting and submitting samples to the CFIA, and for assuming all costs associated with the toxins testing.

Prohibition Orders for closures due to STX group toxins in the DFO Maritimes Region, including the Bay of Fundy, can be searched on the [DFO Orders Registry](#); search "STN-" (meaning "Scotia Toxins") or "MTN-" (meaning "Maritimes Toxins") in the "Order Number" field. Annual reports detailing harvesting closures and case histories of poisonings due to STX group toxins in the Bay of Fundy for the years 1943–1948 and 1950–1979 are available (Medcof 1949a,b,c,d,e,f, 1952, 1953; Bond 1953, 1954, 1955, 1956, 1958, 1959, 1960; Boyd and Lachance 1961, 1962, 1964, 1965, 1966, 1967, 1968, 1969, 1970; Boyd 1962, 1964, 1972, 1973, 1974, 1975, 1976, 1977, 1978, 1979; Boyd

and Turgeon 1971). As discussed in Section 2.4.1.3, White (1987) examined the shellfish toxicity data from the Bay of Fundy over a 40-year period (1944–1983) in relation to environmental data. Martin and Richard (1996) evaluated the data from a 51-year period (1943–1994) in relation to cycles and trends. Hamer et al. (2012) looked for spatial and temporal trends in STX group toxins in soft-shell clams (*Mya arenaria*) from the Bay of Fundy (1943–2010). However, they cautioned that the CFIA shellfish toxicity data were not collected for research purposes, but for human health and to protect the consumer. Martin et al. (2014a) examined datasets of shellfish toxicity, toxic *Alexandrium catenella* (as *A. fundyense*) bloom dynamics and cyst distributions in the Bay of Fundy since 1980, to look for patterns and trends (Section 2.4.1.3). The presence of STX group toxins has been studied in relation to the potential development of scallop aquaculture in the Annapolis Basin, on the NS side of the Bay of Fundy (Smith and Gaul 1988) and Passamaquoddy Bay, southwest NB. It was determined that once the scallop digestive glands accumulated toxins, they were not able to depurate for extended periods of time (years) (K. Haya, DFO, St. Andrews, NB unpubl. data).

2.4.1.1 Organisms contaminated with STX group toxins in the Bay of Fundy

Various animal species have been contaminated by STX group toxins in the Bay of Fundy (Table 7). Toxicity is found mostly in blue mussels (*Mytilus edulis*), soft-shell clams (*Mya arenaria*), and horse mussels (*Modiolus modiolus*). Rough whelks (*Buccinum undatum*) from St. Andrews (NB) were fed scallop (*Placopecten magellanicus*) hepatopancreas containing 1672 µg STX_{eq} 100 g⁻¹, and they accumulated up to 110 µg STX_{eq} 100 g⁻¹ hepatopancreas (Caddy and Chandler 1968).

Table 7. Species known to become contaminated with saxitoxin (STX) group toxins in coastal waters of the Bay of Fundy. The toxin levels (µg STX_{eq} 100 g⁻¹ ww of tissue) are the highest values found for the reference listed. Levels are for the whole animal, unless indicated otherwise. The closure level is 80 µg STX_{eq} 100 g⁻¹ ww of tissue.

Organism			
Common name	Scientific name	STX level	Reference
Molluscs			
Bar clam	<i>Spisula solidissima</i>	3740	Prakash et al. 1971
Blue mussel	<i>Mytilus edulis</i>	28,000	Prakash et al. 1971
		7977	ICES 2017
Bay quahaug	<i>Mercenaria mercenaria</i>	–	Prakash et al. 1971
Ocean clam	<i>Arctica islandica</i>	–	Prakash et al. 1971
Razor clam	<i>Ensis directus</i>	–	Prakash et al. 1971
Horse (red) mussel	<i>Modiolus modiolus</i>	–	Prakash et al. 1971
Arctic wedge clam	<i>Mesodesma arcium</i>	–	Prakash et al. 1971
Rough whelk	<i>Buccinum undatum</i>	110 ¹	Caddy and Chandler 1968
Stimpson's whelk	<i>Cois stimpsoni</i>	–	Prakash et al. 1971

Organism			
Common name	Scientific name	STX level	Reference
Ten-banded whelk (ten-ridged Neptune)	<i>Neptunea decemcostata</i>	–	Prakash et al. 1971
Northern moonsnail	<i>Lunatia heros</i> (= <i>Euspira heros</i>)	–	Prakash et al. 1971
Atlantic dogwinkle	<i>Nucella lapillus</i>	–	Prakash et al. 1971
Sea scallop	<i>Placopecten magellanicus</i>	1672 ¹	Caddy and Chandler 1968
		50,000 ¹	Hsu et al. 1979
		150,000 ¹	Jamieson and Chandler 1983
		45,000 ¹	Watson-Wright et al. 1989
		6180 ¹	Waiwood et al. 1995
		12,720 ¹	Haya et al. 2003
Soft-shell clam	<i>Mya arenaria</i>	7700	Prakash et al. 1971
Wedge clam	<i>Mesodesma arctatum</i>	–	Prakash et al. 1971
Crustaceans			
American lobster	<i>Homarus americanus</i>	124 ²	Watson-Wright et al. 1991
		3200 ²	Haya et al. 1992
		2150 ²	Haya et al. 1994
		85 ²	Lawrence et al. 1994b
		646 ²	Martin et al. 2006c
		447 ²	Sephton et al. 2007
Fishes			
Atlantic mackerel	<i>Scomber scombrus</i>	209 ³	Haya et al. 1990
Atlantic salmon	<i>Salmo salar</i>	<4 ⁴	Sephton et al. 2007

¹ Hepatopancreas / digestive gland² Hepatopancreatic tissue (tomalley)³ Liver⁴ Stomach, gill, muscle tissue

The complex dynamics of STX group toxins in the Bay of Fundy would require an extensive program with frequent monitoring to assure the safety of the shellfish product for consumers (White 1988). At present, this is not considered cost effective for wild blue mussels (*M. edulis*), and as a result there is a permanent closure of all wild blue mussel harvesting areas in the Bay of Fundy (Boyd 1978) (due also to the variable severity in bacterial contamination). In contrast, monitoring has proven

to be cost effective for cultured blue mussels (Sephton et al. 2003). As well, there is a permanent ban on the harvesting of whelks, as the result of the toxin tetramine (Section 1.2). Several soft-shell clam (*Mya arenaria*) harvesting areas (e.g. Crow Harbour) have been closed year-round since 1980, due to high concentrations of these toxins (Martin et al. 1990b); in the past, Crow Harbour was used as an early warning station because toxicity appeared there 7–10 days before anywhere else (Medcof 1971). In addition, marketing of whole (with roe attached) sea scallops (*Placopecten magellanicus*) from the Bay of Fundy is prohibited at all times (CSSP 2019) because of STX group toxin contamination in the viscera. Harvesting of the adductor muscle is permitted because toxin levels are usually below the detection limit ($<30 \mu\text{g STX}_{\text{eq}} 100 \text{ g}^{-1} \text{ ww}$) of the mouse bioassay (Medcof et al. 1947; Hsu et al. 1979; Haya et al. 2003). However, exceptions were maximum values of $60 \mu\text{g STX}_{\text{eq}} 100 \text{ g}^{-1} \text{ ww}$ in adductor muscles from Digby (November 1977) and $120 \mu\text{g STX}_{\text{eq}} 100 \text{ g}^{-1} \text{ ww}$ in Passamaquoddy Bay (July 1978) (Jamieson and Chandler 1983). Scallops depurate these toxins slowly (Shumway 1990; Haya et al. 2003). For example, toxin levels in the digestive gland and mantle were initially 6180 and 1923 $\mu\text{g STX}_{\text{eq}} 100 \text{ g}^{-1}$, respectively, and decreased to 255 and 787 $\mu\text{g STX}_{\text{eq}} 100 \text{ g}^{-1}$, respectively, which still exceeded the regulatory limit after one year of depuration (Waiwood et al. 1995). Periwinkles (*Littorina littorea*) are toxin-free because they feed only on plants that encrust the rocks, and are marketed regularly (Prakash et al. 1971). As in BC, recreational harvesters (mostly tourists) are most vulnerable to poisonings because they may ignore posted warning signs.

Canadian research has determined that toxic *Alexandrium* blooms affect soft-shell clams (*Mya arenaria*) differently depending on the exposure history of the clams to toxic blooms (Bricelj et al. 1991, 2004, 2005, 2010; MacQuarrie and Bricelj 2008; Phillips et al. 2018). For example, post-larval (shell length 4–12 mm) clams from the Lawrencetown estuary (eastern shore of NS) that had no prior exposure to these toxins exhibited mortality in the presence of toxic *A. catenella* (as *A. tamarense*) cells. Furthermore, the burrowing capacity of these clams was compromised, rendering them more vulnerable to predation. This was in contrast to clams from Lepreau Basin, Bay of Fundy, where toxic *A. catenella* blooms occur annually. The clams had to be large enough to consume the toxic cells in order for the muscular paralysis in the foot or pallial cavity cilia to occur. Resistance in *M. arenaria* is caused by a naturally occurring single-point mutation in the sodium-channel gene, which causes a 1000-fold decrease in affinity for the STX group toxins in the resistant clams (Bricelj et al. 2005). A molecular marker for sodium-channel resistance demonstrated that short-term toxification resulted in a significant shift in the genotypic composition, towards the selection of resistant clams (Bricelj et al. 2010). The 2005 paper in the journal Nature by Bricelj et al. (2005) resulted in the NRC-wide Outstanding Achievement Award for research excellence, as well as the 2003 IMB/NRC Outstanding Achievement Award for initiative and leadership in establishing an excellent shellfish research program at IMB/NRC.

STX group toxins are not confined to molluscan shellfish (Shumway 1995), but have also been detected in the hepatopancreas (tomalley, eaten as a delicacy) of lobsters (*Homarus americanus*) from the Bay of Fundy (Watson-Wright et al. 1991, 1992a). Although negligible levels have been found in the tail muscle ($<0.7 \mu\text{g STX}_{\text{eq}} 100 \text{ g}^{-1} \text{ ww}$; Lawrence et al. 1994b; $3.1\text{--}16.0 \mu\text{g STX}_{\text{eq}} 100 \text{ g}^{-1} \text{ ww}$; Sephton et al. 2007), levels of $646\text{--}3200 \mu\text{g STX}_{\text{eq}} 100 \text{ g}^{-1} \text{ ww}$ were found in the hepatopancreas (Haya et al. 1992, 1994; Lawrence et al. 1994b; Martin et al. 2006c; Sephton et al. 2007) (Table 7); these are above the regulatory limit of $80 \mu\text{g STX}_{\text{eq}} 100 \text{ g}^{-1} \text{ ww}$. Boiling or steaming reduced total toxicity by ~65% in the hepatopancreas compared to values obtained from raw lobsters, by leaching out during cooking (Lawrence et al. 1994b). Lobsters fed toxin-contaminated scallop digestive glands accumulated the toxins in their hepatopancreas (but not in

the tail muscle), and then readily excreted them (Haya et al. 1992, 1994). Lobster tomalley can be exported to several countries around the world, including Japan (CFIA 2014b). There is no Canadian regulatory level for STX group toxins in lobster tomalley, although Health Canada (2009) recommends that children not eat lobster tomalley, and that adults restrict their consumption of lobster tomalley to no more than the amount from one cooked lobster per day. The recommendation was prompted by a preliminary survey of ~900 individual lobsters harvested in November and December 2008, from 20 locations along coastal NB and NS, and tested for the presence of STX group toxins in lobster hepatopancreas (Lavallée and Revie 2009). Five of the 20 samples had an average toxin level above $100 \mu\text{g STX}_{\text{eq}} 100 \text{ g}^{-1}$ in hepatopancreas; several individual lobsters from the Bay of Fundy had much higher levels. A similar advisory was issued by the U.S. Food and Drug Administration (FDA) for Maine lobsters in 2008 ([MedPage Today](#)).

Finfish are particularly sensitive to STX group toxins and this can have significant effects throughout the food web in Canadian waters. Mass kills of wild adult Atlantic herring (*Clupea harengus harengus*) (White 1977, 1980, 1984) and mortality or impairment of larval and juvenile stages of fish (White 1981b; White et al. 1989) have resulted from STX group toxins in the Bay of Fundy since the late 1970s. These toxins were detected in the fish stomachs and in zooplankton vectors (White 1981a), including pteropods (*Limacina retroversa*) (White 1977) and the cladoceran *Evadne nordmanni* (White 1980). Other fish, including American pollock (*Pollachius virens*), winter flounder (*Pseudopleuronectes americanus*), Atlantic salmon (*Salmo salar*), and cod (*Gadus morhua*) were all killed by low doses of STX group toxins injected intraperitoneally (White 1981b).

Farming of Atlantic salmon (*Salmo salar*) in inshore waters was started in 1978, and expanded at one point to >90 farms, many of which are located in coastal Grand Manan Island (Martin et al. 2006b). In September 2003, abnormal swimming behaviour and elevated mortalities of farmed salmon were found near Grand Manan Island, when *Alexandrium catenella* (as *A. fundyense*) concentrations reached $3.0\text{--}8.8 \times 10^5 \text{ cells L}^{-1}$ (Martin et al. 2006b). A follow-up study detected only low levels of toxins ($<4 \mu\text{g STX}_{\text{eq}} 100 \text{ g}^{-1} \text{ ww}$; [Table 7](#)) in stomach, gill and muscle tissues of moribund salmon, although the authors suggested that these toxins are very lethal to salmon (Sephton et al. 2007). In 2004, further salmon mortalities were associated with a bloom of *A. catenella* that persisted along the mainland coast of NB between Letang and Lepreau. Concentrations of 4.0×10^5 to $3.0 \times 10^6 \text{ cells L}^{-1}$ were measured and water discolouration was observed (ICES 2004; Martin et al. 2006b; Chang et al. 2007b). Laboratory studies have since shown that live *A. catenella* cells can cause the mortality of Atlantic salmon smolts from the Bay of Fundy (Burridge et al. 2010). Lethal cell concentrations were similar to those found in field situations, and smolt death was rapid; the 6 h LC_{50} was $\sim 7.2 \times 10^5 \text{ cells L}^{-1}$. As in previous studies of other finfish injected with STX group toxins (White 1980, 1981b), affected salmon lost their equilibrium, swam erratically, and exhibited shallow arrhythmic breathing, before becoming immobilized and dying. Approaches for mitigating the problem of STX group toxins at salmon farms in the Bay of Fundy include: using cell counts to provide an early warning of impending *A. catenella* blooms (Chang et al. 2005, 2006, 2007b), water circulation models to predict the movements of these blooms (Chang et al. 2007a), and the application of remote sensing data to locate the blooms (Harrison et al. 2007; Devred et al. 2018).

Atlantic mackerel (*Scomber scombrus*) may be less sensitive to STX group toxins, as elevated levels of these toxins can accumulate in the livers of live fish ([Table 7](#)). Human health may thus be affected if mackerel, containing sub-lethal levels of toxin in their viscera, are eaten whole. Mackerel in the Bay of Fundy accumulate these toxins by feeding on contaminated herring (Haya et al. 1990).

They may thus be a vector for the further transfer of the toxins through the food web (Geraci et al. 1989). Zooplankton are another vector (White 1977, 1979, 1980, 1981a, 1982b; Hayashi et al. 1982; Doucette et al. 2006; Sephton et al. 2007). Indeed, many zooplankton, like the calanoid copepod *Calanus finmarchicus*, can ingest toxic *A. catenella* cells without any obvious effects on their survival or grazing activity (Turner and Borkman 2004). However, significant reductions of egg production and egg viability were observed in *C. finmarchicus* females from the Gulf of Maine after they fed on toxic *A. catenella* cells for seven days (Roncalli et al. 2016). Thus, toxic *Alexandrium* cells can disrupt pelagic communities by having a negative effect on copepod recruitment.

STX group toxins were also found in fecal samples of the North Atlantic right whale (*Eubalaena glacialis*), a highly endangered cetacean, in their summer feeding grounds of the Bay of Fundy (Doucette et al. 2006). The most likely vector for these toxins is the copepod *Calanus finmarchicus*, the whale's main food source. The right whale is chronically exposed to toxic *A. catenella* blooms during several months each summer while feeding in the Grand Manan Basin (Durbin et al. 2002). STX group toxins could affect respiratory capabilities, feeding behaviour, and ultimately the reproductive condition of the whale population. This provides direct evidence that these toxins can pose a threat to the North Atlantic right whale population, and could be a contributing factor to the reproductive dysfunction and compromised health of these whales during the past 25 years.

Farther south, 14 humpback whales (*Megaptera novaeangliae*) died in Cape Cod Bay after eating Atlantic mackerel (*Scomber scombrus*) that contained STX group toxins, during a 5-week period beginning in late November 1987 (Geraci et al. 1989). Because STX group toxins were absent in New England waters during that episode, it was proposed that the mackerel accumulated the toxin while spawning in the Gulf of St. Lawrence. They then delivered it to the Gulf of Maine and Cape Cod Bay, where the whales consumed the contaminated fish. This again demonstrates the vulnerability of large marine mammals to STX group toxins and also shows that toxins may move through the food chain and affect mammals far from the origin of the toxins.

2.4.1.2 Species producing STX group toxins in the Bay of Fundy

Alexandrium catenella (formerly *A. fundyense*) is the primary species that produces STX group toxins in the Bay of Fundy (Needler 1949; Prakash 1963; Prakash et al. 1971; Martin and Richard 1996; Martin et al. 1998, 2008b, 2009; Sephton et al. 2007). It was not until the late 1940s that this dinoflagellate was proposed as the source organism in the Bay of Fundy, based on the coincidence of the rise in shellfish toxicity and its cell abundance in the water column (Medcof et al. 1947; Needler 1949). In the older literature, this species was reported as *Gonyaulax tamarensis* (Needler 1949; Prakash 1963, 1967), *Protogonyaulax tamarensis*, *Gonyaulax tamarens* var. *excavata*, *Gonyaulax excavata* (White and Maranda 1978; White 1979, 1980, 1982a, 1986; White and Lewis 1982; Martin and White 1988), or *A. fundyense* (Needler 1949; Prakash 1963; Prakash et al. 1971; Balech 1985; Martin and Richard 1996; Martin et al. 1998, 2006b, 2008b, 2009, 2014a,c; Page et al. 2006; Sephton et al. 2007). New microsatellite markers have been developed to detect *A. catenella* in the Bay of Fundy and Gulf of Maine (Sehein et al. 2016). *Alexandrium ostenfeldii* has also been detected in the Bay of Fundy (Martin et al. 1999, 2001b, 2006a, 2014b,c; Martin and LeGresley 2014), as well as in the adjacent Gulf of Maine (Anderson et al. 2005a) and tends to co-occur with *A. catenella*. Gribble et al. (2005) tested strains of *A. ostenfeldii* from the Bay of Fundy and found that they do not produce STX group toxins (Section [2.4.2.2](#)), but they do produce spirolides (Section [6.1](#)).

The data on STX group toxin concentrations in shellfish collected from the Bay of Fundy since 1943 provide an important perspective on the inter-annual and seasonal patterns of *A. catenella*. They indicate that the toxins have been present in shellfish throughout much of the Bay of Fundy, particularly the lower Bay, since the early 1940s. Historical records indicate that *A. catenella* and resulting STX group toxins have been an annual occurrence for many years (Gran and Braarud 1935; Needler 1949; Prakash et al. 1971; Martin and White 1988; J.L. Martin unpubl. data). A record high concentration of *A. catenella* (20×10^6 cells L⁻¹) was observed in August 1980, accompanied by water discolouration (Martin and White 1988). As well, Indigenous traditional knowledge prohibiting the consumption of shellfish during certain times of the year indicates a long-term seasonal pattern in *A. catenella* abundance.

Broad surveys were conducted for *A. catenella* (as *A. fundyense*) distribution and abundance in the Bay of Fundy during 1980–1984 (Martin and White 1988), complemented by sediment surveys for resting cysts (White and Lewis 1982; Martin and Wildish 1994; Martin et al. 2014a). Otherwise, *A. catenella* concentrations were not measured continuously until a phytoplankton program was initiated in 1987 (Section 9.2.2.3). Since then, *A. catenella* has indeed been observed annually in the Bay of Fundy. It first appears in offshore and coastal waters (by mid-May), and then in the sheltered inshore areas (late May) (Page et al. 2006; Martin et al. 2009). However, the maximum cell abundance generally occurs earlier (mid-July) for inshore than for offshore waters (mid-August). Furthermore, inshore blooms have had a lower cell concentration than offshore blooms in all years except 2009 (Martin et al. 2014a,c; McGillicuddy et al. 2014), perhaps because of greater mixing (Page et al. 2004, 2006). Blooms last from 42 to 205 days, with a mean of 114 days (Martin et al. 2009). Because nutrient concentrations appear not to be limiting, the magnitude and timing of *A. catenella* blooms may be linked more to conditions associated with fog (Martin et al. 2014a), but not with prolonged sunshine (as experienced in 2012) or extended rain (as experienced in 2014) (Martin et al. 2014c). For example, cold temperatures in April corresponded to the highest cell densities during the summer (Martin et al. 2009). Interestingly, in contrast to *A. catenella* (as *A. tamarensis*) cells in the St. Lawrence Estuary (Cembella and Therriault 1989; Laroque and Cembella 1990), cells in the Bay of Fundy do not migrate vertically in the water column (Martin et al. 2004a, 2005). Therefore, they are found at higher concentrations in the surface waters over a 24-h period. This may result in a greater transport of cells from the offshore to inshore waters, where shellfish become toxified. Farmed salmon tend to concentrate at depth when surface concentrations of *A. catenella* are high, and growers do not feed them when cell concentrations exceed 2.0×10^5 cells L⁻¹, to prevent the fish from moving to the surface waters (Martin et al. 2008b). The bacterium *Alteromonas* sp. has been reported to stimulate the growth of *A. catenella* in the Bay of Fundy (Ferrier et al. 2002), although this study did not determine if it influenced STX group toxin production.

It can be difficult to study factors affecting *A. catenella* bloom dynamics because as few as 200 cells L⁻¹ can result in shellfish toxicity (Page et al. 2004). Since *A. catenella* is often a minor component of the total phytoplankton community, the Bay of Fundy phytoplankton monitoring program (Section 9.2.2.3) has included identification and enumeration of the total phytoplankton community (Martin et al. 2014c). Page et al. (2004) examined the spatial and temporal variation in abundance of *A. catenella* in relation to the total phytoplankton community structure for data collected in 1991, and listed the dominant phytoplankton species (>5000 cells L⁻¹). Another dataset collected during 1987–2013 lists 14 co-occurring species that exceeded 10^4 cells L⁻¹ (Martin et al. 2014c). Major co-existing organisms (e.g. *Ditylum brightwellii*, *Heterocapsa triquetra*, *Guinardia*

delicatula, *Eutreptiella* spp., *Scrippsiella trochoidea*, *Mesodinium rubrum*) varied between years and locations (Martin and LeGresley 2012).

The distribution, dispersal and retention of toxic *A. catenella* vegetative cells (Martin and White 1988), their benthic cysts (White and Lewis 1982; Martin and Wildish 1994; Martin et al. 2014a), and therefore the distribution of contaminated molluscan shellfish, are dependent on the winds, currents, and other oceanographic patterns within the Bay of Fundy. A counterclockwise gyre is superimposed on the enormous tidal variations. *Alexandrium* cysts tend to sediment in the clay, where the currents decrease near the central part of the gyre. Thus, the distribution of cysts is greatest in an area northeast of Grand Manan Island (Martin and Wildish 1994; Martin et al. 2014a), which is believed to serve as a “seed bed”, i.e. the primary source of motile cells that initiate the annual summer blooms. The circulation patterns also tend to retain the cysts and vegetative cells, although some cysts are thought to exit the Bay of Fundy to initiate blooms in the Gulf of Maine (Anderson et al. 2005b). There is thus a strong linkage between the Bay of Fundy cyst bed and subsequent re-seeding in the Gulf of Maine (Anderson et al. 2013). However, in the Bay of Fundy, *A. catenella* cyst numbers were not a good predictor of bloom intensity for the summers that followed (Martin et al. 2014a). This is in contrast to studies in the Gulf of Maine, where cyst numbers have been suggested to predict subsequent blooms in some areas (Anderson et al. 2005b). This did not occur following the prediction of a large regional bloom for 2010, despite the large bloom of *A. catenella* along the coast of Maine in 2009 that yielded the highest cyst numbers ever measured in that region in the fall of 2009 (McGillicuddy et al. 2011, 2014). Thus, cyst numbers are consistently high in the Bay of Fundy and do not appear to influence subsequent cell densities or blooms (Martin et al. 2014a).

Nevertheless, the presence of high concentrations of toxic cysts may provide a mechanism by which molluscan shellfish can remain toxic during the winter, when *Alexandrium* does not bloom (White and Lewis 1982), as was demonstrated in the laboratory (Persson et al. 2006). However in the field, Haya et al. (2003) found no statistical differences in toxin uptake or depuration by sea scallops (*Placopecten magellanicus*) held at the surface compared to those at the bottom. This suggests that *A. catenella* cysts did not contribute to its toxicity; rather, the persistent toxicity could be due to the slow depuration by this scallop (Haya et al. 2003).

Another example is the persistent levels of STX group toxins in soft-shell clams (*Mya arenaria*) from Crow Harbour, a location that has been closed to harvesting clams due to their unacceptable levels of STX group toxins year-round since 1980. Sediments at Crow Harbour were examined for cysts at various times during several years. The consistently low numbers of cysts suggests that they may not be responsible for the persistent toxicity (White and Lewis 1982). A study of the anatomical distribution of their toxins (Martin et al. 1990b) suggested that the clams retain the toxins in tissues other than the digestive gland because the toxins may be metabolically transformed by bacteria, the clams, or the cells themselves. Thus, the apparent ability of clams to detoxify from different tissues and selectively retain high concentrations of toxins may explain their toxicity during non-bloom periods. At other locations, during some years, the persistent low levels of toxins may also be related to blooms that occur in September and October, resulting in less opportunity for clams to depurate the toxins as temperatures decrease.

On Georges Bank, the origin of STX group toxins in scallops has not yet been identified conclusively (Cembella et al. 1993). Blooms of *A. catenella* are found in the surface waters (Deeds et al. 2014), but not at the depths of the scallops (32–91 m), unless the cells sink to the bottom intact. It

is possible that their toxic cysts are ingested by the scallops (Bourne 1965), but the number of cysts may be too low to cause the observed toxicity.

2.4.1.3 Periodicity in shellfish toxicity in the Bay of Fundy

Shellfish toxicity data are collected to protect human health, and not for research purposes. Therefore, there has been much temporal variability in sampling effort and locations, leading to large data gaps (Hamer et al. 2012; Martin et al. 2014a). As well, once the maximum level of $80 \mu\text{g STX}_{\text{eq}} 100 \text{ g}^{-1}$ ww of tissue is reached, harvesting is prohibited, and sampling frequency may decrease. Thus, the maximum toxin levels in the shellfish may not be quantified. These limitations make it difficult to carry out rigorous statistical analyses of the data or to look for long-term trends. Nevertheless, various datasets of the STX group toxin monitoring program have been analyzed since it was initiated in 1943 (White 1987; Messieh and El-Sabh 1990; Martin and Richard 1996; Martin et al. 2004b, 2010a, 2014a; ICES 2005; Hamer et al. 2012), as discussed below.

Shellfish toxicity is an annual event in the Bay of Fundy, with most peaks in the summer months (June to September). Ideally, harvesting should occur between October and May to ensure lower toxin concentrations, although high toxin levels and closures may still occur during that period. Analyses of the CFIA shellfish toxicity data (1943–2010) showed that sites with the highest percent duration of summer closures were Crow Harbour (83%), Lepreau Basin (71%) and Ross Island Throughfare (56%) (Hamer et al. 2012). However, as indicated above, Crow Harbour has been closed to harvesting continuously (100%) since 1980.

Patterns for shellfish toxicity tend to follow those for *Alexandrium catenella* cell abundance (White 1987; Martin and Richard 1996); higher values were detected in the mid-1940s, the early 1960s, late 1970s, and again in 2003–2004. Periods of higher toxicity occurred in blue mussels (*M. edulis*), soft-shell clams (*Mya arenaria*) and horse mussels (*Modiolus modiolus*) in 1943–1945, 1957–1961, 1970–1986, followed by a “mixed period” of toxicity from 1993–2010, with the highest toxicities in 2008 (White 1982a; Martin et al. 2014a), suggesting a periodicity. These earlier data suggested a linkage with an 18.6-year cycle corresponding to the change in the inclination of the moon to the equator. White (1987) concluded that this results in a lunar modulation of the tides, which would marginally increase tidal mixing, but enough to favour the growth of toxic dinoflagellates or advect toxic blooms into shellfish harvesting areas. An analysis of the data, however, has put into question the statistical validity of the 18.6-year lunar tidal cycle (Martin and Richard 1996). The analysis also concluded that the toxic blooms have not grown in intensity. The shellfish toxicity data of White (1987) over a 40-year period (1944–1983) showed some correlations between summer shellfish toxicity and environmental data during pre-bloom months. Using historical data, Messieh and El-Sabh (1990) calculated a mean interval value between PSP episodes of 5.1 years in the Bay of Fundy, with a mean interval between major episodes of 10 years, and noted a correspondence with a 10.8 year sunspot cycle. Year-to-year variations in toxicity recorded from 1944 to 1958 were directly correlated with sunshine duration and salinity in the spring, and inversely correlated with springtime river discharge (Prakash and Medcof 1962). It will be interesting to see if these patterns persist with climate change.

A major intensification of toxicity occurred in the mid- to late-1970s (White 1982a). Based on the analyses of White (1987), the next peak in toxicity was anticipated to be in the late 1990s to early 2000s, but this did not appear. Wyatt (2018) suggested that the nodal signal may be only a part of the

story. Although the period between 1996 and 2000 did experience shellfish toxicities, concentrations of *A. catenella* were low ($< 5000 \text{ cells L}^{-1}$) compared to most years (Martin et al. 2014c). During and after these years of low cell abundance, shellfish toxicity continued annually. For example, there were extensive shellfish closures and mortalities of farmed salmon in 2003 and 2004 (Martin et al. 2006c; Section 2.4.1.1). In 2009, a large bloom of *A. catenella* occurred along the entire coast of Maine and into Passamaquoddy Bay (NB). In most years, shellfish become toxic in Passamaquoddy Bay, although usually at low levels. Occasionally, toxin levels have not increased above the threshold level, allowing shellfish beds to remain open to harvesting. However, 2009 was the first year since the CFIA dataset was initiated that soft-shell clam (*Mya arenaria*) toxicity values exceeded $1000 \mu\text{g STX}_{\text{eq}} 100 \text{ g}^{-1}$ (up to $4120 \mu\text{g STX}_{\text{eq}} 100 \text{ g}^{-1}$). The following year (2010), harvesting was closed outside of Passamaquoddy Bay, when *Mya arenaria* reached $4920 \mu\text{g STX}_{\text{eq}} 100 \text{ g}^{-1}$ (ICES 2011). Another closure occurred in 2011, when mussels (*Mytilus edulis*) reached $3640 \mu\text{g STX}_{\text{eq}} 100 \text{ g}^{-1}$ at Bocabec Bay (in Passamaquoddy Bay), where the *A. catenella* concentration was $1.0 \times 10^5 \text{ cells L}^{-1}$ (ICES 2012). Shellfish beds were closed throughout the year at Deadmans Harbour due to unacceptable levels of STX group toxins ($850 \mu\text{g STX}_{\text{eq}} 100 \text{ g}^{-1}$) during the summer and to low levels during the winter (ICES 2012).

In spite of the great cost and effort required, and the statistical problems related to samples being collected for health protection rather than research purposes, the above examples show the advantage of maintaining long-term databases. Trends can be discerned from the data (White 1987; Martin and Richard 1996; Hamer et al. 2012; Martin et al. 2014a), which will eventually help to discriminate between natural and anthropogenic causes of toxic blooms. For example, it has been determined that *A. catenella* cell abundances appear to be related more to climate or weather than to nutrient flux (Martin et al. 2009, 2014a). Furthermore, examination of relationships between cell density, nutrients and environmental variables indicates there is no evidence that the blooms are linked to eutrophication processes. Nor is there evidence from the Bay of Fundy dataset to show that HABs are increasing over time, even since the advent of aquaculture (Martin and LeGresley 1998). The data analyses show only that there are cyclic periods of higher toxicities that extend back more than one hundred years, corresponding to Indigenous traditional knowledge (Martin et al. 2009).

2.4.2 Nova Scotia – Atlantic coast

2.4.2.1 Organisms contaminated with STX group toxins in NS

After monitoring efforts were expanded in 1975, STX group toxins were found for the first time in east-central NS, in 1992 (Cembella and Todd 1993). Levels up to $67,000 \mu\text{g STX}_{\text{eq}} 100 \text{ g}^{-1}$ tissue were found in wild mussels (*M. edulis*) in June 2000, near Shelburne, on the southwestern shore of NS (Cembella et al. 2002; HAEDAT [CA-00-009](#)) ([Table 8](#)). This finding was associated with the mortality of farmed salmon (*Salmo salar*) at the same site (Cembella et al. 2002; HAEDAT [CA-00-009](#)). Although no toxins were detected in the liver and digestive tract of the fish, the gills contained low levels ($5\text{--}10 \mu\text{g STX}_{\text{eq}} 100 \text{ g}^{-1}$ tissue). The mortality coincided with a massive bloom ($>7 \times 10^5 \text{ cells L}^{-1}$) of *Alexandrium catenella* (as *A. tamarense*), whose cells contained STX group toxins. This provided evidence that the mortalities were caused by exposure to the toxic *Alexandrium* cells and/or to soluble toxins released during the bloom. For example, the mortality of farmed rainbow trout (*Oncorhynchus mykiss*) in Chile was thought to be attributed not to STX group toxins, but rather to the synergistic interaction between reactive oxygen species (superoxide anion) and certain polyunsaturated fatty acids produced by *A. catenella* (Mardones et al. 2015).

Table 8. Species known to become contaminated with saxitoxin (STX) group toxins in Atlantic waters of Nova Scotia. The toxin levels ($\mu\text{g STX}_{\text{eq}} 100 \text{ g}^{-1}$ in edible tissue) are the highest values reported in the reference listed. Levels are for the whole animal, unless indicated otherwise. The closure level is $80 \mu\text{g STX}_{\text{eq}} 100 \text{ g}^{-1}$ in edible tissue. HAEDAT = Harmful Algae Event Database (<http://haedat.iode.org/>)

Organism		STX level	HAEDAT event	Location	Reference
Common name	Scientific name				
Molluscs					
Blue mussel	<i>Mytilus edulis</i>	67,000	CA-00-009	Shelburne, NS	Cembella et al. 2002
		1500	CA-11-003	Mahone Bay, NS	ICES 2012
		363	CA-13-003	Cranes Point, NS	ICES 2014
Soft-shell clam	<i>Mya arenaria</i>	681	CA-13-002	Sambro, NS	ICES 2014
Sea scallop (farmed)	<i>Placopecten magellanicus</i>	196 ¹	–	Inshore sites, NS	Hancock 1994
		46 ²	–	Inshore sites, NS	Hancock 1994
Sea scallop (wild)	<i>Placopecten magellanicus</i>	1440 ¹	–	Georges Bank	Gillis et al. 1991
		7500 ¹	–	Georges Bank	White et al. 1993
Fishes					
Atlantic salmon	<i>Salmo salar</i>	5–10 ³	CA-00-009	Shelburne, NS	Cembella et al. 2002

¹ Hepatopancreas / digestive gland

² Whole animal

³ Gill tissue

Mussels (*M. edulis*) from Mahone Bay (NS) reached $1500 \mu\text{g STX}_{\text{eq}} 100 \text{ g}^{-1}$ ([Table 8](#)) and that area was closed from June 10 to July 13, 2011 (ICES 2012; HAEDAT [CA-11-003](#)). Scallops (*Placopecten magellanicus*), from Snake Island, in Mahone Bay, caused a precautionary closure when they reached $77 \mu\text{g STX}_{\text{eq}} 100 \text{ g}^{-1}$ on June 28, 2017 (CFIA data). A closure was imposed on Snake Island mussels (*M. edulis*), when levels reached $150 \mu\text{g STX}_{\text{eq}} 100 \text{ g}^{-1}$ on June 20, 2018. The peak level in scallops was $55 \mu\text{g STX}_{\text{eq}} 100 \text{ g}^{-1}$ on June 22, 2018. At nearby Indian Point, the level in mussels was $59 \mu\text{g STX}_{\text{eq}} 100 \text{ g}^{-1}$ and $58 \mu\text{g STX}_{\text{eq}} 100 \text{ g}^{-1}$ in scallops on June 21, 2018.

Soft-shell clams (*Mya arenaria*) from Sambro (St. Margarets Bay, NS) attained a maximum of $681 \mu\text{g STX}_{\text{eq}} 100 \text{ g}^{-1}$ on June 27, 2013, and harvesting was closed from June 13 to July 16 (ICES 2014; HAEDAT [CA-13-002](#)). On the southwestern shore of NS (Cranes Point), *M. edulis* reached

363 $\mu\text{g STX}_{\text{eq}} 100 \text{ g}^{-1}$ on June 3, 2013, and the closure lasted until June 17, 2013 (ICES 2014; HAEDAT [CA-13-003](#)) ([Table 8](#)). *Mytilus edulis* in Halifax harbour may also attain high levels of STX group toxins (D. Richard, DFO, Moncton, NB, 2001 pers. comm.). Although this area is permanently closed due to elevated fecal coliform counts, recreational mussel harvesters have nevertheless become ill, after ignoring warning signs when shellfish harvesting is banned.

Other bivalve species have also been affected. Farmed sea scallops (*Placopecten magellanicus*) at inshore sites in NS contained low ($46 \mu\text{g STX}_{\text{eq}} 100 \text{ g}^{-1}$ in the whole animal; $196 \mu\text{g STX}_{\text{eq}} 100 \text{ g}^{-1}$ in the digestive gland; [Table 8](#)) to negligible levels of STX group toxins (Hancock 1994). Offshore, toxins have been found in the digestive glands of sea scallops from the Canadian portion of Georges Bank (Bourne 1965; Jamieson and Chandler 1983; Gillis et al. 1991; Martin et al. 1992; Shumway and Cembella 1992, 1993; Cembella et al. 1993; White et al. 1993). Monitoring of the Canadian fishery for roe-on (gonad attached) sea scallops found toxin levels exceeding the regulatory limit in the scallop roe (Watson-Wright et al. 1993b). This resulted in the closure of the Canadian sector of Georges Bank; the peak hepatopancreas level was $1440 \mu\text{g STX}_{\text{eq}} 100 \text{ g}^{-1}$ (Gillis et al. 1991). Hepatopancreas toxin levels were $7500 \mu\text{g STX}_{\text{eq}} 100 \text{ g}^{-1}$ in mid-July 1989 (White et al. 1993). The scallop fishery was thus closed during the 1989–1990 season. Canada can now harvest whole and roe-on scallops from Georges Bank (as well as from the Gulf of St. Lawrence and the Northumberland Strait, but not the Bay of Fundy), but they must be tested for toxin content before being released to market (CSSP 2019). Note that for the purpose of biotoxin testing, the CFIA now interprets edible tissue in scallops as being the form in which the scallops are marketed to consumers ([CFIA 2019](#)).

In contrast, on the U.S. portion of Georges Bank, only the adductor muscle of sea scallops can be harvested because it does not accumulate STX group toxins (White et al. 1993); the remaining tissues must be discarded at sea (DeGrasse et al. 2014a,b). Given that roe-on scallops are popular in the European Union (EU) and that whole scallops are consumed in Asian countries, a subsequent study was carried out to re-address the potential of a roe-on and whole-scallop fishery for the U.S. portion of Georges Bank (DeGrasse et al. 2014a). Results confirmed that sea scallop viscera toxicity was often above the regulatory limit, leading to the conclusion that the feasibility of a roe-on scallop fishery in this region was uncertain. However, U.S. fishermen can now harvest bar clams (*Spisula solidissima*) and ocean quahogs (*Arctica islandica*) in a newly reopened portion of Georges Bank, provided they follow an onboard screening protocol for STX group toxins (DeGrasse et al. 2014b).

An Enhanced Shellfish Monitoring Program, carried out by DFO's Inspection Services Branch in 1990, determined that levels of STX group toxins were below limit of quantitation in the meat and stomach of lobsters, and levels below the quarantine limit in the hepatopancreas, from sites around NS (Watson-Wright et al. 1991). Likewise, lobsters caught in 1991, off the east coast of NS, Georges Bank and Browns Bank, contained low or non-detectable levels of toxins, including in the hepatopancreas (Todd et al. 1993).

Despite sometimes elevated levels of STX group toxins in NS waters, public health warnings and harvesting closures have prevented serious poisonings. Between 1997 and 2006, there were only 11 reported poisonings due to these toxins (in 1998, 2001 and 2005) (Nova Scotia Health Promotion and Protection 2008), giving an incidence of less than one case per 100,000 population.

2.4.2.2 Species producing STX group toxins on the Atlantic coast of Nova Scotia

The production of STX group toxins on the Atlantic coast has been associated with species in the *Alexandrium tamarense* complex, now designated as *A. catenella* (formerly *A. tamarense*; e.g. Cembella et al. 1998, 2000, 2002). *Alexandrium ostenfeldii* was first observed on the Atlantic coast of NS, in Ship Harbour, where it was associated with SPXs production (Cembella et al. 1998; Section 6.1). Certain strains of *A. ostenfeldii* from European and New Zealand waters are known to produce STX group toxins (see Levasseur et al. 1998; Cembella et al. 2000; Maclean et al. 2003; Salgado et al. 2015; Martens et al. 2017). However, these were not detectable in isolates from eastern Canada (Cembella et al. 2000) and the Gulf of Maine (Gribble et al. 2005). Recently, trace amounts of STX group toxins (0.002–2.4 fmol cell⁻¹) were detected in all *A. ostenfeldii* strains from the south shore of NS (Shelburne, Graves Shoal, Halifax harbour, Ship Harbour), including C1/2, GTX1/4, GTX2/3 and GTX5 (Qiu et al. 2018). One isolate exhibited a unique profile, containing much higher levels of NEO and STX. Indeed, the *sxtA4* domain of the *sxtA* gene, a critical component of the STX biosynthesis pathway, is present in *A. ostenfeldii*, as it is in all other toxigenic *Alexandrium* species (reported in John et al. 2014). *Alexandrium ostenfeldii* is the source of spirolide toxins (Section 6.1), as well as allelopathic compounds against heterotrophic and autotrophic protists (Tillmann et al. 2007).

2.4.3 Gulf of St. Lawrence

A shellfish monitoring program was initiated on the south shore of the St. Lawrence Estuary in 1949, after two fatalities in 1948 (Prakash et al. 1971). Historical evidence, however, suggests the presence of sickness likely caused by STX group toxins going back to 1807 (Prakash et al. 1971). Tennant et al. (1955) describe an incident of PSP and deaths, after mussels were consumed at Métis-sur-Mer, near Mont-Joli (QC) (Fig. 1B), in July 1954. Medcof et al. (1966) surveyed the incidences and risks of PSP in QC prior to 1966. Up to 1970, Medcof (1971) documented 16 cases (one fatal) of poisoning due to the unregulated domestic consumption of rough whelks (*Buccinum undatum*) from the St. Lawrence Estuary (Table 9). An epidemiological surveillance program for the sickness/deaths caused by shellfish poisoning was implemented in QC in 1999 (Duchesne 2002; Duchesne et al. 2002). The objectives were to: 1) estimate the prevalence of poisonings in eastern QC; 2) describe the cases of disease related to toxic shellfish; 3) make health professionals aware of the problem of toxic shellfish; and 4) prevent secondary cases of shellfish-related diseases. As of March 2002, 230 cases of PSP had been identified in QC. There were 15 confirmed episodes of PSP between 1984 and 1998, or an average of one episode per year. Another “confirmed” episode of PSP in July 2000, at Sept-Îles, affecting nine people who had eaten mussels from a culture zone (not indicated if the area was open or closed). Analyses revealed the presence of 2900 µg STX_{eq} 100 g⁻¹ in the flesh of raw mussels. Concentrations of STX group toxins in the blood of three people involved were above the limit of detection (Duchesne et al. 2002). The discussion below is divided between toxicity patterns in the northern (including the estuary) and southern Gulf of St. Lawrence.

2.4.3.1 Organisms contaminated with STX group toxins in the northern Gulf of St. Lawrence

In the northern Gulf of St. Lawrence (Fig. 1A), STX group toxins are found primarily in blue mussels (*Mytilus edulis*) and soft-shell clams (*Mya arenaria*) (Table 9) along the north and south shores of the lower estuary, the Gaspé Peninsula, and in Chaleur Bay (Cembella et al. 1988b; Desbiens et al. 1990; Larocque and Cembella 1990, 1991; Huppertz and Levasseur 1993; Blasco et al. 1998, 2003). *Mytilus edulis* accumulates toxins more rapidly than *Mya arenaria* (Blasco et al. 2003).

Table 9. Species known to become contaminated with saxitoxin (STX) group toxins in coastal waters of the Gulf of St. Lawrence. The whole-animal toxin levels ($\mu\text{g STX}_{\text{eq}} 100 \text{ g}^{-1}$) are the highest values reported in the reference listed. Levels are for the whole animal, unless indicated otherwise. The closure level is $80 \mu\text{g STX}_{\text{eq}} 100 \text{ g}^{-1}$ in edible tissue. HAEDAT information is not included because most of the literature reported is older than the start of that database.

Organism			
Common name	Scientific name	STX level	Reference
Molluscs			
American oyster	<i>Crassostrea virginica</i>	214	Worms et al. 1993
Bar clam	<i>Spisula solidissima</i>	1011	Worms et al. 1993
Bay quahaug	<i>Mercenaria mercenaria</i>	56	Worms et al. 1993
Blue mussel	<i>Mytilus edulis</i>	23,000	Prakash et al. 1971
Horse (red) mussel	<i>Modiolus modiolus</i>	2357	Sephton et al. 2007
Northern moonsnail	<i>Lunatia heros</i> (= <i>Euspira heros</i>)	333	Worms et al. 1993
Rough (common) whelk	<i>Buccinum undatum</i>	600	Prakash et al. 1971
		608	Medcof 1972
		1600 ¹	Medcof 1972
Sea scallop	<i>Placopecten magellanicus</i>	1050 ¹	Worms et al. 1993
Soft-shell clam	<i>Mya arenaria</i>	6600	Prakash et al. 1971
Ten-banded whelk	<i>Neptunea decemcostatata</i>	3000	Prakash et al. 1971
Crustaceans			
American lobster	<i>Homarus americanus</i>	167 ²	Lawrence et al. 1994b
		1550 ²	Cembella and Desbiens 1994
		1512 ²	Desbiens and Cembella 1995
		69 ³	Desbiens and Cembella 1995
Amphipod	<i>Anonyx sarsi</i>	180	Desbiens and Cembella 1995
Fishes			
Atlantic mackerel	<i>Scomber scombrus</i>	367 ⁴	Castonguay et al. 1997

¹ Hepatopancreas / digestive gland

² Hepatopancreatic tissue (tomalley)

³ Claws plus tail muscle

⁴ Liver

As in the Bay of Fundy, these toxins have also been found in the hepatopancreas of lobster (*Homarus americanus*) from the Gaspé Peninsula (Todd et al. 1993; Cembella and Desbiens 1994;

Lawrence et al. 1994b; Desbiens and Cembella 1995). Values ranged from 722 $\mu\text{g STX}_{\text{eq}} 100 \text{ g}^{-1}$ (Todd et al. 1993) to 1550 $\mu\text{g STX}_{\text{eq}} 100 \text{ g}^{-1}$ (Cembella and Desbiens 1994) ([Table 9](#)). Overall, 99% of cooked hepatopancreases contained $<250 \mu\text{g STX}_{\text{eq}} 100 \text{ g}^{-1}$ (Todd et al. 1993). Steamed lobster claws from Gaspé contained 2.7 $\mu\text{g STX}_{\text{eq}} 100 \text{ g}^{-1}$ (Lawrence et al. 1994b). In another study, lobster meat (claws plus tail muscle) showed from 2 to 69 $\mu\text{g STX}_{\text{eq}} 100 \text{ g}^{-1}$ by HPLC (Desbiens and Cembella 1995). Contrary to the Bay of Fundy lobsters, which readily cleared the toxins (Haya et al. 1992), the Gaspé lobsters retained their toxins for almost two months during storage in clean seawater (Desbiens and Cembella 1995). For this reason, it was concluded that long-term holding is not an effective approach for detoxifying contaminated lobsters. As discussed in Section [2.4.1.1](#), Health Canada (2009) has issued an advisory recommending that children not eat lobster tomalley and that adults restrict their consumption of it to no more than the amount from one lobster per day.

Also as with the Bay of Fundy, evidence is accumulating for the Gulf of St. Lawrence about the potential importance of STX group toxins in affecting fish survival and the recruitment of larval fish to adult populations. Laboratory experiments have shown that larvae of capelin (*Mallotus villosus*), Atlantic herring (*Clupea harengus harengus*) (Gosselin et al. 1989), Atlantic mackerel (*Scomber scombrus*) (Robineau et al. 1991b), Atlantic cod (*Gadus morhua*), and winter flounder (*Pseudopleuronectes platessoides*) (Robineau et al. 1991a) are vulnerable to these toxins. The fish accumulated the toxins either by grazing on toxic *Alexandrium catenella* (direct intoxication) or on toxic zooplankton, which had eaten toxic *A. catenella* (vectorial intoxication). Field studies confirmed that the co-occurrence of fish larvae and toxic *A. catenella* could threaten the survival of capelin and other larval fish species in the estuary and Gulf of St. Lawrence (Robineau et al. 1993). The copepod *Calanus finmarchicus*, a dominant member of the zooplankton community of the lower St. Lawrence Estuary, can accumulate STX group toxins when exposed to toxigenic *A. catenella* (as *A. tamarense*; Turriff et al. 1995). Not all cases of fish mortality, however, can be attributed unambiguously to phycotoxins. In early June 1993, STX group toxins (in many cases above the regulatory limit) were found in the livers of both live and dead Atlantic mackerel (*Scomber scombrus*). These fish were caught at the northern tip of Cape Breton Island, NS during a massive fish kill, as well as in the southwestern Gulf of St. Lawrence (Beaulieu et al. 1996; Castonguay et al. 1997). In this case, there is no conclusive evidence that the mortality was caused by the STX group toxins, although they may have been a contributing factor, given the weakened condition of the fish prior to spawning. The toxins were likely accumulated during normal feeding, but because these fish migrate from the mid-Atlantic coastal waters to their spawning grounds in the Gulf of St. Lawrence, it cannot easily be determined where toxins were accumulated. It has been suggested that small amounts of these toxins found in Atlantic mackerel used for lobster bait may influence lobster health (Lavallée 2002). Increased research on the possible impacts of phycotoxins on fish survival and recruitment is warranted, especially in light of the recent decline in the ground fisheries.

In August 2008, an unusually massive bloom of *A. catenella* resulted in the highest ever levels of STX group toxins (10,600 $\mu\text{g STX}_{\text{eq}} 100 \text{ g}^{-1}$) in the Gulf of St. Lawrence (Measures and Lair 2009; Dufour et al. 2010; Gracia 2011; Gracia et al. 2013; Starr et al. 2017; see also [Fatal Red Tide in the St. Lawrence Estuary](#)). The highest concentrations of *A. catenella* ever recorded at Tadoussac from 1995 to 2011 (reaching $7.4 \times 10^4 \text{ cells L}^{-1}$) occurred during this event in early August 2008; the bloom lasted at least 10 days (Scarratt et al. 2014). Heavy precipitation, warm temperatures and calm surface waters favoured the growth of the dinoflagellate at the mouth of the Saguenay River at the end of July to early August. About 100 dead seabirds (eight different species) were first observed on August 8, near Tadoussac at the confluence of the Saguenay River and St.

Lawrence Estuary. From August 5–31, carcasses of 10 beluga whales, eight harbour porpoises, and ~100 grey, harbour and other unidentified seals, were reported on the shores of the St. Lawrence Estuary. In addition, thousands of fish-eating birds (e.g. cormorants, gannets, kittiwakes, loons, eiders, fulmars, four species of gulls, razorbill, heron), fish (e.g. sand lance, rainbow smelt, sturgeon, shad), and invertebrates (e.g. sea cucumbers, crabs, whelks), were also found floating dead, or stranded, as the bloom drifted towards the south shore of the St. Lawrence Estuary and moved eastward with the Gaspé current. The bloom dissipated due to strong winds during the week of August 18, as it reached the Gulf of St. Lawrence.

A follow-up study revealed high levels of STX group toxins in the stomach ($63.2 \mu\text{g STX}_{\text{eq}} 100 \text{ g}^{-1}$) and feces ($56.5 \mu\text{g STX}_{\text{eq}} 100 \text{ g}^{-1}$), as well as other tissues, of beluga whales during the 2008 HAB event (Scarratt et al. 2014; Starr et al. 2017). This strongly indicates that the beluga were ingesting toxic prey, leading to acute poisoning. STX group toxins were also observed in single specimens of minke whale (*Balaenoptera acutorostrata*), northern bottlenose whale (*Hyperoodon ampullatus*) and harbour porpoise (*Phocoena phocoena*) (S. Michaud, Maurice Lamontagne Institute, Mont-Joli, QC unpubl. data). These intoxications, which follow toxic *Alexandrium* blooms, place endangered species such as the St. Lawrence Estuary beluga whale into peril.

2.4.3.2 Species that produce STX group toxins in the Gulf of St. Lawrence

The main dinoflagellate responsible for STX group toxins in the Gulf of St. Lawrence is *Alexandrium catenella* (formerly *A. tamarense*), which has been reported in earlier literature as *Gonyaulax excavata* (White and Lewis 1982), *Protogonyaulax tamarensis* (Cembella et al. 1988a,b; Cembella 1989), *Gonyaulax tamarensis*, and *A. excavatum* (Desbiens et al. 1990; Turgeon et al. 1990; Robineau et al. 1991a,b, 1993; Levasseur et al. 1995; Turriff et al. 1995). *Alexandrium catenella* was most frequently detected during 1994–2008 (in decreasing order) at Mont-Louis, Baie-Comeau, Sainte-Flavie, Sept-Îles, Gascons, Tadoussac and Penouille, and to a lesser degree at Tête-à-la-Baleine, Carleton, Natashquan and Havre-aux-Maisons (Lessard et al. 2020). Its bloom dynamics is described in Section [2.4.3.2.1](#).

The taxonomy of species in the *A. tamarense* complex has long been contentious (Section [2.2](#)) and genetic information for the species in the Gulf of St. Lawrence was based on only one isolate, from Baie-Comeau (Anderson et al. 1994). To examine this complex, Hamel et al. (2007) grew vegetative cells hatched from cysts collected from surface sediments of the St. Lawrence Estuary (between Trois-Pistoles and Sept-Îles), the Gaspé Bay, Anticosti Island, the Northumberland Strait, and in the Cabot Strait. Based on sequences of the large subunit of ribosomal RNA, there is only one population of the *A. tamarense* complex in the Gulf of St. Lawrence; nevertheless Hamel et al. (2007) used the more encompassing name of *A. tamarense/fundyense* (now called *A. catenella*).

Blasco et al. (2003) found a significant correlation between the abundance of *A. catenella* (as *A. tamarense*) and the STX group toxin concentration in mussels (*Mytilus edulis*) or clams (*Mya arenaria*), after analyzing data collected by Blasco et al. (1998) over an 11-year period in the northern Gulf of St. Lawrence. Only 1000 cells L^{-1} were sufficient to raise the toxin level in mussels to the harvesting closure limit. These data strengthen the case for *A. catenella* as the source organism.

A second *Alexandrium* species, *A. ostenfeldii*, was discovered in the Gulf of St. Lawrence in 1994 (Levasseur et al. 1998). Since then, it has been observed most frequently (in decreasing order) at

Mont-Louis, Sainte-Flavie, Sept-Îles, Baie-Comeau and Tadoussac, and to a lesser degree at Tête-à-la-Baleine, Penouille, Natashquan and Gascons, and rarely at Carleton and Havre-aux-Maisons (Lessard et al. 2020). Its presence went unnoticed in previous surveys, although important variations in cell size and morphology were noted in the *Alexandrium* samples. It is differentiated morphologically from *A. catenella* based on its larger cell size ($46 \pm 5 \mu\text{m}$, compared to $33 \pm 4 \mu\text{m}$) and the presence of a prominent kidney-shaped ventral pore on its first apical plate (Levasseur et al. 1998; Cembella et al. 1999, 2000). Both species exhibited overlapping spatial distributions during a 1994–1995 study, and thus exploited a similar ecological niche (Levasseur et al. 1998). The two species can now be differentiated using an rRNA oligonucleotide probe that is specific to *A. ostenfeldii* (John et al. 2003; Gribble et al. 2005). Although *A. ostenfeldii* cells have been shown to produce STX group toxins (Section 2.4.2.2), cells from the Gulf of St. Lawrence study were not tested for the presence of toxins, so their potential contribution to toxicity cannot be ascertained.

Michaud et al. (2002) found STX group toxins in small size fractions (0.22–15 μm , which excludes *A. catenella*) of seawater from the St. Lawrence Estuary, and demonstrated that bacteria were capable of autonomous production of these toxins. Furthermore, these same bacteria were a regular component of the plankton. They emphasized the importance of taking into account the potential presence of STX group toxins in these small size fractions when modelling shellfish intoxication, especially during years when *A. catenella* cell abundance is low.

2.4.3.2.1 Bloom dynamics of *Alexandrium catenella* in the Gulf of St. Lawrence

Highest toxicity of STX group toxins in molluscs is usually found along the north shore of the lower St. Lawrence Estuary in August, just after blooms of *A. catenella* are promoted during periods of water column stability due to freshwater runoff from the Manicouagan and Aux-Outardes rivers (Therriault et al. 1985; Cembella et al. 1988a; Cembella and Therriault 1989). Blooms also develop during the months of May and June, when the St. Lawrence freshwater flow rate is almost twice its usual value (Koutitonsky and Bugden 1991). In April–May, the freshwater runoff induces a large decrease in the mean surface salinity of the estuary and establishes a strong stratification that persists until fall (Therriault and Levasseur 1986). Similar oceanographic conditions, leading to blooms of *A. catenella* within the Moisie River plume, were found at Sept-Îles, in the northern Gulf of St. Lawrence (Weise et al. 2002). Brackish water plumes provide a stratified water column, which is conducive to the growth of *A. catenella*. The densest *A. catenella* blooms have occurred at salinities below 24.5, although this dinoflagellate can also grow, and be found, at a broad range of salinity (20–30) (Parkhill and Cembella 1999; Fauchot et al. 2005b). Salinity could be used as a tool for predicting these toxic blooms (Weise et al. 2002). River water also provides humic substances (dissolved organic substances) that stimulate the growth of *A. catenella* (Gagnon et al. 2005). During 1994–2008, *A. catenella* first appeared in water samples on April 22 and May 1 on the south and north shores of the Gulf of St. Lawrence, respectively, and last appeared at those regions on October 31 (Lessard et al. (2020).

MacIntyre et al. (1997) first demonstrated the capability of Gulf of St. Lawrence *A. catenella* cells to migrate vertically to obtain nitrate at night, at depth, by studying them in a laboratory water column. This would occur during periods of weak winds, which maintain a stable water column that allows the continued growth of *A. catenella* (Weise et al. 2002). Stronger winds ($>25 \text{ km h}^{-1}$) disrupt the bloom by breaking down the stratification. Turbulent waters also prevent *A. catenella* from migrating vertically. Thus, they cannot descend to depth at night, to assimilate nitrate, and swim to the surface

during the day, to take advantage of optimal light intensities (Fauchot et al. 2005a). Both nitrate and ammonium are used by *A. catenella*, but ammonium results in cells with higher toxicity; urea cannot support growth (Levasseur et al. 1995). Nitrate concentration controls the number of cell divisions carried out by *A. catenella*, but the concentration of phosphate controls its growth rate, so it can become limiting (Fauchot et al. 2005b). Bloom dynamics are thus influenced by a complex combination of physical, chemical and biological factors.

The north shore of the lower St. Lawrence Estuary is also the apparent reservoir for benthic cysts that initiate the *A. catenella* blooms found to the immediate south (Cembella et al. 1988b; Larocque and Cembella 1990; Turgeon et al. 1990). On a broader scale, cyst abundance increases from the estuary to the Cabot Strait, on the eastern end of the Gulf of St. Lawrence (Simard and de Vernal 1998); this could provide a vast reservoir for the potential initiation of blooms, as the circulation pattern retains the cysts in the Gulf. Cysts in the sediments require a maturation (dormancy) period before being able to germinate. The length of this period is determined in part by temperature; it is longer, up to 12 months, for years with lower water temperatures (Castell Perez et al. 1998). Surprisingly, short-term increases in light or temperature did not change the germination rate. About 20% of the mature cysts germinated continuously throughout the year in the field, but peaks in germination rate during August to October accounted for the annual blooms. These peaks also occurred in cysts maintained under constant conditions in the laboratory, suggesting control by a circannual internal biological clock (Castell Perez et al. 1998).

After excystment, the motile dinoflagellate cells from the lower St. Lawrence Estuary are transported across the estuary by a freshwater plume from the Manicouagan and Aux-Outardes rivers, to join the strong longshore Gaspé current that flows seaward. The resulting blooms can then contaminate molluscan shellfish along the south shore and around the Gaspé Peninsula in late summer, when the waters become stratified. This lengthy transport of cells and the complicated physical oceanography result in an erratic accumulation pattern of STX group toxins in molluscs along the south shore (Larocque and Cembella 1990). Attempts have nevertheless been made to identify cycles of toxin concentrations, as was done for the Bay of Fundy (Section 2.4.1). Therriault et al. (1985) recognized that *Alexandrium* blooms occur when the proper meteorological and hydrodynamic conditions combine to ensure maximum stability of the water column, but they could not correlate toxicity levels with such factors over a 24-year dataset. Beaulieu and Ménard (1985) had similar difficulty in finding correlations. Using the data of Therriault et al. (1985) and Beaulieu and Ménard (1985), Messieh and El-Sabh (1990) calculated mean cycles of 4.5 to 5.5 years. This was in good agreement with a 4.8 year cycle for the long-term sea level, but they considered that these cycles are still speculative. Compared to the Bay of Fundy, therefore, toxicity outbreaks in the Gulf of St. Lawrence are less regular and less predictable, making them even more difficult to monitor.

Fauchot et al. (2008) assembled available physical and biological data and developed the first coupled physical-biological model of *A. catenella* (as *A. tamarense*) blooms for the lower St. Lawrence Estuary. This was used to explore the interactions between cyst germination, cellular growth, and water circulation. Their model successfully reproduced the timing of the 1998 *Alexandrium* bloom and its coincidence with the combined plumes of the Manicouagan and Aux-Outardes rivers (Fauchot et al. 2005b). Wind-driven circulation patterns are also important for controlling the blooms' distribution between the north and south shores of the estuary. Retention along the north shore may be a pre-requisite for the development of toxic blooms in the lower St. Lawrence Estuary. The large interannual variability of the *A. catenella* blooms must still be explained

before such a model can be used successfully for bloom prediction. Data gathered in the northern Gulf of St. Lawrence as part of the phytoplankton monitoring program (Section [9.2.2.4](#)) should provide some of the additional information required for this.

New *A. catenella* (as *A. tamarensis*) resting cysts were mapped 11 months after a massive bloom in August 2008 (Gracia 2011; Gracia et al. 2013). Contrary to the hypothesis, there was no strong spatial correlation between areas where shellfish toxicity was highest during the bloom and those where the highest cyst concentrations were observed. Furthermore, the cyst concentrations detected at the beginning of July 2009 were not as high as expected, considering the abnormally dense *A. catenella* bloom the previous year, and not even as high as recorded in 1988. This illustrates the complicated relationship between seasonal cyst abundance and bloom intensity the following season (discussed in Léger and Bates 2012). Indeed, both biological and physical factors must be taken into account to determine the success of the excystment in producing viable vegetative cells that may proliferate. Thus, it is important to incorporate physical oceanographic data into monitoring and forecasting programs (Section [9.1](#)).

2.4.3.3 STX group toxins in the southern Gulf of St. Lawrence

The southern Gulf of St. Lawrence, including the gulf coasts of NB (Bourne 1965; Prakash et al. 1971; Beaulieu and Ménard 1985), NS and PE, was originally thought to be free from *Alexandrium catenella* (as *A. tamarensis*) and STX group toxins (Boyd 1978), despite its proximity to the Gaspé Peninsula. This prompted Trites and Drinkwater (1991) to examine the possible relationship, with respect to prevailing currents, between toxic blooms on the Gaspé Peninsula and these toxins in molluscan shellfish in the southern Gulf of St. Lawrence for 1988, based on the limited available data. STX group toxins were found first in shellfish along the northern Gaspé coast in April, a relatively common phenomenon. However, during May to mid-August, toxicity had spread around the Gaspé Peninsula ([Fig. 1B](#)), into the mouth of Chaleur Bay (NB), and finally onto the western and northern shores of PE. This was the first time that STX group toxins had been reported from those areas. Results of a model showed that this spatial and temporal pattern of toxin outbreaks could be explained by the transport of toxic *Alexandrium* cells around the Gaspé Peninsula by mean residual surface drift plus time-dependent wind-driven currents (Trites and Drinkwater 1991).

In contrast to the above, however, Blasco et al. (2003) did not observe a downstream progression of the timing of the toxic shellfish outbreaks from the head of the Lower Estuary towards the tip of the Gaspé Peninsula, in spite of the strong Gaspé current. The onset of mussel toxicity was, in fact, in the opposite direction. These findings about a lack of an “upstream” source of toxic *Alexandrium* cells are supported by the outcome of the high toxicity event in August 2008, described above, for the St. Lawrence Estuary (Section [2.4.3.1](#)). It was anticipated that some effect of this bloom, even residual, might be manifested, for example in Miramichi Bay, on the northeastern coast of NB. However, no STX group toxins were found in mollusc samples at that time, or later that season (Léger and Bates 2012). Interestingly, these toxins were in fact detected in Miramichi Bay and western Northumberland Strait molluscs in 2008, but this occurred in May and June, a full month before the St. Lawrence Estuary bloom.

Because no PSP problem had been documented prior to 1988 in the southern Gulf of St. Lawrence, DFO’s Inspection Branch needed to carry out only minimal monitoring at the time. However, after the 1987 DA outbreak on PE (Section [3.2.1](#)), an expanded monitoring program

revealed the presence of STX group toxins in a variety of molluscs from Chaleur Bay (NB), the northern coasts of NB and NS, and PE (Worms et al. 1993). The survey found these toxins mainly in bar clams (*Spisula solidissima*) (Table 9), but a few other bivalve species were also affected. These included mussels (*Mytilus edulis*), soft-shell clams (*Mya arenaria*), oysters (*Crassostrea virginica*), sea scallops (*Placopecten magellanicus*), and bay quahaugs (*Mercenaria mercenaria*) (Worms et al. 1993). In addition, the northern moonsnail (*Lunatia heros*; also known as *Euspira heros*), a carnivorous gastropod mollusc sampled on the northeast coast of NB, contained STX group toxins above the regulatory limit ($333 \mu\text{g STX}_{\text{eq}} 100 \text{ g}^{-1}$). Elsewhere, other moonsnails also contained unaccepted levels of toxins: $136 \mu\text{g STX}_{\text{eq}} 100 \text{ g}^{-1}$ (Richibucto Harbour, May 27, 1991); $217 \mu\text{g STX}_{\text{eq}} 100 \text{ g}^{-1}$ (Brul e Point, June 16, 1991); $155 \mu\text{g STX}_{\text{eq}} 100 \text{ g}^{-1}$ (Oak Point, July 7, 1991) (Bouchard-Steeves et al. 1993c). Levels below the acceptable limit were found in moonsnails in northern NS (e.g. $47 \mu\text{g STX}_{\text{eq}} 100 \text{ g}^{-1}$; Merigomish, May 15, 1991). The moonsnails may have slowly bioaccumulated the toxins by preying on contaminated bivalve shellfish, rather than deriving them directly from the plankton, which contained no *Alexandrium* cells (Worms et al. 1993). This example, as well as that described for the Pacific coast (Matter 1994), shows the importance of testing new species prior to exploiting them commercially.

Lobsters caught off NB and PE in 1991 contained low or non-detectable levels of STX group toxins (Todd et al. 1993). For example, of the 152 lobsters sampled off the shores of northern NB during the spring, three contained toxins in the hepatopancreas peaking at $44 \mu\text{g STX}_{\text{eq}} 100 \text{ g}^{-1}$ (Bouchard-Steeves et al. 1993c), which is below the bivalve shellfish regulatory level of $80 \mu\text{g STX}_{\text{eq}} 100 \text{ g}^{-1}$.

There was no marked occurrence of STX group toxins on the Acadian Peninsula region of northeast NB between 1992 and 1999. However, harvesting closures were again imposed in Miramichi Bay and adjacent waters in the 2000s (summarized by L ger and Bates 2012). These closures were mostly in May, June or August 2000–2008 (although not in 2005). The highest level reached was on June 14, 2006, with a record high of $4368 \mu\text{g STX}_{\text{eq}} 100 \text{ g}^{-1}$, at Val-Comeau, north of Miramichi Bay. No toxins were detected during 2009 to 2011. The causes of these periods of blooms and non-blooms remain elusive.

The possible source of toxic *A. catenella* (as *A. tamarense*) cells in Miramichi Bay was discussed by L ger and Bates (2012). Their study produced no hard evidence that *A. catenella* cysts from within Miramichi Bay were the origin of the toxic dinoflagellate blooms in that bay. Their conclusion was based on the paucity of *Alexandrium* cysts from the sediment surface of Miramichi Bay in 2007 and 2009, even after a bloom had occurred there in 2006 that resulted in the closure of molluscan harvesting due to unacceptable levels of STX group toxins. This conclusion was also supported by the lack of *Alexandrium* cysts in an area offshore of Miramichi Bay, in a previous study (Hamel et al. 2007). Other bays, with restricted exchange with offshore waters, are likely to have resident populations of *Alexandrium* that may “re-seed” themselves with cysts from a previous bloom (summarized in L ger and Bates 2012). This appears not to be the case in Miramichi Bay, which does allow an exchange of water between the bay and the offshore. Thus, the only remaining possibility is that the organisms are transported into the bay from offshore, although it is not known if this would ultimately be from the St. Lawrence Estuary, as has been found for other locations (Trites and Drinkwater 1991; Blasco et al. 2003).

2.4.4 Saxitoxin group toxins in Newfoundland and Labrador

Sporadic outbreaks of STX group toxicity, causing human illness, were reported for the first time in NL ([Fig. 1B](#)) in 1982, when monitoring for phycotoxins first started (White and White 1985). For example, the first case of PSP in NL occurred on September 25, 1982, when a 69-year-old man from Harbour Grace (Conception Bay) consumed mussels (*M. edulis*) from that area (Hockin et al. 1983). Bottled mussels in vinegar, and wild mussels collected from the same area on October 14, contained $690 \mu\text{g STX}_{\text{eq}} 100 \text{ g}^{-1}$ and $1200 \mu\text{g STX}_{\text{eq}} 100 \text{ g}^{-1}$, respectively. These mussel beds had been harvested for >40 years without reported incident (Hockin et al. 1983). Seven of 82 recreational mussel (*M. edulis*) sites harvested between 1982 and 1984, mostly along the south coast, were found to be contaminated. Trinity Bay was closed to shellfish harvesting due to STX group toxins from August to late October 1992; Green Bay was also temporarily closed, and four other areas tend to remain permanently closed (J.C. Powell, DFO, St. John's, NL, 1993 pers. comm.). There were no closures between 2003 and 2009. Monitoring of mussel tissue has increased along the northeast coast, where mussel and scallop culture industries have recently developed.

A preliminary study of cultured sea scallops (*P. magellanicus*) showed that they contained no STX group toxins at the two sites examined (McKenzie et al. 1991). One would expect that cultured blue mussels (*M. edulis*) would be the species most at risk of accumulating these toxins. However, in contrast to the harvesting of wild mussels described above, there is only a single result of cultured mussels exceeding the regulatory limit since 2000: $102 \mu\text{g STX}_{\text{eq}} 100 \text{ g}^{-1}$ in mussels from Brown's Arm (Shoal Harbour) on July 13, 2000 (CFIA data). Low or non-detectable levels of these toxins were reported from lobsters off NL in 1991 (Todd et al. 1993).

The responsible organism is believed to be *Alexandrium catenella* (identified at the time as *A. fundyense*; C.H. McKenzie unpubl. data). As in the Bay of Fundy, STX group toxin incidences are not confined to the summer-fall months, but are also reported throughout the winter at some sites. Winter resuspension of resting cysts of *A. catenella* from the sediments can lead to the occurrence of these toxins in blue mussels (*M. edulis*) (Schwinghamer et al. 1994; Harper 1997; Harper et al. 1997). Cyst surveys have been conducted, especially at aquaculture sites (McKenzie and Schwinghamer 1994), using a living cyst collection method developed for NL sediments (Schwinghamer et al. 1991). Cysts are found throughout NL, occurring in numbers ranging from only a few cysts cm^{-2} to >1000 cysts cm^{-2} (McKenzie 1996). The surveys found a positive correlation between the number of cysts in the stomach of mussels and toxin levels. A search of the water column must still be carried out to determine the role of vegetative *A. catenella* cells in toxifying mussels. Given the presence of STX group toxins found in the cysts and the risk of resuspension, the selection of aquaculture sites must be considered carefully (McKenzie et al. 1998).

2.4.5 Saxitoxin group toxins in the Canadian Arctic

Increased access and shipping to arctic waters due to reduction in sea ice has prompted interest and concern about the presence and movement of toxic algae into those waters (e.g. Walsh et al. 2011; Lefebvre et al. 2016), including the Canadian Arctic (the northern portion of the territories of Yukon, Northwest Territories, and Nunavut) (Laget 2017; Dhifallah 2019; Pućko et al. 2019). The establishment of newly introduced species in the Canadian Arctic depends on their physiological requirements (e.g. light, temperature) and environmental characteristics of arctic waters. Poulin et al. (2011) compiled a list of marine phytoplankton and sea ice algae of the Canadian Arctic. This area

includes the Hudson Bay system (Hudson Bay, Hudson Strait, and Foxe Basin), the eastern Arctic (southern Davis Strait to northern Baffin Bay and Nares Strait up to Lincoln Sea), the western Arctic (Beaufort Sea), the Canadian Archipelago (including Amundsen Gulf, and Franklin Bay), and the Canada Basin (see map Fig. 1 in Pućko et al. 2019). *Alexandrium catenella* (as *A. tamarense*) has been reported in sea ice and in plankton (Bursa 1961a,b; Hsiao 1983; Hsiao et al. 1984; Hsiao and Pinkewycz 1985b; Percy et al. 1992; Riedel et al. 2003; Niemi et al. 2011; M. Poulin unpubl. data), whereas *A. ostenfeldii* has only been reported in the plankton (Bursa 1961b; Hsiao 1983; Simard et al. 1996; Harvey et al. 1997; M. Poulin unpubl. data) ([Appendix 1](#)). *Alexandrium* sp. was observed at Churchill, Decpion Bay, and Milne Inlet, and *A. ostenfeldii* was observed in ballast tanks, but not in the water column (Dhifallah 2019). The species ability to produce STX group toxins was not investigated.

Alexandrium catenella (as *A. fundyense*) has more recently been reported in arctic and subarctic samples from the Labrador, Greenland, and Norwegian Seas, and in the Northwest Passage in the Canadian Archipelago (reported in Richlen et al. 2016). Baggensen et al. (2012) recorded the first detection of STX group toxins in cultured *A. catenella* (as *A. fundyense*) from western Greenland waters associated with toxic wild scallops. These observations, and concerns over how *Alexandrium* may survive in a warming climate, prompted a study of *A. catenella* (as *A. fundyense*) cysts in the arctic waters of western Greenland (Richlen et al. 2016). This type of study will become increasingly useful for understanding the potential for the dispersal of *Alexandrium* species in Canadian Arctic regions under warmer conditions.

3.0 Domoic Acid Group (Amnesic Shellfish Poisoning; ASP)

3.1 Description of domoic acid and its biological effects

Until 1987, PSP was the only type of shellfish poisoning of concern in Canada. Then an outbreak of a new poisoning due to eating blue mussels (*M. edulis*) from Cardigan Bay, eastern PE ([Fig. 2B](#)), led to the discovery of Amnesic Shellfish Poisoning (ASP) (Bates et al. 1988, 1989, 2018, 2019; Subba Rao et al. 1988a,b; Todd 1990, 1993; Bates 1997, 1998, 2004, 2006; Bates and Trainer 2006; Trainer et al. 2008, 2012; Lelong et al. 2012). This event had considerable political implications (Gray 1988).

Chronologies of the event are given in Gray (1988), Kosatsky (1992), Todd (1993), Anderson et al. (2001) and Hanic (2014). In brief, >150 people became seriously ill and three elderly people died between November 22–25, 1987. On November 29, NHW (now called Health Canada) epidemiologists correlated the illness with mussels eaten at restaurants and DFO inspectors then traced the mussels to Cardigan Bay (PE), site of a major aquaculture industry. Laboratory testing of those mussels showed neurotoxic symptoms in the AOAC PSP mouse bioassay. The entire Atlantic shellfishery was closed on December 11. On December 17, the potent neuroexcitatory amino acid, domoic acid (DA) ([Structure 2](#)), was identified as the phycotoxin causing ASP, after an unprecedented 102-hour non-stop search at NRC Halifax (Bird et al. 1988; Quilliam and Wright 1989; Wright et al. 1989; Hanic 2014; La Barre et al. 2014). After an analytical method was developed and

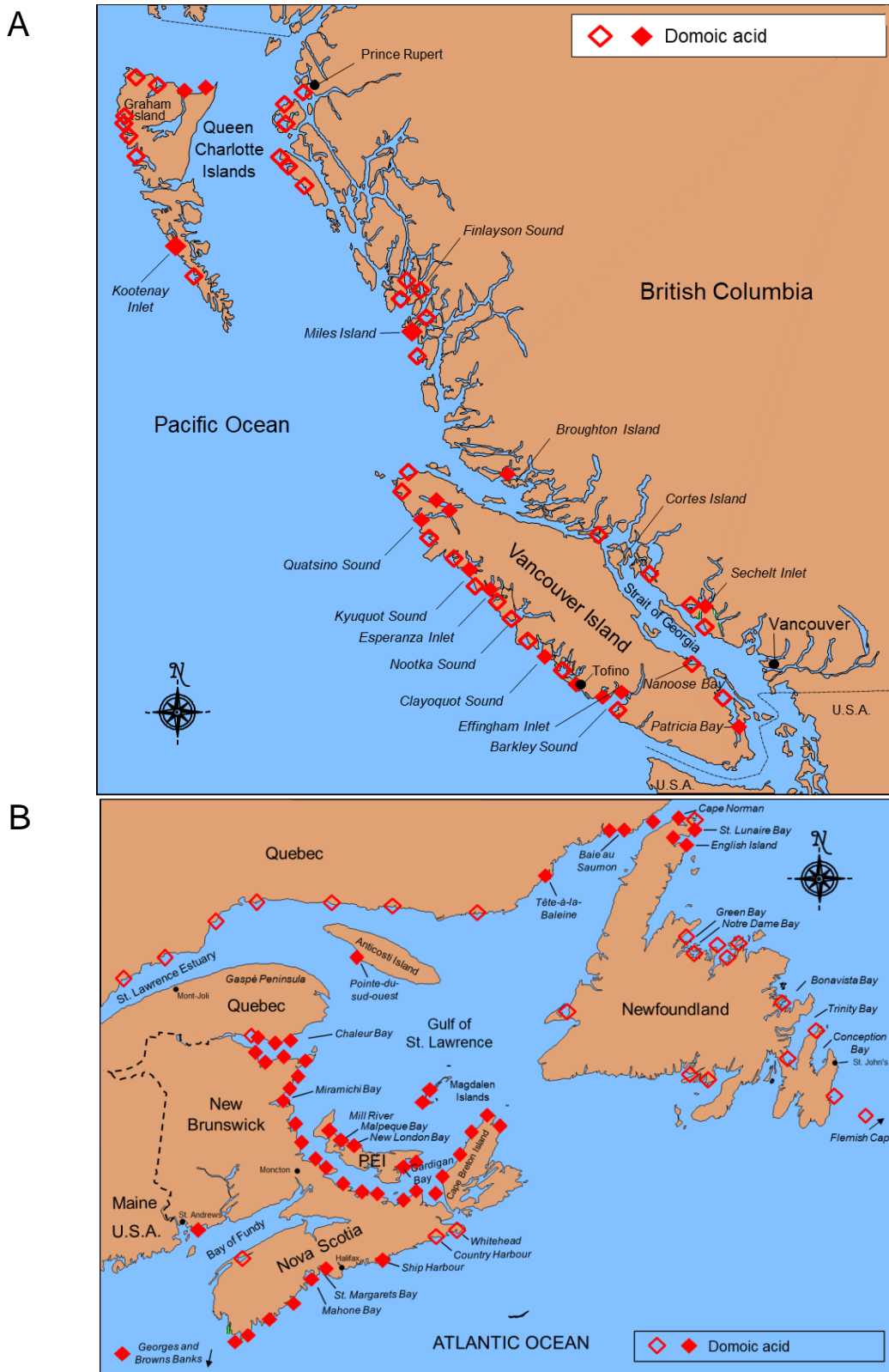
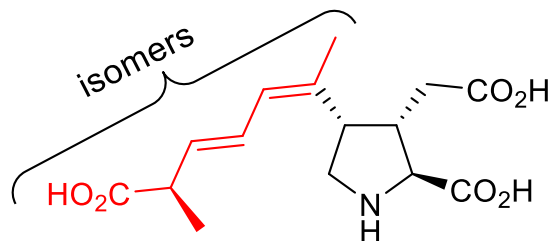


Figure 2. Location of domoic acid along the coast of A) British Columbia; B) Atlantic Canada. Open symbols: below the regulatory action level ($20 \mu\text{g g}^{-1}$ shellfish tissue); closed symbols: shellfish harvesting area closed due to exceeding the regulatory action level.



Structure 2. The structure of domoic acid. A number of minor isomers have been identified where the side chain varies through *cis-trans* configurations and positioning of the double bonds (Sawant et al. 2007; Lelong et al. 2012; La Barre et al. 2014).

deployed to inspection laboratories (Lawrence et al. 1989; Quilliam et al. 1989), it was determined that the DA contamination was localized only to Cardigan Bay, allowing the rest of the Atlantic shellfishery to be re-opened starting on January 7, 1988 (Anderson et al. 2001). The source of the DA was confirmed by culture studies to be the pennate diatom *Pseudo-nitzschia multiseries* on April 18, 1988 (Subba Rao et al. 1988a; Bates et al. 1989). A symposium to discuss the chronology and early biological and epidemiological findings on DA toxicity was held in Ottawa on April 11–12, 1989 (Hynie and Todd 1990).

Kosatsky (1992) provides lessons learned from the 1987 event, from an epidemic control viewpoint. These include: prompt and effective reporting by suspicious clinicians in order to recognize the outbreak in a timely fashion; sentinel reports indicating a risk of mass illness must be transmitted promptly throughout the public health network; public health authorities must be ready to take early and definitive action on the basis of emerging epidemiologic and laboratory evidence, even if that evidence is not complete. Kosatsky (1992) concludes that this toxic mussel outbreak shows the importance of reliable phycotoxin surveillance, of central coordination and leadership, and of decision making that both acknowledges and reduces uncertainty. Finally, an open review of how and when policy was made in such an outbreak may provide lessons for next time.

Symptoms of ASP in humans include abdominal cramps, vomiting within the first few hours, neurologic responses involving disorientation and either partial or permanent loss of short-memory, and, in the worst cases, death (Todd 1993; Trainer et al. 2008; Pulido 2014; Novelli et al. 2014; Vale 2014). Currently, the only approved therapies for humans intoxicated with DA are anticonvulsant drugs and maintenance therapy, although a number of glutamatergic antagonists are in preclinical development (Schroeder et al. 2015; Tasker 2016). Lahvis (2017) demonstrated that the behaviours and neuropathology resulting from prenatal exposure to DA are strikingly similar to those featured in autism spectrum disorder. This is supported by rodent studies indicating that young animals are substantially more sensitive than adults to the toxicity of DA (e.g. Levin et al. 2006).

There are eight natural isomers of DA (isodomoic acids A through H), plus the diastereoisomer 5'-*epi*-domoic acid, but all are less potent than DA (Sawant et al. 2007; Lelong et al. 2012; La Barre et al. 2014; Stonik and Stonik 2020). The regulatory action limit for DA is 20 $\mu\text{g g}^{-1}$ of tissue (Gilgan et al. 1990; CSSP 2019) (Table 3), which is the same as 20 mg kg^{-1} used by Health Canada (Health Canada 2016) and by the EU (EFSA 2009b). This limit is based on a retrospective estimation of the level of DA that caused illness in 1987 (200 $\mu\text{g g}^{-1}$) and incorporating a 10-fold safety factor

(Waldichuk 1989; Wekell et al. 2004). However, given the above assessment that shellfish consumption may be a risk factor for autism spectrum disorder, and that the limit was established for adults, Lahvis (2017) recommended that the regulatory limit for DA be adjusted to protect children, although a level was not given. As with STX group toxins, when DA levels exceed (or start to approach) the action limit, the CFIA recommends to DFO that the area be closed to shellfish harvesting. DFO then issues a Prohibition Order to legally prevent harvesting. The closure is enforced by Fishery Officers from DFO's Conservation and Protection Branch. An area is reopened only when three samples over a 14-day period are found to have a DA concentration $<20 \mu\text{g g}^{-1}$, and the DA levels are decreasing (CSSP 2019). More information about ASP is found at the [CFIA website](#) (CFIA 2014a).

Hydrophilic DA may be analyzed by chemical (Quilliam 2003b; Zhao et al. 1997a), *in vitro* (Cembella et al. 2003), and *in vivo* (Fernández et al. 2003) assays. Although the AOAC mouse bioassay for detecting STX group toxins (AOAC 1990) was originally used to detect high levels of DA (up to $\sim 1000 \mu\text{g g}^{-1}$) during the 1987 event in eastern Canada, it is not sensitive enough (limit of detection $\sim 40 \mu\text{g g}^{-1}$) to be used for regulatory purposes. DA is therefore analyzed by high performance liquid chromatography with ultraviolet detection (HPLC–UVD) (Quilliam et al. 1989, 1995; Quilliam 2003b; Mafra et al. 2009c), or by HPLC with fluorescence detection (HPLC–FD; Pocklington et al. 1990), and most recently with various mass spectrometry (MS, MS/MS) techniques (McNabb et al. 2005; Mafra et al. 2009c; de la Iglesia et al. 2011; McCarron et al. 2014a; Zendong et al. 2015; Beach et al. 2016a, 2018b). The analytical methods currently used by the CFIA are LC–UV and LC–MS/MS (W.A. Rourke unpubl. data). An immunoassay test, for rapid screening of DA in shellfish samples, was developed in Canada by Dr. Joanne Jellett and the NRC, in Halifax, NS (Jellett et al. 2004; Rafuse et al. 2004). It is now distributed by [Scotia Rapid Testing Ltd.](#) A novel technique, using capillary electrophoresis–reversed-phase liquid chromatography–tandem mass spectrometry (CE–MS/MS), analyzes polar toxins, including DA (Beach et al. 2018b). The Biotoxin Metrology group of the NRC, in Halifax (NS), produces certified calibration solutions and mussel tissue reference material for the determination of DA by chemical analysis methods ([NRC Certified Reference Materials](#); [Wikipedia](#); Hardstaff et al. 1990; Quilliam 2006; McCarron et al. 2017).

Since the 1987 event in eastern Canada, DA has been found to contaminate marine animals consumed by humans elsewhere in the world: in waters of the United States, Mexico, New Zealand, Portugal, Spain, France, Scotland, Ireland, and Korea (Trainer et al. 2008, 2012; Lelong et al. 2012; Bates et al. 2018). It has also been found in various marine fish, birds, and mammals in these and other countries (Bates and Trainer 2006; Lelong et al. 2012; Bates et al. 2018; Soliño et al. 2019), thus becoming a problem in marine ecosystems worldwide. Aside from being a threat to human health, DA also acts as an ecosystem disruptor by affecting the foraging responses and swimming performances of fish, leading to increasing fish vulnerability and mortality (Lefebvre et al. 2012; Mincarelli et al. 2018). Dissolved DA also provoked a significant reduction in growth and survival of king scallop (*Pecten maximus*) larvae (Liu et al. 2007), suggesting that DA exposure might influence the recruitment of this scallop species. Finally, DA has caused the direct mortality of marine whales (Fire et al. 2009, 2010), sea lions (Scholin et al. 2000), and birds (Fritz et al. 1992; Work et al. 1993; Soliño et al. 2019), although this has so far not been documented for Canadian waters. Further impacts of DA moving up the food web are reviewed by Lelong et al. (2012) and Bates et al. (2018).

Further details about DA are summarized by the FAO (2004) and Stonik and Stonic (2020). To date, 54 valid species (including three varieties: *P. pungens* var. *cingulata*, *P. pungens* var.

aveirensis, and *P. cuspidata* var. *manzanillensis*) of *Pseudo-nitzschia* have been discovered worldwide, of which 26 have been proven to produce DA (Teng et al. 2014; Bates et al. 2018; Kudela 2018; [Wikipedia](#)), although some strains are not toxic (Lelong et al. 2012; Bates et al. 2018). Over 2300 searchable references on the chemistry, biological source, food web impact, pharmacology, and toxicology of DA and ASP are maintained on the DFO [Domoic Acid and Pseudo-nitzschia References website](#) (Bates 2020). The following sections describe the impacts of DA and *Pseudo-nitzschia* in Canadian waters, starting on the east coast, where the toxin was first discovered. When available, HAEDAT event numbers for harvesting closures are given ([Table 10](#)).

3.2 Domoic acid on the Atlantic coast

3.2.1 Domoic acid in the southern Gulf of St. Lawrence

The event in PE described above resulted in an immediate cessation of mussel harvesting in all of Atlantic Canada on December 11, 1987 (HAEDAT [CA-87-006](#); [Tables 1, 10](#)) with gradual openings from January 7, 1988 until April 1988 (HAEDAT [CA-88-003](#)). This closure caused an economic loss in the millions of dollars and considerable political pressure to solve the problem (Todd 1993; Cembella and Todd 1993; Wessells et al. 1995; Anderson et al. 2001; Sanseverino et al. 2016). However, consumer confidence returned once the product was again declared safe. After a slight dip in production during 1987 to 1988, cultured mussel production in PE again increased, from 1036 metric tonnes (CAN\$1.7 million) in 1987 to 10,000 metric tonnes (CAN\$12.1 million) just 10 years later ([Fig. 3](#)) (PE DAFA 2007); in 2016, the industry produced 22,730 metric tonnes (worth CAN\$28.3 million), representing 80% of Canadian mussel production ([Mussel Industry Council of Prince Edward Island](#); [Prince Edward Island Agriculture and Fisheries](#)). This shows that the industry is able to recover after a “crisis”. An expanded shellfish monitoring program (Section [9.2.2.2](#)), implementing regulatory closures of harvesting sites when DA levels reach the regulatory limit ($20 \mu\text{g g}^{-1}$), has subsequently prevented any further cases of ASP in consumers in Canada (Worms 1991; Bugden et al. 1992; Bouchard-Steeves et al. 1993a,b,c,d) and elsewhere in the world. At the same time, mussel growers have learned to cope with the presence of any toxic mussels by shifting the harvesting to unaffected bays, and by waiting for the shellfish to depurate (detoxify) prior to harvest, thereby limiting economic losses.

Table 10. Closures of shellfish harvesting caused by levels of domoic acid (DA; $\mu\text{g g}^{-1}$) above the regulatory limit ($20 \mu\text{g g}^{-1}$ ww), or of other events related to DA, on the east and west coasts of Canada, and the event numbers in the ICES-IOC Harmful Algal Event Database ([HAEDAT](#)). Values are for whole blue mussels (*Mytilus edulis*), unless otherwise indicated. ND: below the detection limit

Date	Location	DA	HAEDAT	Reference
December 11, 1987	Eastern PE	790	CA-87-006	Bates et al. 1989
September 7, 1988	Chamcook Harbour, NB (SW Bay of Fundy)	38	–	Haya et al. 1991
September 22, 1988	SW Bay of Fundy	74	CA-88-002	Gilgan et al. 1990
September 29, 1988	SW Bay of Fundy	60	–	Haya et al. 1991
November, 1988	Eastern PE	280	CA-88-003	Smith et al. 1990a; Worms et al. 1991
November, 1989	Eastern PE	16	CA-89-001	Smith et al. 1990b

Date	Location	DA	HAEDAT	Reference
November, 1990	Cardigan Bay, PE	0.6	CA-90-003	Bouchard-Steeves et al. 1993a,b
Mid-June, 1991	NW Bay of Fundy	ND ¹	CA-91-001	–
September 27, 1991	New London Bay, PE	29	CA-91-007	Bouchard-Steeves et al. 1993c
October 1, 1991	New London Bay, PE	49	–	Bouchard-Steeves et al. 1993c
October 31, 1991	Malpeque Bay, PE	18	CA-91-006	Bates 2004
May 19, 1992	Deadmans Harbour, NB (SW Bay of Fundy)	ND ¹	CA-92-002	–
Mid- to late October, 1992	New London Bay, PE	ND ¹	CA-92-005	–
August – November, 1992	Quatsino Sound, Clayoquot Sound, Barkley Sound, BC	96 276 ²	CA-92-010	Forbes and Chiang 1994
October 28, 1994	New London Bay, PE	31.5	CA-94-005	Bouchard-Steeves et al. 1993a,c
July 21, 1994 ³	St. Lunaire Bay, NL	6.3	CA-94-008	–
September 12, 1994	Green Bay, NL	0.9	CA-94-009	–
August 6, 1994	Connaigre Bay, NL	0.2	CA-94-010	–
August 28, 1995	Lepreau Harbour, NB (SW Bay of Fundy)	10.6	CA-95-001	–
August 20, 1997	Wolves Islands, NB (Bay of Fundy)	ND ¹	CA-97-001	–
August 26, 1997	Wolves Islands, NB (SW Bay of Fundy)	ND ¹	CA-97-002	–
August 4, 1998	Wolves Islands, NB (SW Bay of Fundy)	ND ¹	CA-98-007	–
August 5, 1999	Magdalen Islands, QC	550 ⁴	CA-99-001	Couture et al. 2001
July to October 2000	Magdalen Islands, QC	265 ⁴	CA-00-001	Couture et al. 2001
August 19, 2000	Mill River, PE	33	CA-00-008	Bates and Richard 2000
July 9 – October 6, 2001	Tête-à-la-Baleine, QC	342 ⁴	CA-01-006	–
October 23, 2001	March Water, PE	27	CA-01-020	–
July 12, 2001	Ship Harbour, NS	44	CA-01-026	–
September 11, 2001	St. Lunaire Bay, NL	83	CA-01-010	–
December 20, 2001	Little Harbour, NS	21	CA-01-027	–
December 22, 2001	Malagash, NS	21	CA-02-001	–
December 22, 2001	Northern NS	–	CA-01-019	–
June 5, 2002	St. Lunaire Bay, NL	30	CA-02-012	–
April 14, 2002	Chaleur Bay, QC	90	CA-02-013	–
April 15, 2002	Stonehaven, NB	85	CA-02-034	–
April 16, 2002	Covehead Bay, PE	26	CA-02-030	–
April 22, 2002	Richibucto, NB	200	CA-02-002	–
April 24, 2002	Richibucto, NB	24	CA-02-032	–

Date	Location	DA	HAEDAT	Reference
June 14, 2002	Cape Norman to English Island, NL	49	CA-02-022	—
November 3, 2002	Espinosa Inlet, BC	31	CA-02-031	—
August 2, 2003	Tête-à-la-Baleine, QC	23 ⁴	CA-03-021	—
August 26, 2003	Anticosti Island, St. Lawrence Estuary, QC	403	CA-03-032	—
July 31, 2003	Magdalen Islands, QC	29 ⁴	CA-03-035	—
July 25, 2004	Tête-à-la-Baleine, QC	40 ⁴	CA-04-038	—
August 24, 2004	Magdalen Islands, QC	117 ⁴	CA-04-039	—
July 10, 2005	Tête-à-la-Baleine, QC	126 ⁴	CA-05-011	—
August 26, 2005	Magdalen Islands, QC	114 ⁴	CA-05-017	—
September 14, 2006	Burnt Arm North, Notre Dame, NL	43	CA-06-019	—
June 6, 2006	Kootenay Inlet (Haida Gwaii archipelago), BC	36 ⁵	CA-06-022	—
July 27, 2008	Baie Jacques-Cartier, QC	289 ⁶	CA-08-062	—
August 12, 2008	Lime Kiln Bay, NB (SW Bay of Fundy)	7	CA-08-001	Fernandes et al. 2014; Martin et al. 2014c
July 22, 2008	Magdalen Islands, QC	238 ⁴	CA-08-069	—
August 8 – September 21, 2009	Magdalen Islands, QC	67	CA-09-025	—
July 8, 2012	Baie au Saumon, QC	21 ⁶	CA-12-022	—
September 18, 2012	Graham Island (Haida Gwaii archipelago), BC	42 ⁷	CA-12-012	ICES 2014; Russell 2016
July 10, 2013	Ship Harbour, NS	20	CA-13-004	—
July 28, 2013	Havre des Belles Amours, QC	48	CA-13-022	ICES 2014
August 6, 2013	Île d'Entrée, Magdalen Islands, QC	420 ⁸	CA-13-023	ICES 2014
August 6, 2013	Griquet Harbour, Northwest Bay, NL	15	CA-13-024	—
June 1, 2014	Baie au Saumon, QC	58 ⁶	CA-14-011	—
October 1, 2014	Deadmans Harbour, NB (SW Bay of Fundy)	320 ⁶	CA-14-004	ICES 2017
July 22, 2014	Magdalen Islands, QC	104 ⁴	CA-14-012	—
July 22 – August 14, 2015	Baie au Saumon, QC	23 ⁵	CA-15-009	—
Mid-June, 2015	Esperanza Inlet, West Vancouver Island, BC	75	CA-15-001	Haigh et al. 2016
November 9, 2015	Saanich Inlet, south Vancouver Island, BC	54	CA-15-006	Haigh et al. 2016
June 30 – July 20, 2016	Baie au Saumon, QC	23 ⁶	CA-16-006	—
September 8, 2016	Miles Island, mainland BC	25 ⁵	CA-16-024	—

Date	Location	DA	HAEDAT	Reference
September 26 – October 4, 2016	Deep Brook, NS (SE Bay of Fundy)	30 ⁸	CA-16-020	–
September 9 – 26, 2016	Clam Cove/Northern Harbour, NB (SW Bay of Fundy)	30 ⁹	CA-16-019	ICES 2017
September 21, 2016	Clam Cove, NB (SW Bay of Fundy)	21 ⁹	CA-16-026	–
December 13, 2017	Indian Point, Mahone Bay, NS	75 ⁵	CA-17-018	–
November 8, 2018	Snake Island, Mahone Bay, NS	30 ⁵	–	–
December 3, 2018	Indian Point, Mahone Bay, NS	37 ⁶	–	–

¹ Elevated concentration of *Pseudo-nitzschia pseudodelicatissima* present

² Dungeness crab (*Metacarcinus magister* = *Cancer magister*) hepatopancreas

³ First detection of DA in NL (July 21, 1994). No closure order issued, as below the action limit.

⁴ Sea scallop (*Placopecten magellanicus*) digestive gland (= hepatopancreas)

⁵ Geoduck clam (*Panopea generosa*) viscera

⁶ Whole sea scallop (*Placopecten magellanicus*)

⁷ Razor clam (*Siliqua patula*)

⁸ Sea scallop (*Placopecten magellanicus*) gonads

⁹ Soft-shell clam (*Mya arenaria*)

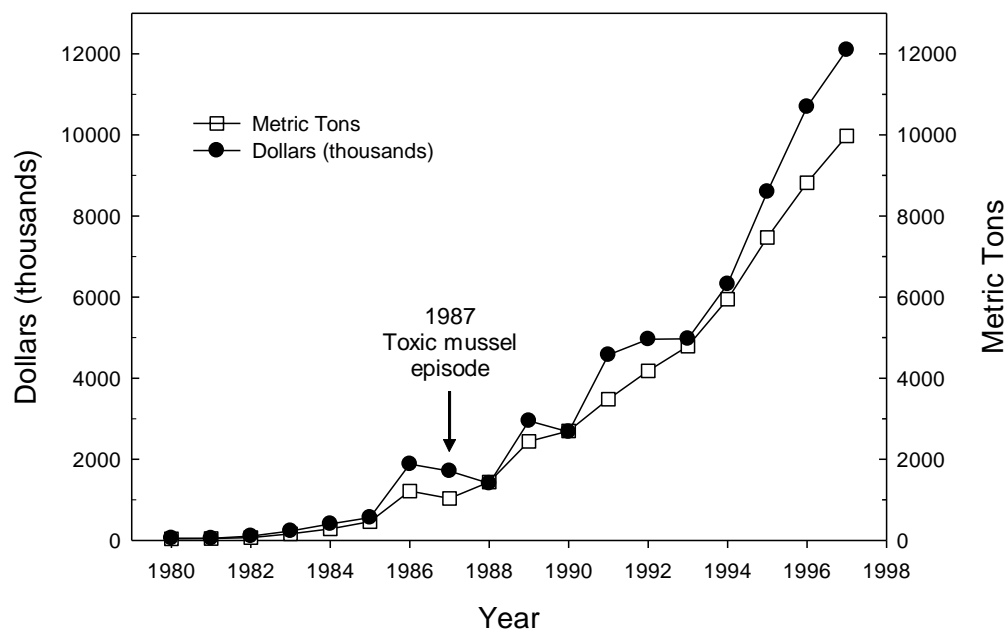


Figure 3. Prince Edward Island mussel landings and values: 1980–1997. Created from data found in PE DAFA (2007). Original data from: Fisheries and Oceans Statistics Division (Moncton, NB) and Prince Edward Island Department of Agriculture and Fisheries.

During the original Cardigan Bay event in 1987, DA was found at the highest level ever recorded ($790 \mu\text{g g}^{-1}$) in blue mussels (*M. edulis*), and at lower levels ($38 \mu\text{g g}^{-1}$) in soft-shell clams (*Mya arenaria*) (Bates et al. 1989). Subsequently, only five molluscan shellfish species have been shown to accumulate DA in eastern Canadian waters (Table 11). This is in contrast to other regions of the world, where DA has been shown to accumulate in or cause the mortality of molluscs (34 species), crustaceans (16 species), fish (25 species), birds (7 species), mammals (11 species), and other (4 species) (Lelong et al. 2012; Trainer et al. 2012; Bates et al. 2018; Soliño et al. 2019; Van Hemert et al. 2020). In particular, no marine seabirds or mammals have been affected in Canadian waters due to DA, perhaps because no intermediary vectors have been shown to be affected.

When DA was found to be the toxin responsible for the 1987 PE event, the literature showed that one source of this compound was the red macroalga *Chondria armata*, found in Japan (references in Bates et al. 1988, 1989). A congener of *C. armata*, *C. baileyana*, occurs in the southern Gulf of St. Lawrence and was indeed shown to contain DA (Bird et al. 1988). However, no *C. baileyana* was found in the vicinity of the toxic mussels. Furthermore, the digestive gland of the mussels was gorged with fragments of a diatom that was later identified as the pennate diatom *Pseudo-nitzschia multiseries* (Bates et al. 1989). Cultures of that diatom produced DA in culture, proving that this was the source of the toxin (Subba Rao et al. 1988a,b; Bates et al. 1989). In the older literature, this diatom was called *Nitzschia pungens* forma *multiseries* (*Nitzschia pungens* f. *multiseries*) and *Pseudonitzschia multiseries* (Hasle 1994, 1995).

Table 11. Shellfish species known to become contaminated with domoic acid (DA; $\mu\text{g g}^{-1}$) in coastal waters of eastern Canada. The whole-animal toxin levels ($\mu\text{g g}^{-1}$ ww of tissue) are the highest values found for the reference listed. Levels are for the whole animal, unless indicated otherwise. The closure level is $20 \mu\text{g g}^{-1}$ ww of tissue.

Organism		DA	Location	Reference
Common name	Scientific name			
Blue mussel	<i>Mytilus edulis</i>	790	Cardigan Bay, PE	Bates et al. 1989
Blue mussel	<i>Mytilus edulis</i>	200	Richibucto, NB	Mafra et al. 2009a
Blue mussel	<i>Mytilus edulis</i>	74	Passamaquoddy Bay, NB (SW Bay of Fundy)	Gilgan et al. 1990
Eastern oyster	<i>Crassostrea virginica</i>	0.9	Neguac Bay, NB	Mafra et al. 2009a
Horse (red) mussel	<i>Modiolus modiolus</i>	7.5	Yarmouth, NS (SE Bay of Fundy)	Gilgan et al. 1990
Sea scallop	<i>Placopecten magellanicus</i>	550 ¹	Magdalen Islands, QC	Couture et al. 2001; Levasseur et al. 2001
Sea scallop	<i>Placopecten magellanicus</i>	93.4 ¹	German Bank (43°20'N, 66°25'W)	Douglas et al. 1997

Organism		DA	Location	Reference
Common name	Scientific name			
Sea scallop	<i>Placopecten magellanicus</i>	4300 ¹	Browns Banks (42°45'N, 66°W)	Reported in Douglas et al. 1997
Sea scallop	<i>Placopecten magellanicus</i>	83	St. Lunaire Bay, NF	CFIA data
Soft-shell clam	<i>Mya arenaria</i>	38	Cardigan Bay, PE	Bates et al. 1989
Soft-shell clam	<i>Mya arenaria</i>	45	Passamaquoddy Bay, NB (SW Bay of Fundy)	Gilgan et al. 1990

¹ Hepatopancreas / digestive gland

In PE, blooms of toxic *P. multiseriis* have been restricted to the autumn (Subba Rao et al. 1988b; Smith et al. 1990a,b; Bates et al. 1998). The presence of a nontoxic species, *P. pungens* (previously called *P. pungens* f. *pungens* and was once considered the same species as *P. multiseriis* [Hasle 1995]), complicates programs that monitor for the presence of toxic phytoplankton (Smith et al. 1990b). Aside from their ability to produce DA, the two species could previously only be distinguished by electron microscopy (Hasle 1965). Now, they can also be distinguished by immunochemical (Bates et al. 1993) and lectin-binding (Fritz 1992; Pauley et al. 1994a) assays, as well as by differences in the nucleotide sequences of the ribosomal RNA gene (Douglas et al. 1994; Scholin et al. 1994, 1996; Manhart et al. 1995; Miller and Scholin 1996; Scholin 1998; Lim et al. 2018). In Canada, however, the main method for identifying *Pseudo-nitzschia* species is scanning electron microscopy (SEM) (e.g. by James Ehrman and Irena Kaczmarek at the [Digital Microscopy Facility](#), Mount Allison University, Sackville, NB). When present, autumn blooms of the toxic *P. multiseriis* have usually been preceded by the nontoxic *P. pungens* (Smith et al. 1990a). Interestingly, the same sequence of succession was observed for these two species in the Gulf of Mexico (Fryxell et al. 1990).

Blooms of the toxic *P. multiseriis* have declined dramatically in eastern PE since the original 1987 episode. This has accounted for a parallel decreasing level of DA in Cardigan Bay whole blue mussels (*M. edulis*). For example, maximum values of DA ($\mu\text{g g}^{-1}$ ww) during November were 790 in 1987 (Bates et al. 1989), 280 in 1988 (Smith et al. 1990a; Worms et al. 1991; HAEDAT [CA-88-003](#); [Table 10](#)), 16 in 1989 (Smith et al. 1990b; HAEDAT [CA-89-001](#)), 0.6 in 1990 (Bouchard-Steeves et al. 1993a,b; HAEDAT [CA-90-003](#)), and non-detectable in 1991, 1992 and 1993 (Bouchard-Steeves et al. 1993c,d). An examination by SEM ([Digital Microscopy Facility](#)) of Cardigan Bay phytoplankton during those years revealed that the nontoxic *P. pungens* was present in greater numbers than toxigenic *P. multiseriis*, which was virtually absent. As well, in the autumn of 1990, northerly winds dispersed an incipient bloom of *P. multiseriis*, and in 1991, a violent storm may have done the same (Bates et al. 1998).

Several shellfish harvesting areas, however, were later closed in northern PE ([Fig. 2B](#)) due to elevated levels of DA: Malpeque Bay on October 31, 1991 (HAEDAT [CA-91-006](#); $18 \mu\text{g g}^{-1}$), and October 23, 2001 (HAEDAT [CA-91-020](#); $27 \mu\text{g g}^{-1}$); and New London Bay on September 27, 1991 (HAEDAT [CA-91-007](#); $29 \mu\text{g g}^{-1}$; *Mytilus edulis*; 8.0×10^5 *P. multiseriis* cells L^{-1}), and October 24, 1994 (HAEDAT [CA-94-005](#); $31.5 \mu\text{g g}^{-1}$; *M. edulis*; 3.1×10^5 *P. multiseriis* cells L^{-1}) (Bouchard-Steeves et al. 1993a,c; Bates 2004). In mid- to late October 1992, low levels of DA were detected in *P. multiseriis* (blooming at 3.0×10^4 cells L^{-1}) in New London Bay, but this did not result in any harvesting closures. Mussels in Darnley Basin contained $14 \mu\text{g g}^{-1}$ on October 28, 1991 (Bouchard-Steeves et al. 1993c).

Subsequently, blue mussels (*M. edulis*) collected on September 19, 2000, from the Mill River, western PE, were found to contain DA at $33 \mu\text{g g}^{-1}$ (Bates and Richard 2000). DFO therefore closed all molluscan shellfish harvesting on September 22, for the Mill River area (HAEDAT [CA-00-008](#)). This is the first ever closure due to DA in western PE. The concentration of *P. multiseriis* cells in a water sample from Mill River was $\sim 2.2 \times 10^5$ cells L^{-1} on September 23. Although cell counts this high were not uncommon in the autumn, they had been dominated by the nontoxic *P. pungens* since at least 1994. No *P. pungens* cells were found in the September 2000 samples, which was unusual. The *P. multiseriis* cells appeared in poor condition, suggesting that they may have been nutrient limited, which laboratory studies have shown to be correlated with high toxicity (reviewed in Lelong et al. 2012). Within eight days of the peak in shellfish toxicity, the event was over, when only trace levels of DA ($1 \mu\text{g g}^{-1}$) were reported in the mussels; shellfish harvesting resumed on October 6. Closures caused by DA then occurred in New London Bay and Cascumpec Bay, on November 5, 2002, for 28 days.

On December 22, 2001, ~ 200 km of coastline along the Northumberland Strait (Malagash Point to Cape George, northern NS) was closed due to DA (HAEDAT [CA-01-019](#), [CA-02-001](#)). The event was unusual because this was the first closure for this location, at a time of year when DA had not previously been observed, and it covered a very large area. DA was first detected in blue mussels (*M. edulis*) at low concentrations ($6.6 \mu\text{g g}^{-1}$) from Caribou Harbour, Pictou. Values reached only $21 \mu\text{g g}^{-1}$, at Little Harbour, Pictou (HAEDAT [CA-01-027](#)), but this was enough to trigger the closure of shellfish harvesting in that area. Oysters (*Crassostrea virginica*) were non-detectable for DA. The closure occurred during the Christmas period, when sales of molluscan shellfish are traditionally high. Of the potentially toxic diatoms, *P. pseudodelicatissima* (likely nontoxic) was the most abundant, followed by *P. multiseriis* (the likely toxin source), *P. fraudulenta* (a first record for the Gulf of St. Lawrence), and *P. pungens*. The bloom was likely advected onshore from the southern Gulf of St. Lawrence, although DA was undetectable in mussels from Cape Breton Island to the east. The area was reopened on January 10, 2002 (HAEDAT [CA-02-001](#)).

In northern and eastern PE, toxigenic blooms of *Pseudo-nitzschia* had always occurred only in the fall. It was therefore a great surprise when the CFIA biotoxin monitoring program detected DA in blue mussels (*M. edulis*) from a processing plant at New London Bay (PE), on March 5, 2002. Phytoplankton sampling was immediately carried out through the ice in New London Bay, and a high proportion of *P. seriata* (identified using SEM at the [Digital Microscopy Facility](#)) was found. This led to more extensive monitoring of mussels and to a shellfish harvesting closure, on April 5, 2002 (HAEDAT [CA-02-002](#)), in nine bays of northern PE ([Fig. 2B](#)). For example, other locations were Covehead (PE), with $26 \mu\text{g g}^{-1}$, on April 16 (HAEDAT [CA-02-030](#)), and Richibucto (NB), and Stonehaven (NB), with $85 \mu\text{g g}^{-1}$ on May 15, 2002 (HAEDAT [CA-02-034](#)).

This was the first closure ever due to DA during the spring (Bates et al. 2002; Bates 2004). Closures were expanded to portions of Chaleur Bay (QC) and to the entire coast of eastern NB on April 17, then to portions of Cape Breton Island (NS) on May 1, when it was discovered that the presence of the DA was more widespread (van de Riet et al. 2006). Most areas were reopened by the end of May, 2002. Such a broad extent of closures, encompassing most of the southern Gulf of St. Lawrence, suggests a wide-spread bloom of toxic diatom cells. The bloom was likely distributed throughout the southern Gulf of St. Lawrence by the prevailing water currents, which move towards the southeast. Water samples collected at each affected site showed that *P. seriata* was the only potentially toxic diatom present. The highest cell concentration was 4.0×10^5 cells L^{-1} , at St. Peter's Bay, on April 10, 2002. This corresponded to the highest concentration of DA in mussels (*M. edulis*) at that location ($71 \mu g g^{-1}$), on April 16 (CFIA data). In NB, *P. seriata* reached 3.0×10^5 cells L^{-1} on April 22, in Richibucto, giving the highest DA concentration in mussels of $200 \mu g g^{-1}$ (HAEDAT [CA-02-002](#)). On the QC shore of Chaleur Bay (Carleton and Gascons), the closure started on April 14 (HAEDAT [CA-02-013](#)). The *P. seriata* concentration was 5.4×10^4 cells L^{-1} at Port Daniel on April 19, and the DA concentration in mussels was $89.5 \mu g g^{-1}$ ([Table 10](#)). This area was reopened on May 4, 2002.

The triggers for this unusual 2002 spring bloom are unknown, but the ice conditions were different compared to previous years. For example, due to well above normal temperatures during the fall and the early part of the winter in 2001, the total ice coverage for the Gulf of St. Lawrence was the sixth lowest on record (Canadian Ice Service 2002). Freeze-up was delayed by 2 to 3 weeks (arriving late January to early February) and ice departed early (April 8), compared to the long-term mean ice departure of April 14. This may have provided the increased light conditions required for the bloom to develop. It should be noted that there have been no subsequent closures due to DA in the southern Gulf of St. Lawrence, as concentrations of toxigenic *Pseudo-nitzschia* species have not reached dangerous levels, for reasons that are still not understood.

As a side note, it is interesting that, although the highest DA value in *Mytilus edulis* was $200 \mu g g^{-1}$, at Richibucto (NB), in April 2002 (van de Riet et al. 2006; Mafra et al. 2009a; HAEDAT [CA-02-002](#)), American oysters (*Crassostrea virginica*) from the same location showed no detectable levels of DA (CFIA data). Elsewhere along the northwestern coast of NB, the highest DA level in *C. virginica* was only $0.9 \mu g g^{-1}$, at Bouctouche, in August 2002. Laboratory studies demonstrated that different feeding mechanisms by *M. edulis* and *C. virginica* resulted in the lower feeding of *Pseudo-nitzschia* cells by *C. virginica* (Mafra et al. 2009a,b). As well, *C. virginica* was shown to have negligible filtration rates, compared to *M. edulis*, at the low temperatures found during this spring closure (Comeau et al. 2008). These findings led to a proposal that harvesting closures could be determined on a molluscan species-specific basis under certain circumstances (Bricelj, Bates and Mafra; unpubl. document: "Domoic Acid in Oysters [*Crassostrea virginica*] and Mussels [*Mytilus edulis*]; Management Recommendations", submitted to the CFIA, April 2011). A CSSP policy was issued to allow the harvest of oysters during DA closures. Although "management by species" would undoubtedly cause logistical problems for monitoring and enforcement, it is possible to overcome these issues with appropriate planning when supported by scientific data, and these situations are evaluated on a case-by-case basis. One example is that the CFIA in BC announced in 2017 that it will now analyze for biotoxins in harvest areas where Pacific geoduck clams (*Panopea generosa*) and other bivalve species are commercially harvested ([Notice to Industry - Marine Biotoxin Monitoring in Geoducks in British Columbia website](#)). Prior to this, the CFIA had used mussels as the only sentinel species in BC to provide early warning or an

indication of contamination in all commercially harvested species. The change was made because “recent information reveals that marine biotoxin levels in mussels do not consistently and accurately predict the marine biotoxin levels in geoduck clams (i.e. while biotoxin levels in mussels are acceptable and an area is ‘Open’, geoduck clams may have unacceptable biotoxin levels)” (CFIA 2017a).

Twelve clones of *P. seriata* were isolated from several southern Gulf of St. Lawrence sites during the 2002 event, and were grown in culture at 8 °C. Tests showed that the clones are capable of producing DA during mid-stationary phase (day 25) in batch culture, although at low levels (<1 pg cell⁻¹). In late-stationary phase (day 51), levels had increased to 7 pg cell⁻¹ (S.S. Bates and C. Léger unpubl. data). Known as a “cold-water” diatom, *P. seriata* is found only in northern latitudes (Hasle 2002; Poulin et al. 2011; Lelong et al. 2012; Bates et al. 2018). It is normally present at low concentrations year round in the Gulf of St. Lawrence, and has been associated with high levels of DA in Magdalen Islands sea scallops (*Placopecten magellanicus*) and with trace levels in soft-shell clams (*Mya arenaria*), since 1998 (Couture et al. 2001; Levasseur et al. 2001). There have been no other harvesting closures due to DA in the southern Gulf of St. Lawrence since 2002.

In earlier studies, a *P. seriata* strain isolated from Cardigan Bay (PE) in March 1988, was shown to be nontoxic for DA in culture (Bates et al. 1989), as was a *P. seriata* strain from Narragansett Bay, RI (Hargraves et al. 1993). However, isolates of *P. seriata* from Danish (Lundholm et al. 1994) and Scottish (Fehling et al. 2004a,b, 2005) waters produced the toxin at levels comparable to those found in *P. multiseriata* from Cardigan Bay. This suggests that toxic and nontoxic strains exist of the same species, as has been documented for other species of *Pseudo-nitzschia* around the world (Lelong et al. 2012; Bates et al. 2018). The finding is significant because *P. seriata* is a common species of *Pseudo-nitzschia* in the North Atlantic, and its presence may explain the detection of DA in the absence of *P. multiseriata*.

It must be noted that there may be sources of DA other than *P. multiseriata* and *P. seriata* in the southern Gulf of St. Lawrence. For example, the benthic pennate diatom *Halamphora coffeiformis* (sometimes spelled as *H. coffeaeformis*, and previously called *Amphora coffeaeformis*) was isolated from toxic mussels of the original 1987 episode in Cardigan Bay (PE), and was shown to produce low amounts of DA (Shimizu et al. 1989; Maranda et al. 1990). However, other isolates from elsewhere in the world proved to be nontoxic (Bates et al. 1989) and there is some uncertainty about the identity of the *H. coffeiformis* isolate from PE (Bates 2000). DA was later found in plankton from a sub-ice spring bloom in the Cardigan River (Pauley et al. 1993). These samples did not contain *P. multiseriata*, and *P. seriata* was not mentioned, but cell counts of the diatom *Fragilaria* correlated well with changes in DA concentration. Isolates of *P. delicatissima* (previously called *P. actyodrophila*) from Cardigan Bay may be another low-level producer of DA (5 fg cell⁻¹), based on the retention time of the compound using HPLC (Smith et al. 1990b, 1991). Subsequently, *P. delicatissima* was redefined and split into at least three species, which may explain in part why some isolates have been reported to be nontoxic and others toxic (reviewed in Lelong et al. 2012; Trainer et al. 2012).

In a comprehensive study of HAB species in 14 inlets of PE during 2001 to 2003, *P. multiseriata* was infrequently found, and only in low numbers (Bates and Strain 2006). Other species of *Pseudo-nitzschia*, including the nontoxic *P. calliantha*, *P. pungens* and *P. delicatissima*, bloomed in its place. Two new species of *Pseudo-nitzschia* (*P. americana* and the tentatively identified *P.*

subpacifica) were reported for the first time in the Gulf of St. Lawrence. *Pseudo-nitzschia fraudulenta* was identified for the first time in PE inlets.

Research has attempted to identify factors that are responsible for initiating *Pseudo-nitzschia* blooms in coastal waters of PE, and for explaining the inter-annual variability and/or decline in toxicity. The conditions that apparently contributed to the 1987 bloom were a prolonged dry period in summer, followed by an unusually rainy autumn that may have provided nutrients via river runoff (Smith et al. 1990a). This meteorological situation has not recurred, but numerous other biological and non-biological factors may contribute to bloom formation and decline. For example, the life history and dioecious mode of sexual reproduction of *P. multiseriis*, discovered for the first time by Davidovich and Bates (1998a,b), may play a role. Knowledge of the rate of decrease in *P. multiseriis* cell size, and of differences in toxicity of large and small cells, may allow one to predict the timing (Bates et al. 1998) and toxicity (Sauvey 2018) of their blooms. It is not known, however, where the cells disappear to during the spring and summer, and what makes them return in the autumn.

Another biological factor is the discovery of oomycete and chytrid parasites that infect *P. pungens* (Pauley et al. 1994b; Hanic et al. 2009), *P. seriata* and *P. delicatissima* (Pauley et al. 1994b) cells in Cardigan Bay (PE). The parasites had a dramatic effect on *Pseudo-nitzschia* population dynamics (Pauley et al. 1994b). Interestingly, however, no *P. multiseriis* cells from PE have ever been shown to be infected by these parasites. So, the apparent decline in numbers of that species in waters of PE cannot be attributed definitively to this factor. Nevertheless, morphologically similar parasites have been shown to infect *P. multiseriis*, *P. pungens* and *P. australis* cells in WA State, U.S. (Lelong et al. 2012), *P. pungens* in PE and Helgoland, Germany (Buaya et al. 2017), and *Pseudo-nitzschia* sp. in the Bay of Seine, France (Thorel et al. 2017). Aside from the diatom *Pseudo-nitzschia*, chytrid parasites have been shown to play a significant ecological role in the population dynamics of other HAB species (Gleason et al. 2015). It was hypothesized that chytrids may even cause phycotoxin production by phytoplankton as a defensive response to infection (Gleason et al. 2015). On the other hand, the role of viruses in controlling *Pseudo-nitzschia* bloom dynamics or modifying cell physiology to trigger DA production is just in its early stage of study (Lelong et al. 2012).

3.2.2 Domoic acid in the northern and central Gulf of St. Lawrence

The illness of 20 people caused by consuming mussels and scallops contaminated by OA group toxins in the Magdalen Islands in the summer of 1998 (Section 4.4.3) prompted an investigation by the CFIA and the DFO laboratory in Mont-Joli (QC) into the extent of phycotoxin contamination at sites in the Magdalen Islands, the Gaspé Peninsula, the Lower St. Lawrence River, and the north shore of the St. Lawrence River (Levasseur et al. 2001). Aside from OA group toxins, the investigation found DA for the first time in QC waters (Couture et al. 2001).

Results from the CFIA documented the spatial and temporal changes in DA contamination in the northern and central QC waters of the Gulf of St. Lawrence between 1998 and 2000 (Couture et al. 2001). Trace amounts of this toxin were first detected in the gonads of sea scallops (*Placopecten magellanicus*) from fishing areas offshore of the Magdalen Islands (Fig. 2B) in the summer of 1998. In 1999, the concentration of DA in the digestive glands of scallops from the same area reached 550 (HAEDAT CA-99-001; Table 10) to 585 $\mu\text{g g}^{-1}$ (Couture et al. 2001), whereas the adductor muscles were below the level of detection. At the same time, DA reached 25 $\mu\text{g g}^{-1}$ in the digestive glands of scallops from the Havre-aux-Maisons lagoon. Trace amounts were

also measured for the first time in soft-shell clams (*Mya arenaria*) collected on the Lower North Shore of the Gulf of St. Lawrence. In 2000, the digestive glands of scallops from the Magdalen Islands remained toxic from the 1999 event. DA was again reported in Magdalen Islands scallops ([Table 10](#)): 117 $\mu\text{g g}^{-1}$ on August 24, 2004 (HAEDAT [CA-04-039](#)); 114 $\mu\text{g g}^{-1}$ on August 26, 2005 ([CA-05-017](#)); 238 $\mu\text{g g}^{-1}$ digestive gland on July 22, 2008 (HAEDAT [CA-08-069](#)); 420 $\mu\text{g g}^{-1}$ digestive gland on August 6, 2013, in scallop gonads (ICES 2014; HAEDAT [CA-13-023](#)); and 104 $\mu\text{g g}^{-1}$ digestive gland (= 21 $\mu\text{g g}^{-1}$ whole tissue) on July 22, 2014 (HAEDAT [CA-14-012](#)), at l'Île d'Entrée. Between August 8 and September 21, 2009, DA, at 49–67 $\mu\text{g g}^{-1}$, was detected in commercial mussels (*M. edulis*) from the Magdalen Islands (HAEDAT [CA-09-025](#)).

Trace amounts of DA were detected in molluscs all along the Lower North Shore, from Tadoussac to Havre-Saint-Pierre. The maximum DA value in sea scallop hepatopancreas from Tête-à-la-Baleine ([Fig. 2B](#)) was 342 $\mu\text{g g}^{-1}$ during July 9 to October 6, 2001 (HAEDAT [CA-01-006](#)). That same region experienced DA again, at 23 $\mu\text{g g}^{-1}$, on August 2, 2003 (HAEDAT [CA-03-021](#)); DA was present from April 26 to November 10, 2003. The following year, the region was again impacted by DA, at 40 $\mu\text{g g}^{-1}$, on July 25, 2004 (HAEDAT [CA-04-038](#)); DA was present from July 4 to August 29, 2004. On July 10, 2005, DA was again found in sea scallops at Tête-à-la-Baleine at a concentration of 123 $\mu\text{g g}^{-1}$ (HAEDAT [CA-05-011](#)); DA was present from June 26 to October 18, 2005. DA was reported on Anticosti Island (St. Lawrence Estuary), at 403 $\mu\text{g g}^{-1}$, on August 26, 2003 (HAEDAT [CA-03-032](#)); DA was present from April 26 to October 11, 2003.

In addition to the CFIA data, the Toxic Algae Monitoring Program (Section [9.2.2.4](#)) of DFO's Maurice Lamontagne Institute (MLI) revealed the presence of *Pseudo-nitzschia seriata* (a producer of DA; see above and below) in the Gulf of St. Lawrence, and reported on its distribution, abundance, and relationship with temperature, salinity and nutrients between 1994–2008 (Lessard et al. 2020). This species had been previously identified there (Roy et al. 1996). A link was found between DA in some molluscs from the Magdalen Islands and the Lower North Shore and the presence of *P. seriata* (Couture et al. 2001). Dense blooms of *P. delicatissima* (with no *P. seriata*) did not cause toxicity. Laboratory analyses performed on a *P. seriata* strain isolated from the St. Lawrence Estuary during a toxic event showed the ability of that strain to produce DA, whereas all isolates of *P. delicatissima* from other regions of eastern Canada have so far been negative. It may be that cells identified at the time as *P. delicatissima* (Couture et al. 2001) were actually one of the at least four new species in the *delicatissima* complex that are morphologically identical, but genetically distinct (i.e. they are “cryptic” species), some of which are nontoxic (Bates et al. 2018).

Independently, a study funded by the DFO ACRDP was carried out at two sea scallop (*Placopecten magellanicus*) aquaculture sites (Baie au Saumon and Baie Jacques-Cartier) in the northeast Gulf of St. Lawrence ([Fig. 2B](#)) during 2006–2008 (Scarratt et al. 2007; Scarratt 2009). The study was prompted by finding elevated levels of DA in whole scallops at Baie au Saumon: 39 $\mu\text{g g}^{-1}$ on August 1, 2004, and 126 $\mu\text{g g}^{-1}$ on July 10, 2005 (CFIA data). The goals were to identify the source organism responsible for these intoxications and to investigate the environmental conditions that led to the toxic blooms. During 2008, harvesting was closed at the Baie Jacques-Cartier site when DA in the whole scallops reached 72 $\mu\text{g g}^{-1}$ on July 22 and peaked at 289 $\mu\text{g g}^{-1}$ on July 27 (HAEDAT [CA-08-062](#); [Table 10](#)); the toxicity persisted for several weeks. Concurrently, a positive linear correlation was observed between the concentration of particulate DA in the phytoplankton and the concentration of *P. seriata* cells. Other species of *Pseudo-nitzschia* (*P.*

obtusa, *P. delicatissima*, and *P. calliantha*) were also present, but were not correlated with any toxicity, suggesting that the source of DA in the scallops was *P. seriata*. This is the same species that was shown to be responsible for the closures in more southerly parts the Gulf of St. Lawrence (Couture et al. 2001). A strong negative correlation was found between temperature and the cellular concentration of DA in *P. seriata*, for temperatures between 11.4 °C and 12.5 °C. Otherwise, no obvious conclusions could be drawn about the environmental causes of the toxic blooms at Baie au Saumon and Baie Jacques-Cartier. The contamination of molluscs by DA due to *P. seriata* in the Gulf of St. Lawrence therefore represents a risk that needs to be considered in the future. The study also concluded that the Scotia Rapid Test for ASP Toxins (developed by Joanne Jellett; [Scotia Rapid Testing Ltd.](#)) could be used by the aquaculturists as an early warning of the presence of DA in the scallops. DA was again detected in whole scallops at Baie au Saumon: 21 µg g⁻¹ on July 8, 2012 (HAEDAT [CA-12-022](#)); 58 µg g⁻¹ on June 1, 2014 (HAEDAT [CA-14-011](#)); 23 µg g⁻¹ between July 22 and August 14, 2015 (HAEDAT [CA-15-009](#)); and 23 µg g⁻¹ between June 30 and July 20, 2016 (HAEDAT [CA-16-006](#)).

The Mecatina Trough region (Lower North Shore) experienced closures of blue mussel (*M. edulis*) harvesting due to unsafe levels of DA from July 28 to August 8, 2013. The highest value (48 µg g⁻¹) was detected on July 28, at Havre des Belles Amours (ICES 2014; HAEDAT [CA-13-022](#); [Table 10](#)).

From July 22 to October 1, 2015, DA levels reached 23 µg g⁻¹ in *Placopecten magellanicus* in the in the Strait of Belle Isle (ICES 2016). These levels fluctuated, and it is speculated that the DA was not a result of a bloom in 2015, but rather from residual toxicity from the above large toxic bloom in July 2013. It is known that *P. magellanicus* requires long periods of depuration, which is consistent with these residual values. The DA values continued to be above the closure limit (23–29 µg g⁻¹) in scallops in 2016 (ICES 2017).

3.2.3 Domoic acid in Nova Scotia – Atlantic coast

The occurrence of DA is more widespread on the Canadian east coast than initially thought during the original 1987 outbreak. Since 1988, low levels of DA have been found in scallop digestive glands from Country Harbour and Whitehead ([Fig. 2B](#)), on the southeast shore of NS (M.W. Gilgan, DFO, Halifax, NS, 1992 pers. comm.). Surveys carried out in 1988 also discovered low levels of DA (up to ~30 µg g⁻¹ ww) in digestive glands of clams, mussels and scallops collected at various sites on the Scotian Shelf, Georges Bank, and in the Gulf of Maine (reported in Addison and Stewart 1989). As part of a routine monitoring for phycotoxins in the roe-on scallop fishery, extremely high levels of DA were found in sea scallops (*Placopecten magellanicus*) from Georges, German, and Browns banks ([Fig. 2B](#)) in May 1995 (reported in Douglas et al. 1997). The sample with the highest DA concentration analyzed was from Browns Bank, and showed the following tissue distribution: digestive gland (4300 µg g⁻¹), roe (55 µg g⁻¹), gills plus mantle (19 µg g⁻¹), and adductor muscle (0.62 µg g⁻¹). Similarly elevated levels were reported in a single sea scallop from the offshore banks during that time (van de Riet et al. 2006). The source of the DA in this incident is not known, but it is possible that a toxic bloom of *Pseudo-nitzschia* growing in the upper, sunlit layers of the water settled to the benthos, where it was fed upon by the scallops. No product reached the market, and all adductor muscles had DA levels well below the safety guideline. However, this incident effectively stopped the Canadian industry from further harvesting for the roe-on market in 1995.

Immediately following this episode, an increase in monitoring efforts in 1995 revealed the presence of DA near or exceeding the action level in some samples of soft-shell (*Mya arenaria*), propeller (*Cyrtodaria siliqua*) and bar (*Spisula solidissima*) clams, blue mussels (*M. edulis*), bay quahaugs (*Mercenaria mercenaria*), Stimpson's surf clam (*Mactromeris polynyma*), Jonah crabs (*Cancer borealis*), American lobster (*Homarus americanus*) hepatopancreas, and sea scallop (*Placopecten magellanicus*) digestive glands collected along the southwest coast of NS (S. Stephen, DFO, Ottawa, ON and B.G. Burns, DFO, Halifax, NS 1995 pers. comm.), resulting in a closure of that area ([Fig. 2B](#)). The great diversity in the types of organisms in which DA was found would have been missed were it not for an unusually extended sampling effort. It is not known if the two incidents were related, in spite of their geographic proximity.

In July 1996, DA was found for the first time in cultured sea scallops (*Placopecten magellanicus*), at levels up to $99 \mu\text{g g}^{-1}$ of digestive gland, from the Annapolis Basin, Digby, NS (northeast Bay of Fundy) (S. Hancock, NS Department of Fisheries & Aquaculture, 1996 pers. comm.). The juvenile scallops later depurated the toxin *in situ*, and the whole animals were marketed successfully. These episodes highlight our need to better understand the kinetics of phycotoxin uptake and depuration by different age groups of bivalve molluscs. The causative organism(s) in the above incidents was not identified, although *P. seriata* was present.

Ship Harbour (NS) was closed for the first time due to DA, when it reached $44 \mu\text{g g}^{-1}$ on July 12, 2001 (HAEDAT [CA-01-026](#)). On July 10, 2013, DA was again detected at the closure limit ($20 \mu\text{g g}^{-1}$) in Ship Harbour, and harvesting was closed (ICES 2014; HAEDAT [CA-13-004](#); [Table 10](#)). Monitoring programs have noted the presence of *P. multiseriata* and *P. pseudodelicatissima* in several sites in NS (Ship Harbour, Tor Bay, Digby, Woods Harbour, St. Margarets Bay) (Bugden et al. 1992).

DA was also detected above the closure limit (at $75 \mu\text{g g}^{-1}$) in sea scallops (*Placopecten magellanicus*) from Indian Point, Mahone Bay (NS) on December 11, 2017 (HAEDAT [CA-17-018](#)); mussels did not exceed the action limit. At nearby Snake Island, scallops peaked at $65 \mu\text{g g}^{-1}$ on December 8, 2017. On November 8, 2018, Snake Island sites were closed when DA reached $30 \mu\text{g g}^{-1}$ in scallops; levels increased to $45 \mu\text{g g}^{-1}$ on November 16. Sites at Indian Point again closed when the DA level in scallops reached $37 \mu\text{g g}^{-1}$ on December 3, 2018. Interestingly, no DA was detected in mussel samples during this event. A new closure was imposed on western Mahone Bay on January 14, 2019 ([MTN-2019-001](#)), when DA in scallops reached $30 \mu\text{g g}^{-1}$, although this is likely the same event as the December 3, 2018 closure.

3.2.4 Domoic acid in the Bay of Fundy

In response to a rapidly growing salmonid aquaculture industry, DFO initiated a Bay of Fundy phytoplankton monitoring program in 1987 (Section [9.2.2.3](#)), to provide phytoplankton data that could be used as a benchmark or baseline for comparison in future years (Wildish et al. 1988). This was expanded to include the Passamaquoddy Bay region in early 1988. Following the DA events in PE in 1987, shellfish in Atlantic Canada were being monitored for DA, and it was detected in blue mussels (*M. edulis*) and soft-shell clams (*Mya arenaria*) in Passamaquoddy Bay during August–October 1988 (Haya et al. 1991). For example, wild mussels contained $38 \mu\text{g g}^{-1}$ on September 7 ([Table 10](#)), although clams contained only $2.6 \mu\text{g g}^{-1}$. Highest values detected were $60 \mu\text{g g}^{-1}$ and $53 \mu\text{g g}^{-1}$ in blue mussels and soft-shell clams, respectively, on September 29 at the Bar Road. As a result of the Bay of Fundy phytoplankton monitoring program, the predominant phytoplankton species was

determined to be *Pseudo-nitzschia pseudodelicatissima* (Wildish et al. 1990; Haya et al. 1991), which was concluded to be the major source of the toxin (Martin et al. 1990a, 1993). Isolates of *P. pseudodelicatissima* produced low levels of DA in culture (Martin et al. 1990a). Red (horse) mussels (*Modiolus modiolus*) and the digestive gland (but not the adductor muscle) of sea scallops (*Placopecten magellanicus*) from the Bay of Fundy also contained DA (Gilgan et al. 1990). *Pseudo-nitzschia pseudodelicatissima* was a dominant species during 1988–1991, and has been present ever since (Martin et al. 1995, 2014c).

Low levels of DA have been detected in the plankton since 1988, and shellfish harvesting was again temporarily closed along the NB side of the Bay of Fundy, except for Passamaquoddy Bay, in September 1995, due to elevated levels of DA (approaching $100 \mu\text{g g}^{-1}$). Concentrations of *P. pseudodelicatissima* exceeded 10^6 chains of cells L^{-1} at the time and DA was found in net tow phytoplankton samples (J.L. Martin unpubl. data). However, it must be noted that some isolates of *P. pseudodelicatissima* from Denmark (Lundholm et al. 1994), and those from blooms that occur during late May to early June in the Bay of Fundy (J.L. Martin unpubl. data), have failed to produce detectable DA in culture. DA was generally detected during *P. pseudodelicatissima* blooms (when concentrations exceeded 10^6 chains of cells L^{-1}) that occurred from late July or August and persisted in some years to late September (Martin et al. 1998). Note that some *P. pseudodelicatissima* in the *pseudodelicatissima* complex have recently been assigned to other species (Lelong et al. 2012; Bates et al. 2018). Therefore, what was reported as *P. pseudodelicatissima* in the Bay of Fundy may have actually been another nontoxic *Pseudo-nitzschia* species. Molecular methods are required to distinguish such “cryptic” species that look similar morphologically but differ molecularly (Bates et al. 2018). Intra-specific variability in DA production may also be explained in part by the role that bacteria play in influencing the level of toxicity (Bates et al. 1995; Doucette 1995). An examination of the gene(s) responsible for DA production is the first step required to answer questions about the ability of different species and strains to biosynthesize DA. Research on this topic is in progress outside of Canada (Brunson et al. 2018; Maeno et al. 2018).

Interestingly, results from the 1987 phytoplankton monitoring showed *P. delicatissima* (which is called *P. pseudodelicatissima* in Wildish et al. 1990) at a concentration of 2.5×10^5 chains of cells L^{-1} at Frye Island, in Bliss Harbour (Wildish et al. 1988). During 1987, samples were unfortunately not collected in Passamaquoddy Bay (where DA was detected in 1988), nor in the offshore Bay of Fundy because there were no salmon aquaculture operations located at those sites. The other 1987 results, and records from the early work of Gran and Braarud (1935), suggest that *P. pseudodelicatissima* was not a new species to the Bay of Fundy. Therefore, DA may have been present in Bay of Fundy shellfish in earlier years, but went undetected, perhaps because high concentrations are required before the toxic shellfish show harm that is different from toxicity due to STX group toxins. As well, toxicity does not occur every year, and shellfish beds may have already been closed to harvesting because of unsafe levels of STX group toxins during a toxic *Pseudo-nitzschia* event.

No further closures due to DA occurred in the Bay of Fundy until mid-August 2008, when the entire coast of NB was closed, extending to the U.S. border, as a precautionary measure. At that time, *P. pseudodelicatissima* concentrations of 1.6×10^6 chains of cells L^{-1} were observed and DA levels in mussels (*M. edulis*) reached $7.1 \mu\text{g g}^{-1}$ at Lime Kiln Bay on August 12 (HAEDAT [CA-08-001; Table 10](#)) (Fernandes et al. 2014; Martin et al. 2014c). Harvesting was re-opened on September 3, 2008.

Just south of the Bay of Fundy, the Maine Department of Marine Resources (DMR) closed shellfish harvesting along the coast of Maine, after a recall of certain mussels, quahaugs, clams and European oysters in late September 2016; DA in shellfish samples from some areas exceeded $100 \mu\text{g g}^{-1}$ (Lewis et al. 2017). Clark et al. (2019b) describe the dynamics of *Pseudo-nitzschia* species in the Gulf of Maine during 2012–2016, and the regionally unprecedented levels of DA in 2016. In early October 2016, a number of shellfish harvesting areas on both the NB and NS sides of the Bay of Fundy were closed, after DA levels reached $30 \mu\text{g g}^{-1}$ in soft-shell clams (*Mya arenaria*) from Deep Brook (NS), on the southeast Bay of Fundy (HAEDAT [CA-16-020](#); [Table 10](#)). DA was also detected in soft-shell clams (*M. arenaria*) during September 9–26, 2016, at levels of 21– $30 \mu\text{g g}^{-1}$, from Clam Cove (NB), on the southwest Bay of Fundy (HAEDAT [CA-16-019](#); [CA-16-026](#)). *Pseudo-nitzschia australis* was determined to be the source of the DA in the Gulf of Maine and Bay of Fundy. Its identification was confirmed by SEM ([Digital Microscopy Facility](#)) and molecular sequencing (K. Hubbard, Florida Fish and Wildlife Conservation Commission, 2016 pers. comm.; Bates et al. 2018; Clark et al. 2019b). This is highly unusual, as this species has never before been reported on the east coast of North America.

The Bay of Fundy continues to experience brief harvesting closures due to the presence of DA. For example, on October 12, 2018, a clam Harvest Area 7 (Passamaquoddy Bay and St. Croix River, NB; Prohibition Order [MTN-2018-021](#)) was closed due to rising levels of DA. The area was re-opened on October 19, 2018 (Prohibition Order [REV-MTN-2018-021](#)). The causative diatom is unknown, as no phytoplankton samples were collected.

Prior to 2016, in addition to *P. pseudodelicatissima*, other *Pseudo-nitzschia* species identified in the Bay of Fundy include *P. americana*, *P. delicatissima*, *P. fraudulenta*, *P. multiseriata*, *P. pungens*, *P. seriata*, and *P. subpaciifica* (Kaczmarek et al. 2005, 2007, 2008; Martin et al. 2009; Martin and LeGresley 2008, 2014), some of which produce DA (Bates et al. 2018).

3.2.5 Domoic acid in Newfoundland and Labrador

In July 1994, DA was found for the first time in coastal NL. The highest concentration was only $6.3 \mu\text{g g}^{-1}$ in cultured blue mussels (*Mytilus edulis*) from Drac Bay, on the northern peninsula ([Fig. 2B](#)). However, low levels of toxin (trace to $2.2 \mu\text{g g}^{-1}$) were also detected in at least 20 other inlets around the entire coast (including Grand Banks, Notre Dame Bay, Port au Port Bay, Connaigre Bay, Badger Bay, Green Bay, Goose Arm Narrows, Bonne Bay, Dildo Run Provincial Park, and Spaniards Bay), in both cultured and wild mussels and scallops (J.C. Powell, DFO, St. John's, NL, 1994 pers. comm.). Low levels of DA were again detected in mussels during 1995: Long Arm/Pelleys Tickle/Charles Arm, Notre Dame Bay, Dildo Run Provincial Park, Great Harbour, Connaigre Bay (trace to $0.3 \mu\text{g g}^{-1}$); Northern Arm, Badger Bay ($0.2\text{--}1.6 \mu\text{g g}^{-1}$); Little Shellbird Bight (trace to $1.6 \mu\text{g g}^{-1}$); Spaniards Bay ($1.1 \mu\text{g g}^{-1}$); Grand Banks ($1.1 \mu\text{g g}^{-1}$ in surf clams); Lomond, Bonne Bay ($2 \mu\text{g g}^{-1}$); Piccadilly Bay, Port au Port Peninsula (trace to $0.4 \mu\text{g g}^{-1}$ in scallops); and Goose Arm narrows ($1.7 \mu\text{g g}^{-1}$). No harvesting areas were closed because the levels remained low. The source of the toxin in these areas is not known, although the potential DA producers *Pseudo-nitzschia seriata* (Penney et al. 2001) and *P. delicatissima* were common components of the phytoplankton assemblage in this and other inlets (C.H. McKenzie unpubl. data).

The first harvesting closure due to elevated levels of DA in NL occurred on September 11, 2001, in St. Lunenburg Bay (northeast tip of Great Northern Peninsula; [Fig. 2B](#)) (HAEDAT [CA-01-010](#)),

after DA in whole sea scallops (*Placopecten magellanicus*) collected on August 22, 2001, reached $83 \mu\text{g g}^{-1}$ (Tables 10, 11). *Pseudo-nitzschia seriata* was the suspected source organism. The following year, on June 5, 2002, the scallops continued to test positive for DA ($30 \mu\text{g g}^{-1}$), so the site remained closed (HAEDAT CA-02-012). Interestingly, blue mussels (*M. edulis*) from that location tested negative at the same time. No water samples were available to examine for the presence of *Pseudo-nitzschia* cells. It is therefore not known if the scallops had retained the DA from the previous year, or if they had continued to be exposed to recent blooms of toxic species of *Pseudo-nitzschia*. *Pseudo-nitzschia seriata* was reported from Charles Arm (Notre Dame Bay) during a 1989–1992 survey (Penney et al. 2001)

On June 14, 2002, elevated levels of DA in scallops ($49 \mu\text{g g}^{-1}$) resulted in an extended closure from Cape Norman to English Island, on the northeast tip of NL (Fig. 2B) (CFIA data; Table 10). This was in the same vicinity as the above closure in St. Lunaire Bay, in September 2001. Also near St. Lunaire Bay, DA was found in blue mussels (*M. edulis*) at $15 \mu\text{g g}^{-1}$ at Griquet Harbour, Northwest Bay, on August 6, 2013, resulting in a precautionary closure (HAEDAT CA-13-024).

A bloom of *Pseudo-nitzschia* cf. *seriata* (4.0×10^5 cells L^{-1}) was found at the Atlantic Zone Monitoring Program (AZMP) Station FC 31, near the edge of the Flemish Cap (off the coast of eastern NL; Fig. 2B), in November 2003. Highest concentrations were found at 10 m depth. Clams collected from that area were positive for DA, although the levels were not specified.

These east coast episodes clearly indicate that: 1) DA events have had important impacts on the molluscan shellfish industry; 2) phytoplankton monitoring programs must be in place prior to a toxic bloom in order to understand factors controlling its initiation and to identify the responsible organism(s); and 3) DA-producing organisms other than *Pseudo-nitzschia* species may be present.

3.3 Domoic acid on the Pacific coast

On the Pacific coast of Canada, the first recorded incidence of DA was in 1992 (Table 1). High (unspecified) levels of DA appeared in Dungeness crabs (*Metacarcinus magister*, previously called *Cancer magister*) in Quatsino Sound (northwestern Vancouver Island, BC; Fig. 2A) in late August (Forbes and Chiang 1994). DA has since been found in additional molluscan shellfish species, as well as in the Humboldt squid (*Dosidicus gigas*) and red rock crab (*Cancer productus*) (Table 12). Studies carried out in DFO's Pacific Biological Station (Nanaimo) showed that, when exposed to toxic *Pseudo-nitzschia multiseries*, Pacific oysters (*Crassostrea gigas*) and California mussels (*Mytilus californianus*) accumulated DA (Jones et al. 1995a,b; Whyte et al. 1995, 1996). Razor clams (*Siliqua patula*), unlike other bivalves examined to date, appear to retain DA for long periods, especially in the foot tissue (Horner et al. 1993; Whyte et al. 1994a). Peak concentrations of DA were $277 \mu\text{g g}^{-1}$ in crab viscera (September 17, 1992; Coal Harbour, Quatsino Sound) and $112 \mu\text{g g}^{-1}$ in razor clams (September 27, 1992; Cox Bay, near Tofino) (DFO 1992). Humboldt squid (*Dosidicus gigas*), stranded on the west coast of Vancouver Island (Chesterman Beach, Tofino) in the fall of 2009, contained low levels of DA ($0.23 \mu\text{g g}^{-1}$) in the digestive gland (Braid et al. 2012).

Table 12. Shellfish species known to become contaminated with domoic acid (DA; $\mu\text{g g}^{-1}$ ww of tissue) in coastal waters of British Columbia. The whole-animal toxin levels are the highest values found for the reference listed. The closure level is $20 \mu\text{g g}^{-1}$ ww of tissue.

Organism		DA	Location	Reference
Common name	Scientific name			
California mussel	<i>Mytilus californianus</i>	98	Barkley Sound, Vancouver Island	Whyte et al. 1994b
Dungeness crab	<i>Metacarcinus magister</i> (= <i>Cancer magister</i>)	117	Crab Island, NW Vancouver Island	DFO 1993
Geoduck clam	<i>Panopea generosa</i>	36 ¹	Kootenay Inlet	HAEDAT CA-06-022
Humboldt squid	<i>Dosidicus gigas</i>	0.23 ¹	Tofino, western Vancouver Island	Braid et al. 2012
Pacific oyster	<i>Crassostrea gigas</i>	36.3	Nanaimo, eastern Vancouver Island	Jones et al. 1995a
Razor clam	<i>Siliqua patula</i>	112	Cox Bay, western Vancouver Island	DFO 1992

¹ Digestive gland

Note that many regions along the BC coast are closed permanently due to difficulties in sampling remote areas. Data from DFO's Inspection Branch records indicate two periods when invertebrates have been contaminated with DA. Between August 1992 and May 1993, the main areas affected were along the west coast of Vancouver Island, primarily in Quatsino, Clayoquot and Barkley sounds, with additional isolated locations on Graham Island (the northernmost large island in the Haida Gwaii archipelago, formerly the Queen Charlotte Islands), the Central Coast, and the Desolation Sound area of the inner South Coast ([Fig. 2A](#)). Since September 1993, DA has become even more widely distributed, occurring in the same areas as previously noted, but also more extensively in the Haida Gwaii archipelago, on the North Coast, and in inner South Coast waters. Quatsino Sound was closed to harvesting of crabs in August 1992, and again in October 1993, due to excessive levels of DA (Forbes and Chiang 1994). As well, portions of Barkley Sound were closed to harvesting of crabs and all molluscan shellfish in November 1993. Low levels of DA from the second event persisted in razor clams from the Haida Gwaii archipelago and in molluscan shellfish from Barkley Sound until the autumn of 1994. In the autumn of 1995, harvesting of razor clams (*Siliqua patula*) was again suspended on Graham Island (Haida Gwaii archipelago) due to high (unspecified) levels of DA. The most extensive DA-producing bloom recorded in BC to that point struck the west coast of Vancouver Island in the summer of 1998, subsiding in September ([Red Tides Newsletter](#) 1999). On November 3, 2002, there was a closure at Espinosa Inlet (on the east coast of Vancouver Island), when DA reached $31 \mu\text{g g}^{-1}$ in mussels (HAEDAT [CA-02-031](#); [Table 10](#)). On June 6, 2006, there was a closure at Kootenay Inlet (Haida Gwaii archipelago), caused by DA at $36 \mu\text{g g}^{-1}$ in geoduck clam (*Panopea generosa*) viscera (HAEDAT [CA-06-022](#); [Tables 10, 12](#)). On September 18, 2012, DA above the

regulatory level ($42 \mu\text{g g}^{-1}$) was measured in razor clams (*Siliqua patula*) on Graham Island (Haida Gwaii archipelago) (ICES 2014; Russell 2016; HAEDAT [CA-12-012](#)).

In 2015, an unprecedented toxic *Pseudo-nitzschia* bloom event occurred along the North American west coast, from California to Alaska, including BC ([Fig. 4](#)), associated with anomalously warm water temperatures (a warm water mass dubbed “the Blob”) (Cavole et al. 2016; ICES 2016, 2017; Trainer et al. 2017; Ekstrom et al. 2020; Moore et al. 2020; [CBC News](#)). It was initiated nearly synchronously in late spring/early summer (April–June), with the onset of seasonal upwelling, and endured through the end of 2015 (Du et al. 2016; Trainer 2016; McCabe et al. 2016; McKibben et al. 2017). Harvesting of razor clams (*Siliqua patula*) and Dungeness crabs (*Metacarcinus magister*) was prohibited as far north as the Columbia River, WA ([Longview Daily News](#)), which borders BC. This resulted in enormous economic costs (Ritzman et al. 2018).

Compared to the long-lasting harvesting closures that occurred to the south (in WA, Oregon [OR], and California [CA]), BC harvesters were less affected. The bloom was first detected on April 29, 2015, on the west coast of Vancouver Island, and expanded along the entire coastline. It appeared to peak around June 17, with another peak in August (HAEDAT [CA-15-001](#)). DA was found in shellfish samples from sites on the west coast of Vancouver Island, from May to October, with a maximum in blue mussels (*M. edulis*) of $75 \mu\text{g g}^{-1}$ in Esperanza Inlet in mid-June (Haigh et al. 2016; HAEDAT [CA-15-001](#); [Table 10](#)). However, there were no closures due to DA at that time, as areas affected were already closed due to high STX group toxin levels. In November 2015, a sample of blue mussels from Patricia Bay (Saanich Inlet, southeast Vancouver Island) contained DA at $54 \mu\text{g g}^{-1}$ (Haigh et al. 2016; HAEDAT [CA-15-006](#)). Four of five fin whales (*Balaenoptera physalus*) (up to 120.3 ng mL^{-1} ; May 2015; Vancouver Harbour) and two humpback whales (*Megaptera novaeangliae*) (up to 612.8 ng mL^{-1} ; July 2015; Tofino and Estavan) contained DA (Savage 2017).

Water samples taken from southern Saanich Inlet at that time contained *P. australis* (identified by SEM at the [Digital Microscopy Facility](#)), and DA was also detected (V.L. Trainer, NOAA, Seattle, WA, 2015 pers. comm.; Haigh et al. 2016). Other locations in BC where concentrations of total *Pseudo-nitzschia* and particulate DA were detected are shown in McCabe et al. (2016). The algal bloom was visible in satellite imagery ([Fig. 4](#)). On September 8, 2018, a closure occurred at Miles Island (central coast of BC; [Fig. 2A](#)), with DA at $25 \mu\text{g g}^{-1}$ in geoduck clam (*P. generosa*) viscera (HAEDAT [CA-16-024](#); [Table 10](#)).

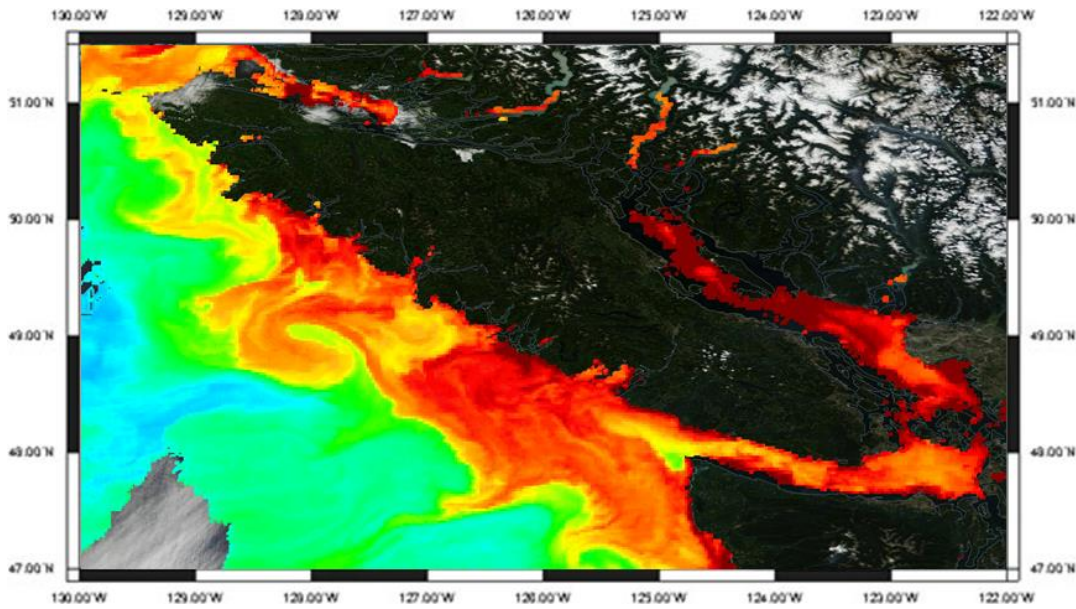


Figure 4. Satellite imagery taken on July 18, 2015, and enhanced to show the extensive algal bloom, which contains toxicogenic *Pseudo-nitzschia australis*, off the west coast of Vancouver Island. Red indicates the highest concentration of chlorophyll in the algal bloom, and green to blue indicate lower concentrations. Image sources: [NASA Earth Observations](#); [Fish Info & Services Co. Ltd.](#) See also <https://modis.gsfc.nasa.gov/>.

Although the CSSP supports the monitoring and management of harvest areas in BC and maintains a biotoxin surveillance program of commercial shellfish in some First Nations growing areas, the program does not address all self-harvesting areas. The extensive 2015 toxic bloom made it clear that risk of DA in other food sources, e.g. invertebrates and planktivorous fish, remains unknown. Therefore, a multi-stakeholder [Marine Biotoxin Workshop](#) was convened in October 2016, in North Vancouver, to better understand gaps in existing monitoring programs and the potential health risks to self-harvesters, particularly First Nations, whose diet is rich in shellfish (BC Centre for Disease Control and First Nations Health Authority 2017). These populations may face health problems when exposed to repeated diets containing low levels of DA (Grattan et al. 2016b, 2018a; Tracy et al. 2016). The importance of monitoring for phycotoxins in these areas is discussed in Section [9.2.1](#).

Based on SEM ([Digital Microscopy Facility](#)), the following *Pseudo-nitzschia* species were found at “La Perouse Project” station LB12 (offshore of southwest Vancouver Island), on July 22, 2015: *P. fraudulenta*, *P. pungens*, *P. americana*, *P. heimii*, *P. pseudodelicatissima*-complex species, and *P. delicatissima*-complex species (M. Galbraith, DFO, Institute of Ocean Sciences, Sidney, BC 2015 pers. comm.). *Pseudo-nitzschia fraudulenta* comprised 32% of all diatoms, and 19% of all microplankton, sampled in early July 2015, at the shelf break (Perry and Peña 2016). Of these species, only some strains of *P. fraudulenta* and *P. pungens* are occasionally toxic (Bates et al. 2018). There was no evidence of the highly toxic *P. australis* in samples collected at that time. However, samples taken a month earlier (June 11, 2015), at Long Beach, near Tofino on the west coast of Vancouver Island ([Fig. 2A](#)), had a mixture of diatom species, including *P. australis*. This is consistent with *P. australis* occurring in Spring Cove, WA (just south of BC) during this period (McCabe et al. 2016).

The following week, however, *P. fraudulenta* was the dominant species at the same location, with no *P. australis* observed (N. Haigh unpubl. data).

From January to June 2016, sites on the southeast side of Vancouver Island were monitored for a recurrence of the Saanich Inlet *Pseudo-nitzschia* bloom. During this time, low concentrations of several species of *Pseudo-nitzschia* were seen, including *P. australis*, *P. pungens*, and *P. delicatissima*-complex species. However, no DA was detected in shellfish samples routinely tested by the CFIA (Johnson et al. 2016a).

Other DA-producing *Pseudo-nitzschia* species (i.e. *P. multiseriata*, *P. australis*, *P. pseudodelicatissima*, *P. seriata*) and the potentially toxic *Halamphora coffeiformis* (as *Amphora coffeiformis*) have been reported in coastal BC waters (Shim 1976; Brown et al. 1988; Forbes and Denman 1991; Waters et al. 1992; Forbes and Waters 1993a,b; Taylor et al. 1994; Taylor and Haigh 1996). These species are also found to the south in WA, OR, and CA (Horner and Postel 1993). Research on the physiology of these toxigenic diatoms, as well as on the presence of DA in the marine food web, is lacking in BC.

Of particular concern is the dynamics of DA uptake and retention in Dungeness crabs (*Metacarcinus magister*). During late August to early September 1992 (HAEDAT [CA-92-010](#); 276 $\mu\text{g g}^{-1}$ hepatopancreas), and early October 1993, virtually all crabs in Quatsino Sound were contaminated, while a much lower proportion of bivalves was affected. Meanwhile, mussels in Barkley Sound and other areas of the west coast of Vancouver Island became contaminated in early September 1992 (HAEDAT [CA-92-010](#); 96 $\mu\text{g g}^{-1}$) and mid-October 1993 (Forbes and Chiang 1994).

3.4 Domoic acid in the Canadian Arctic

The following potentially toxic diatoms of the genus *Pseudo-nitzschia* are reported in the five Canadian Arctic regions (Hudson Bay system, eastern Arctic, western Arctic, Canadian Archipelago, and Canada Basin) either in the sea ice and/or in the plankton ([Appendix 1](#)).

Pseudo-nitzschia arctica has been described and reported for the first time in phytoplankton from Barrow Strait in the Canadian Archipelago and in the Beaufort Sea (Percopo et al. 2016).

Pseudo-nitzschia delicatissima has been reported in sea ice and plankton from all five Canadian Arctic regions, except for no record in plankton from the Canada Basin (Davidson 1931; Seidenfaden 1947; Bursa 1961a, 1971; Adams 1975; Thomson et al. 1975; Sekerak et al. 1976a, 1979; Bain et al. 1977; MacLaren Atlantic Limited 1978; Hsiao 1979a, 1980, 1983; MacLaren Marex 1979a,b; Cross 1980, 1982, 1987; Anderson et al. 1981; Grainger and Hsiao 1982; Pett et al. 1983; Poulin et al. 1983; Booth 1984; Hsiao et al. 1984; Hsiao and Pinkewycz 1985a,b, 1987; Pinkewycz and Hsiao 1987; Pinkewycz et al. 1987; Woods and Smiley 1987; Percy et al. 1992; Simard et al. 1996; Harvey et al. 1997; von Quillfeldt 2000; Lovejoy et al. 2002; Melnikov et al. 2002; Riedel et al. 2003; Róžańska et al. 2009; Mather et al. 2010; Simo-Matchim et al. 2017; M. Poulin unpubl. data).

Pseudo-nitzschia granii has been reported only once in plankton from Baffin Bay by Lovejoy et al. (2002), whereas *P. obtusa* has been reported in plankton from the eastern and western Arctic

as well as the Canadian Archipelago (von Quillfeldt 2000; Simo-Matchim et al. 2017; Crawford et al. 2018; M. Poulin unpubl. data).

Pseudo-nitzschia pseudodelicatissima has been reported in sea ice from the Canada Basin, the eastern and western Arctic, and the Canadian Archipelago (Simard 2003; Róžańska et al. 2008 [although Percopo et al. (2016) questioned its identification], 2009; Mather et al. 2010) and in plankton from all Canadian Arctic regions, except in the Canada Basin (Percy et al. 1992; von Quillfeldt 2000; Róžańska et al. 2008, 2009; McLaughlin et al. 2009; Mundy et al. 2011; Simo-Matchim et al. 2017; M. Poulin unpubl. data).

Pseudo-nitzschia pungens has been reported in sea ice only once from Franklin Bay (Róžańska et al. 2009) and in the plankton from the eastern Arctic and the Hudson Bay system (Bursa 1961a,b, 1971; Hsiao 1983; Percy et al. 1992).

Pseudo-nitzschia seriata has been reported in sea ice from all Canadian Arctic regions (Thomson et al. 1975; Hsiao 1979a,b,c, 1980, 1983; Cross 1980, 1982, 1987; Grainger and Hsiao 1982; Hsiao et al. 1984; Woods and Smiley 1987; Hopky et al. 1994a,b; Kikuchi-Kawanobe and Kudoh 1995; Melnikov et al. 2002; Róžańska et al. 2009), and in plankton from all Canadian Arctic regions, except from the Canada Basin (Mann 1925; Davidson 1931; Polunin 1934; Grøntved and Seidenfaden 1938; Seidenfaden 1947; Bursa 1961a,b, 1971; Adams 1975; Thomson et al. 1975; Foy and Hsiao 1976; Hsiao 1976, 1983; Sekerak et al. 1976a,b, 1979; Bain et al. 1977; MacLaren Atlantic Limited 1977, 1978; MacLaren Marex 1979a,b; Hsiao and Trucco 1980; Anderson et al. 1981; Grainger and Hsiao 1982; Hsiao and Pinkewycz 1983, 1984, 1985a,b, 1987; Pett et al. 1983; Hsiao et al. 1984; Pinkewycz and Hsiao 1987; Pinkewycz et al. 1987; Percy et al. 1992; Simard et al. 1996; Harvey et al. 1997; von Quillfeldt 2000; Lovejoy et al. 2002; McLaughlin et al. 2009; Mather et al. 2010; Brown et al. 2014; Simo-Matchim et al. 2017).

Pseudo-nitzschia turgidula has been reported in sea ice once from Franklin Bay (Róžańska et al. 2009) and in plankton from the western Arctic and the Canadian Archipelago (McLaughlin et al. 2009; M. Poulin unpubl. data) ([Appendix 1](#)).

Arctic strains of *P. seriata* and *P. obtusa* have been shown to produce DA (Hansen et al. 2011; Harðardóttir et al. 2015). DA was not detectable in *P. arctica*, which has so far not been shown to be toxigenic (Bates et al. 2018; [Wikipedia](#)). Earlier research had identified *P. seriata* in the bottom-sea ice community of Resolute Bay, Nunavut (Smith et al. 1994b), although its ability to produce DA was not addressed. It is possible that the sea ice community acts as a seed population for water column blooms, or that toxins are produced within the ice and transferred to the benthic or pelagic food webs. Species of *Pseudo-nitzschia* (*P. australis*, *P. calliantha*, *P. granii*, *P. pseudodelicatissima*, and *P. pungens*) are reported in Russian Arctic waters of the Barents Sea and White Sea (Vershinin and Orlova 2008).

During a DFO research program ([Marine Environmental Assessment of the Canadian Beaufort Sea](#)), a bloom of *P. seriata* was observed in coastal waters of the Beaufort Sea in the summer of 2014 (A. Niemi, DFO, Freshwater Institute, Winnipeg, Manitoba 2015 pers. comm.). More recently, as part of the Canadian initiative, the Beaufort Sea Regional Ecosystem Assessment ([BREA](#)), *P. seriata* was again observed in coastal waters of the Canadian Beaufort Sea (C. Michel unpubl. data). At the same time, the presence of low levels of DA in scallops (*Similipecten greenlandicus*)

was observed during this bloom (W.A. Rourke unpubl. data). These findings, and the evidence of accumulation of toxins in marine mammals in the Beaufort Sea (Lefebvre et al. 2016), raise growing concerns about the occurrence of toxic algal blooms and the transfer of phycotoxins to harvest species in Canadian Arctic waters.

The development of DA-producing algal blooms, their fate, and the dynamics underlying toxin-production in the Arctic are just starting to garner the attention of the scientific community. Recent studies in eastern Baffin Bay show that the copepod *Calanus* feeding on *Pseudo-nitzschia* do not discriminate between toxic *P. seriata* and nontoxic *P. obtusa* cells. They are thus capable of retaining DA in their tissue and transferring it through the food web (Tammilehto et al. 2012; Harðardóttir et al. 2015). DA was also shown to affect the escape response of the copepods *Calanus hyperboreus* and *C. glacialis*, two pivotal species in arctic marine food webs (Harðardóttir et al. 2018). The DA was not harmful to these copepods, as it did not affect their grazing or reproduction (Tammilehto et al. 2012; Harðardóttir et al. 2015; Miesner et al. 2016).

DA was reported in 13 species of marine mammals from U.S. waters of the Beaufort Sea, which obtained the toxin from their planktivorous prey (Lefebvre et al. 2016). The greatest prevalence of DA was in bowhead whales (*Balaena mysticetus*; in 68% of the individuals) and harbor seals (*Phoca vitulina richardsi*; 67%); the highest concentrations were in Pacific walruses (*Odobenus rosmarus*), similar to those detected in California sea lions that exhibited clinical signs of DA toxicosis (seizures) off the coast of central CA (Scholin et al. 2000). Additionally, fetuses from a beluga whale, a harbor porpoise and a Steller sea lion in the Alaskan Arctic contained detectable concentrations of DA, documenting maternal toxin transfer in these species. These results of DA occurrence in the Arctic deserve more scientific attention. We must learn more about how this toxin may impact the health of Arctic Canadian marine mammals and possibly coastal communities through traditional harvest.

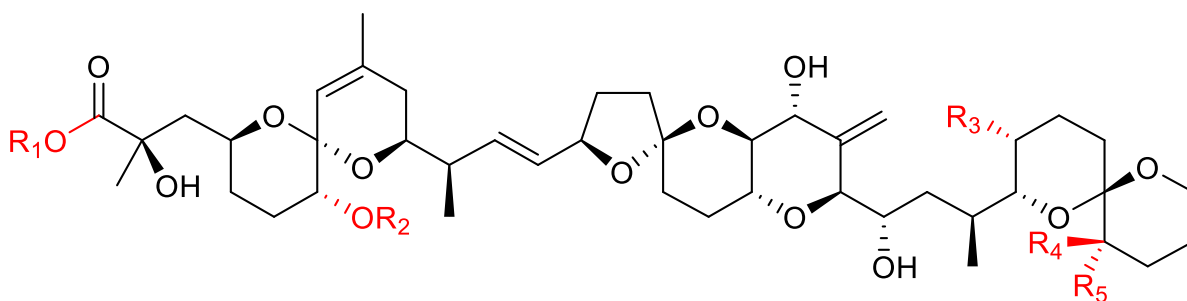
Unique conditions, such as the long summer photoperiod that occurs in the Arctic, are shown experimentally to increase toxin production (Fehling et al. 2005), as does nutrient depletion following intense phytoplankton blooms (Bates et al. 2018). These factors, as well as the role of pivotal species in marine food webs, need to be taken into account to understand phycotoxins and their transfer to higher trophic levels in the Arctic. Because phycotoxin poisonings are generally underreported, the actual impact of DA on human and marine ecosystem health, including in the Arctic, is unknown (Lewitus et al. 2012).

4.0 Okadaic Acid Group Toxins (Diarrhetic Shellfish Poisoning; DSP)

4.1 Description of okadaic acid group toxins and their biological effects

Diarrhetic Shellfish Poisoning (DSP) is a severe gastrointestinal illness, whose symptoms appear between 30 min to a few hours after ingesting contaminated seafood (Sosa and Tubaro 2016) and include diarrhea (92%; hence the name), nausea (80%) and vomiting (79%) (Tubaro et al. 2008a). The symptoms usually disappear within a few days. Unlike PSP, DSP has not resulted in deaths in Canada nor elsewhere worldwide, although the illness is unpleasant for the consumer and the shellfish industry is negatively impacted when humans become intoxicated (FAO 2004; Lawrence et al. 2011; Fessard 2014; Johnson et al. 2016b; [WHOI](#)). As well, some of its toxins have other concerns to human health, as reviewed by Valdiglesias et al. (2013) and discussed below.

DSP was first identified as a problem in Japan in the 1970s (Murata et al. 1982; Yasumoto et al. 1985), but was later also found in northern Europe and South America (reviewed in Reguera et al. 2014; Reguera and Blanco 2019). DSP is caused by OA group toxins (Toyofuku 2006), which are composed of derivatives of three primary lipophilic, acid polyether toxins ([Structure 3](#)): okadaic acid (OA), dinophysistoxin-1 (DTX1), and dinophysistoxin-2 (DTX2). OA was initially isolated from the sponge *Halichondria okadai* from which it gets its name (Tachibana and Scheuer 1981). However, OA group toxins have now been identified in dinoflagellate species of the genera *Dinophysis* and *Prorocentrum*, and in contaminated shellfish (Marr et al. 1992a,b; Hu et al. 2017; Sibat et al. 2018). Dinophysistoxin-3 (DTX3) is a series of 7-*O*-fatty acid ester of OA, DTX1 or DTX2 produced during metabolism in marine organisms such as shellfish and crabs (Yasumoto et al. 1985; Torgersen et al. 2005). DTX3 is not a discrete compound and can refer to esters with variable chain length and saturation. Henceforth, we shall use “OA/DTX acyl esters” rather than DTX3. It should be noted that Yasumoto et al. (1985) originally referred to DTX3 as acyl esters of DTX1 specifically, and did not include acyl esters of OA or DTX2. However, it is now generally accepted that DTX3 covers acyl esters of OA, DTX1, and DTX2. Other dinophysistoxins (DTX4, DTX5a, and DTX5b) are more complex derivatives of OA, and are reported in *Prorocentrum maculosum* (Macpherson et al. 2003). Diol esters of OA group toxins (Hu et al. 1992b, 2017) form readily due to enzymatic action on sulfated dinophysistoxins such as DTX4, DTX5a, and DTX5b (Quilliam et al. 1996; Windust et al. 1997). See the [CFIA website](#) (CFIA 2014a) for more information.



Structure 3. The structure of okadaic acid and its analogues. The principal toxin, okadaic acid, has the substituents R₁, R₂, R₄, R₅ = H and R₃ = CH₃.

After detecting DTX1 in Canadian waters in 1990 (Section [4.4.1](#)) (Quilliam et al. 1991, 1993), an interim closure level of 1 µg combined DTX1 plus OA per g of digestive gland was established in 1994. As of July 2011, shellfish areas are now closed when OA group toxin (OA and/or DTX1, singly or in combination) levels reach 0.2 µg g⁻¹ whole tissue (CSSP 2019). Health Canada (2016b) indicates a closure limit for OA group toxins (sum of OA plus DTX1, DTX2 and OA/DTX acyl esters) not to exceed 1 µg g⁻¹ in bivalve shellfish digestive tissue, or 0.2 µg g⁻¹ in bivalve shellfish edible tissue ([Table 3](#)); these values are still under review (Health Canada 2016). Inclusion of OA/DTX acyl esters was prompted by high levels of this toxin association with the August 2011 closure in BC (Taylor et al. 2013; Section [4.5](#)). The CFIA has now more clearly specified these action levels (CFIA 2017b), and this is reflected in [Table 3](#). Note that the CSSP (2019) indicates the toxins in “whole tissue”, whereas the CFIA (2017b) indicates this as “edible tissue”. In Canada, CFIA data show that OA group toxins have been found at levels up to 5.2 µg g⁻¹ whole tissue at 153 sites on both coasts: 88 in BC ([Fig. 5A](#)), 16 in NB, 9 in NL, 22 in NS, 5 in PE, and 13 in QC ([Fig. 5B](#)).

Although OA group toxins have not led to any documented human fatalities (FAO 2004; van Egmond 2011), OA has been reported as a tumour promoter (Fujiki et al. 1988; Haystead et al. 1989). It also inhibits protein phosphatases, which are important modulators of enzyme activity and cell signaling pathways (reviewed in Twiner et al. 2016); this characteristic led to one method of detection (Section 4.3). Other research has revealed that OA is cytotoxic to lymphocyte cells (Martín-López et al. 2012) and alters the angiogenesis in developing chick embryos (Jiao et al. 2017). Exposure of mussels (*Mytilus galloprovincialis*) to *Prorocentrum lima* cells that produced OA induced early genotoxicity in hemocytes, as a consequence of oxidative DNA damage, and DNA damage to gill cells (Prego-Faraldo et al. 2016), indicating potential ecosystem disruption. Sublethal doses of OA group toxins stimulate the secretion of pro-inflammatory cytokines and chemokines in macrophages (del Campo et al. 2017). In the same study, DTX1 was found to be 10 times more potent than OA, whereas they are now considered equivalent in international legislation (European Commission 2002; EFSA 2008b). This is of particular significance in Canada, where DTX1 is the primary OA group toxin observed in shellfish. Furthermore, del Campo et al. (2017) recommended that “the safe limit regulation should be changed to DSP toxins zero tolerance in the shellfish to be consumed by humans”. Their results raise questions about current regulatory levels in Canada (see below), which are under review by Health Canada (Table 3).

The Canadian action level ($0.2 \mu\text{g g}^{-1}$) is similar to the European Commission regulatory level of $160 \mu\text{g kg}^{-1}$ ($0.16 \mu\text{g g}^{-1}$) for OA group toxins and PTXs (Section 5.1) in combination, in the “whole body or any part edible separately” (European Commission 2002; EFSA 2008b), a value that the CFIA must follow to export product (CFIA 2015a). The action level of $160 \mu\text{g kg}^{-1}$ for OA group toxins is also specified by the U.S. Food and Drug Administration (FDA 2012) and in the United Nations Food and Agriculture Organization (FAO) Codex Alimentarius International Food Standards (FAO 2008). DTX2 undergoes more thermal degradation than does OA (McCarron et al. 2008). Therefore, a greater reduction in toxicity can be expected when the mussels are heated during the industrial canning procedure, if DTX2 is the main toxin (Blanco et al. 2016). An important new aspect to consider for assessing toxicology is the exposure to mixtures of lipophilic phycotoxins, for example co-exposure of OA in combination with PTX2 or YTX (Alarcan et al. 2018).

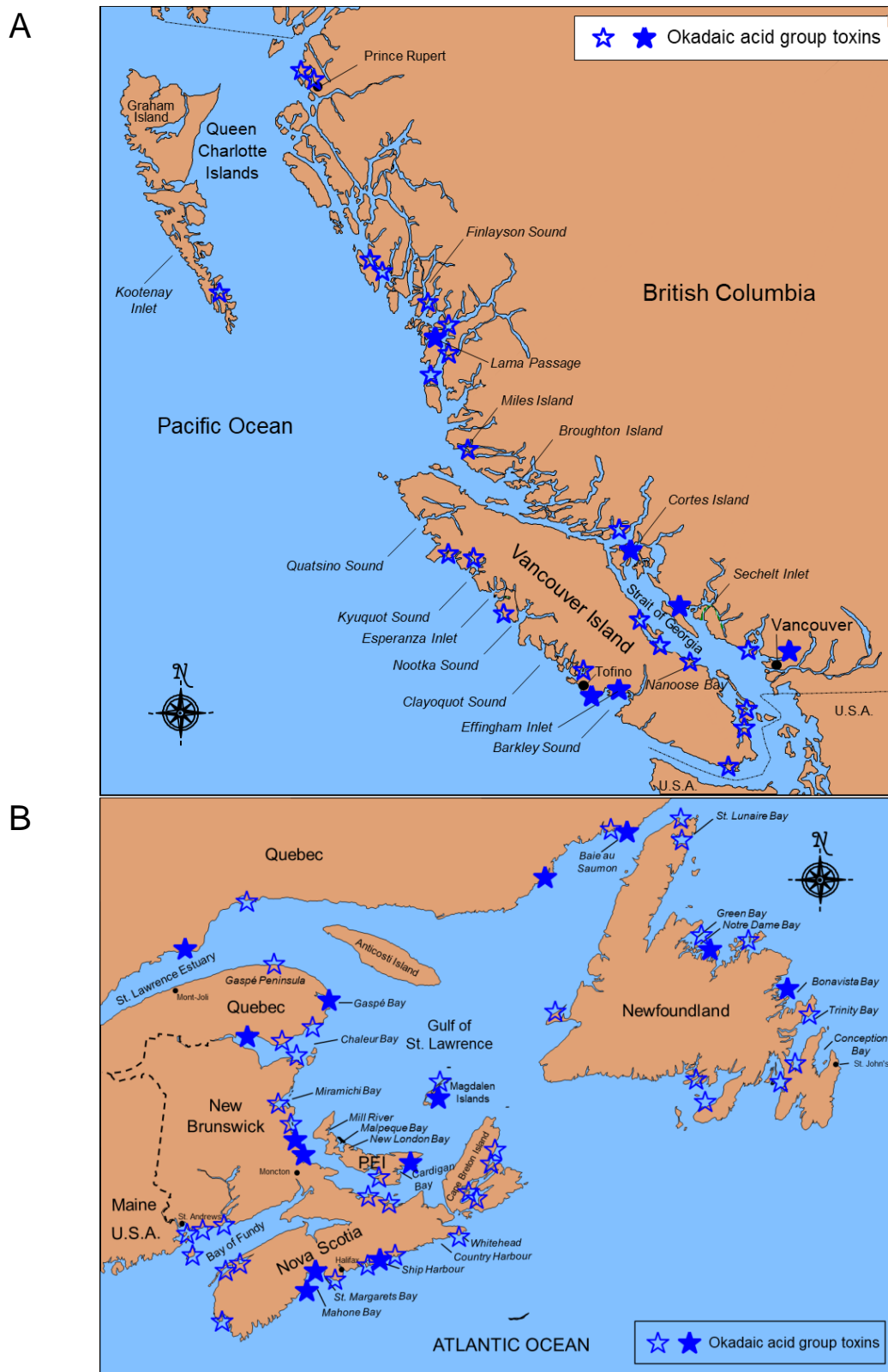


Figure 5. Location of okadaic group toxins along the coast of A) British Columbia; B) Atlantic Canada. Open symbols: below the regulatory action level ($0.2 \mu\text{g g}^{-1}$ in edible tissue); closed symbols: shellfish harvesting area closed due to exceeding the regulatory action level.

4.2 Organisms that produce okadaic acid group toxins

Worldwide, two species of the dinoflagellate genus *Prorocentrum* (reviewed in Hoppenrath et al. 2013), at least ten species of *Dinophysis*, and two of *Phalacroma* (reviewed in Reguera and Pizarro 2008; Reguera et al. 2012, 2014; Blanco 2018) have been shown to produce and/or contain OA group toxins and/or PTXs. On the Canadian east coast (Section 4.4), *Prorocentrum lima* is a confirmed producer of OA group toxins (Hu et al. 1992b, 1995a; Marr et al. 1992a,b; Jackson et al. 1993; Luu et al. 1993; McLachlan et al. 1994; Bauder et al. 1996, 2001; Quilliam et al. 1996; Cembella et al. 1997; Pan et al. 1999). Although not found in Canadian waters, *Prorocentrum maculosum* (formerly *P. concavum*) was studied by Canadian researchers at NRC (Halifax, NS). They discovered that it produced a new toxin, prorocontrolide B, in addition to OA group toxins (Hu et al. 1996b). This more polar toxin was in a butanol-soluble fraction, rather than in the lipid-soluble fraction, in which OA group toxins are extracted. Furthermore, it behaves as a “fast-acting toxin” (Section 6.1), differentiating it from OA group toxins. NRC researchers then studied the biosynthesis of DTX5a and DTX5b by this same species (Macpherson et al. 2003).

Strains of *Dinophysis acuta* from New Zealand (Miles et al. 2004; Suzuki et al. 2004) and northwest Spain (Sibat et al. 2018) contained high levels of PTXs, but little OA. Instead, they produced diol esters of OA, reported for the first time in *D. acuta*. Indeed, the cellular toxin content and profiles of OA group toxins and PTXs (Section 5.1) were different among nine species of *Dinophysis*, and even strains of the same species, from Japan (Uchida et al. 2018a). The annual cycle of *D. acuta* populations, their production of toxins, and accumulation of toxins in blue mussels (*M. edulis*) from Scotland is discussed by Swan et al. (2018). Note that *D. rotundata* is currently regarded as a taxonomic synonym of *Phalacroma rotundatum* ([WoRMS](#)).

Until recently, confirmation of OA group toxin production by *Dinophysis* species in culture, as well as studies of the ecophysiology of toxin production, was hampered because these organisms could not be successfully cultured. It was then discovered, during the first successful culture of *Dinophysis*, that *D. acuminata* required a ciliate prey, *Mesodinium rubrum* (Section 7.3.4), which in turn must be fed the cryptophyte *Teleaulax amphioxeia* (Park et al. 2006). This discovery allowed studies of the effects of environmental factors and growth stage on the production of these toxins by *D. acuta* in culture (e.g. Fux et al. 2011; Nielsen et al. 2013; Tong et al. 2015a). As well, *D. acuminata* was fed *M. rubrum* that was maintained with another cryptophyte, *Geminigera cryophila* (Tong et al. 2011, 2015a) for similar studies (see below). The ability to grow *D. acuminata* with *M. rubrum* in culture has also allowed studies on its inorganic and organic nitrogen, and phosphate, requirements (Tong et al. 2015b). This provided support for the hypothesis that accelerated nitrogen loading in coastal ecosystems can promote toxic blooms of *D. acuminata* and may be partly responsible for the recent expansion of these species across North America (Hattenrath-Lehmann and Gobler 2015). In addition, the ability to grow *D. acuta* allowed a study of the accumulation and depuration of OA group toxins by blue mussels (*M. edulis*) (Nielsen et al. 2016). The laboratory study confirmed previous field observations by demonstrating that a concentration of only 75 *D. acuta* cells L⁻¹ of a moderately toxic culture is enough to make 60-mm long mussels exceed the regulatory threshold within 10 days. Lee et al. (2016) reviewed studies on the mechanism of OA group toxin production. Galician strains of *D. acuminata* and *D. acuta* feeding on two strains of *Mesodinium* (which were feeding on the cryptophyte *Teleaulax amphioxeia*) had different toxin and metabolomic profiles, and also varied between the different prey organisms (García-Portela et al. 2018a). This allowed a study of the effect of light on growth, cellular toxin content and photosynthesis of these two *Dinophysis* species

(García-Portela et al. 2018b). *Dinophysis acuminata* cultures were scaled up to study their growth and toxin production in 8-L photobioreactors (Hernández-Urcera et al. 2018). Adding the lysate of ciliate prey (*Mesodinium rubrum*) in predator-prey co-incubations increased the growth and toxicity (OA and DTX1 production) of *D. acuminata* (Gao et al. 2019b). Studies on mechanisms of action and what components of the lysate are responsible will help to predict and manage toxic blooms. Wolny et al. (2020) identified *D. acuminata*, *D. norvegica*, and a “small *Dinophysis* sp.” in samples from West Jeddore (NS) and the mid-Atlantic region of the U.S.

4.3 Methods for detecting and quantifying okadaic acid group toxins

The analytical chemistry of OA group toxins has turned out to be complex, which explains the difficulty in obtaining reliable data about their prevalence. It has taken many years to understand all known derivatives and analogues, and therefore to develop analytical methods that can reliably detect them (FAO 2004). Much of this work has been carried out at the NRC in Halifax, NS (Plesance et al. 1990; Hu et al. 1992a,b, 1995a,b; Quilliam 1995, 2003c; Quilliam et al. 1996; Wright and Cembella 1998; Doucet et al. 2007). In particular, modern analytical methods usually include a hydrolysis step to convert any ester derivatives to release free OA group toxins during sample preparation (Mountfort et al. 2001; Doucet et al. 2007; McCarron et al. 2014a). It is therefore important to specify when, and by which method, the OA group toxin has been identified, in order to assess the reliability of their detection in Canadian waters. In general, the certainty and accuracy of the identification method increase in the following order: immunological (including Lateral Flow Immunoassays; LFIs) and receptor-binding assays (Lawrence et al. 1998b; Cembella et al. 2003; Laycock et al. 2006; Sassolas et al. 2013a,b; Leonardo et al. 2018a); intraperitoneal injection mouse bioassay; phosphatase inhibition assay (Holmes 1991; Luu et al. 1993); HPLC; liquid chromatography-mass spectrometry (LC–MS); then liquid chromatography-tandem mass spectrometry (LC–MS/MS) (Quilliam 2003c; McCarron et al. 2014a). Depending on the assay, therefore, some of the newer derivatives and analogues of OA group toxins may have been missed in the earlier studies, described in Section [4.4.1](#).

The analytical method currently used by the CFIA to quantify OA group toxins is LC–MS/MS. It should be noted that the analysis of lipophilic shellfish toxins (LSTs) was initially performed only on the digestive gland of molluscan shellfish in order to increase the sensitivity. As instrumentation became more advanced with better sensitivity, testing shifted to whole tissue. This change happened on April 26, 2007; thus, results prior to this date were based on digestive gland, and those after were based on whole tissue. However up to 2011, in order to meet EU export requirements, mouse bioassays continued to be performed on digestive glands only (e.g. for Ship Harbour). After 2011, the LC–MS/MS method became the reference method. Currently, all routine testing is performed on whole tissue (for STX group toxins, DA, and LSTs). However, there are some exceptions to this on the Pacific coast, e.g. geoduck, for which the tissue analyzed is the “gutball” or viscera. Previously, some scallops were analyzed as digestive gland and remaining tissue, but as of May 10, 2019, whole scallops are now analyzed (CFIA 2019).

A comparison of the available rapid test methods to detect these toxins is given in Johnson et al. (2016b), including a lateral flow immunochromatography test developed in NS ([Scotia Rapid Testing Ltd.](#)). The Biotoxin Metrology group of the NRC, in Halifax (NS), produces certified calibration solutions and mussel tissue reference materials for the determination of several OA group toxins

by chemical analysis methods ([NRC Certified Reference Materials](#); [Wikipedia](#); Quilliam 2006; Beach et al. 2016b; McCarron et al. 2016, 2017).

Note that on February 5, 2015, the EU agreed to modify the procedure to determine the presence of LSTs in mussels after they are processed ([Fish Info & Services](#)). This was prompted by the finding that, although LST levels in the fresh product may be below those allowed ($160 \mu\text{g kg}^{-1}$), they increase once the mussels are cooked or sterilized for processing because of dehydration. The solution proposed was to rehydrate the samples before analysis, adding back liquid in different proportions depending on the processing method.

A brief discussion of the advantages and disadvantages of the above methods for detecting OA group toxins follows (also reviewed by Prego-Faraldo et al. 2013) because this is also relevant for interpreting the accuracy of each report in Canadian waters. The intraperitoneal injection mouse bioassay is not inherently specific, and its reliability is dependent on the chemical extraction and fractionation methods (methanolic extraction and chloroform partitioning was used for OA group toxins in Canada [Gilgan et al. 1995]). Its use has been severely criticized in the UK's DSP monitoring program due to "procedural artefacts", lack of proper validation and animal rights issues (Combes 2003). There are still questions about the effect of using male versus female mice (Suzuki 2013). A further complication was observed when mouse deaths occurred after intraperitoneal injection with methanol extracts of the hepatopancreas of blue mussels (*M. edulis*) from Ship Harbour and Wine Harbour, eastern NS (Lawrence et al. 1994a). These were later shown to be negative for OA group toxins, STX group toxins, and DA. It was determined that certain polyunsaturated fatty acids, not necessarily of algal origin, were mainly responsible for the toxicity. These fatty acids are of concern because they can yield positive results in the DSP mouse bioassay and because their relationship to human illness is not known. Because of the lack of specificity, and harm to mice, the mouse bioassay is no longer used to detect OA group toxins (FAO 2004).

Two immunoassay kits were used in the early 1990s: UBE DSP Check kit (SCETI Laboratories Ltd., Tokyo, Japan) and the ELISA Okadaic Acid test kit (Rougier Biotech Ltd., Montreal, QC). The UBE kit (which is apparently no longer available), was specific for OA, and detected DTX1 only 1/3 to 1/5 times as well as OA (M.W. Gilgan, DFO, Halifax, 1995 pers. comm.). The Rougier kit was also specific for OA, and was not intended for quantification of DTX1 (according to the manufacturer's instructions), although it detected DTX1 with a 20-fold lower affinity (Chin et al. 1995). Rougier Bio-tech Ltd. was purchased by Erfa Biotech (Val-Morin, QC) in 1999, and still produces an [ELISA kit for quantitation of OA](#). Early versions of these kits, therefore, are not very specific for DTX1, the toxin that occurs most frequently in Canadian waters. Furthermore, these kits cannot quantify the individual toxins when there is a mixture of OA and DTX1, and may thus underestimate total OA group toxins in shellfish (Morton and Tindall 1999). The immunoassay may be more useful in European waters where OA is the principal toxin, although the European profile also includes DTX2, which may limit their effectiveness if the cross-reactivity to DTX2 is too low. Finally, although there may be excellent correlations between the immunoassay and LC-MS results (Chin et al. 1995; Morton and Tindall 1999), there are also undocumented reports of false positive responses when the test kits were used for mussel extracts, phytoplankton cultures, and phytoplankton field samples. Nevertheless, the antibody that recognizes OA (previously supplied by Rougier Biotech Ltd.) was used to locate where OA is stored within *Prorocentrum lima* cells (Zhou and Fritz 1994) and to immunolabel DSP-producing dinoflagellates (Lawrence and Cembella 1999). A rapid test kit for "DSP toxins" (OA group toxins not specified) based on lateral flow immunochromatography, for

use in the field, was developed by Dr. Joanne Jellett and the NRC (Halifax, NS), and is now sold by [Scotia Rapid Testing Ltd.](#); it is described by Laycock et al. (2006).

Another approach to detect OA group toxins is to use a protein phosphatase inhibition assay (PPIA). An early PPIA required that the sample be first passed through an HPLC column to obtain fractions containing OA and DTX1 (Holmes 1991). These fractions were then tested radiometrically for their ability to inhibit two protein phosphatase enzymes. The method is complex, requiring numerous steps, including several difficult enzyme preparations. It detects any compound (including microcystin-LR, Section [10.3.1](#)) that inhibits the phosphatase enzymes, which makes it prone to false positives in the absence of OA group toxins. However, this is mitigated by the requirement of the initial HPLC step. False negatives could also occur with improper enzyme preparations. Naturally occurring esters of OA, such as DTX3 (OA/DTX acyl esters) and OA diol-esters, do not act as phosphatase inhibitors, so are not detected by this method (Hu et al. 1992b), unless a base hydrolysis is used to release the parent toxins. Various improvements to the PPIA have been published and even tested in interlaboratory trials (Tubaro et al. 1996; Mountfort et al. 1999; González et al. 2002). A newer PPIA produced by [ZeuLab](#) is simpler to use (Smienk et al. 2012).

An instrumental method using a pre-column derivatization with the anthryldiazomethane (ADAM) reagent, followed by HPLC with fluorescence detection, determines OA, DTX1 and DTX2 (Quilliam 1995). However, the ADAM reagent is unstable and subsequent reaction cleanup is time consuming. Furthermore, the resulting chromatograms are often very complex, due to reagent impurities and the formation of numerous fluorescent derivatives (Pleasant et al. 1990), leading to the possibility of false positives. These problems can be minimized with an *in situ* reaction procedure developed for use with mussel tissue (Quilliam et al. 1998). Since the technique is cumbersome for routine analyses, it is rarely used now.

LC–MS/MS is by far the most reliable method to detect OA group toxins. It allows trace detection and unambiguous identification of polar toxins by electrospray ionization (also known as “ion-spray”) mass spectrometry, without the need for sample cleanup (Pleasant et al. 1990; Quilliam 1995; van de Riet et al. 1995; Suzuki and Quilliam 2011). Although the initial costs of the instrumentation are high, these are soon recovered due to the efficiency of the technique. The LC–MS/MS method had been routinely used for Canadian samples (Quilliam 2003c), but it has recently been replaced by a faster ultra performance liquid chromatography-tandem mass spectrometry (UPLC–MS/MS) method ([Appendix 3](#)). Broader availability of high-resolution mass spectrometry (HRMS) systems have added a further level of selectivity to analyze marine algal toxins, including OA group toxins (e.g. Blay et al. 2011; Zendong et al. 2015).

It is important to note that esters of OA group toxins, such as the diol-esters and OA/DTX acyl esters, must be taken through a base hydrolysis to release parent toxins for instrumental analysis (Suzuki and Yasumoto 2000; Mountfort et al. 2001) and even for assays. OA/DTX acyl esters can induce illness in people, presumably due to hydrolysis to the parent toxins in the gastrointestinal tract (García et al. 2005). In some shellfish samples, most of the OA group toxins may be in the form of OA/DTX acyl esters, as illustrated by a mass toxicity event in Norway in 2002 due to brown crabs (*Cancer pagurus*) (Torgersen et al. 2005). It is possible to analyze the OA/DTX acyl esters directly with LC–MS/MS (Quilliam 1995; Suzuki and Quilliam 2011), but the quantitative analysis is complicated to carry out because of the great complexity of analogues and the unavailability of calibration standards.

The above detailed examination of OA group toxin detection techniques emphasizes the need to carefully interpret the analysis of these toxins (including all derivatives) in field samples. Great care must be exercised in evaluating the data because each detection method carries with it a different degree of certainty about the presence and identity of the toxins. As discussed below, the discovery of novel OA group toxins, and the dinoflagellates that produce them, has gone hand-in-hand with the development of new analytical methods.

4.4 Okadaic acid group toxins on the Atlantic coast

4.4.1 Okadaic acid group toxins in Prince Edward Island, Nova Scotia, and New Brunswick

Closures caused by the presence of OA group toxins for the east and west coasts of Canada, and their HAEDAT event numbers (when available), are given in [Table 13](#). The HAEDAT database also contains records of high concentrations of potentially toxic *Dinophysis* spp.: [CA-90-007](#); [CA-90-002](#); [CA-90-004](#); [CA-92-003](#); [CA-92-009](#); [CA-91-004](#); [CA-92-007](#); [CA-92-006](#); [CA-96-003](#); [CA-96-004](#); [CA-97-007](#); [CA-97-006](#); [CA-11-002](#).

In June to August, 1989, DTX1 was recorded for the first time in Canadian mussels and *Prorocentrum lima* (from eastern PE), using an HPLC-linked PPIA and confirmed by LC–MS (Holmes 1991; Luu et al. 1993) ([Table 1](#)). In September 1989, a small population of *Dinophysis acuminata* was shown, by the phosphatase-inhibition assay, to be associated with significant amounts of what was probably DTX1 (Smith et al. 1991). No other such events occurred on PE until June 21, 2018, when mussels (*M. edulis*) from Boughton River ([Fig. 5B](#)) were shown to contain $0.32 \mu\text{g g}^{-1}$ DTX1 and $0.24 \mu\text{g g}^{-1}$ DTX1 esters (HAEDAT [CA-18-002](#); [Table 13](#)). Although not proven OA group toxin producers in Canadian waters, *D. caudata* and *D. norvegica* were reported from Malpeque Bay, PE (Sita Devi and Lakshminarayana 1989).

During July–August 1990, OA was detected with an immunological test kit (but not confirmed by any other means) in the Bedford Basin (near Halifax/Dartmouth, NS), and was associated with a bloom of *D. norvegica* (Subba Rao et al. 1993). Scallops experimentally contaminated with OA from that bloom depurated the toxin slowly, according to test kit results. The photosynthetic characteristics of *D. norvegica* blooms in Bedford Basin were studied on five occasions, between July 25 and August 3, 1990 (Subba Rao and Pan 1993).

Table 13. Closures of shellfish harvesting caused by levels of okadaic group toxins above $0.2 \mu\text{g g}^{-1}$ (ww) on the east and west coasts of Canada, and the event numbers in the ICES-IOC Harmful Algal Event Database ([HAEDAT](#)). Values are for blue mussel (*Mytilus edulis*) whole tissue, unless otherwise indicated. NR = value not reported.

Date	Location	Max toxin ($\mu\text{g g}^{-1}$)	HAEDAT	Reference
August 2, 1990	Mahone Bay, NS	1.0	CA-90-006	Quilliam et al. 1993
June 1991	Mahone Bay, NS	NR	CA-91-003	Watson-Wright et al. 1992a
August – September 1992	Mahone Bay, NS	0.45	CA-92-004	Gilgan et al. 1995

Date	Location	Max toxin ($\mu\text{g g}^{-1}$)	HAEDAT	Reference
July – October 1993	Mahone Bay, NS	0.48	–	Gilgan et al. 1994b
October 27, 1993	Bonavista Bay, NL	4.0 ¹	CA-93-004	Gilgan et al. 1994a
January – August 8, 1994	Bonavista Bay, NL	0.16	CA-94-001	DFO Closure Orders
August 1994	Mahone Bay, NS	NR	–	Gilgan pers. comm.
July 1998	Magdalen Islands, QC	0.4	CA-98-002	Couture et al. 2001
July – August 1999	Magdalen Islands, QC	2.5 ²	CA-99-002	–
July 2000	Gaspé Bay, QC	0.93	CA-00-002	–
July 18 – August 25, 2000	Ship Harbour, NS	0.76	CA-00-024	–
July 24, 2000	Mahone Bay, NS	0.70	CA-00-025	–
April 29, 2001	Magdalen Islands, QC	0.62	CA-01-011	–
May – June, 2001	Notre Dame Bay, NL	NR	CA-01-009	–
June 5, 2001	Gaspé Bay, QC	0.89	CA-01-012	–
June 9, 2001	Tête-à-la-Baleine, QC	0.48	CA-01-007	–
June 9, 2001	Île Brûlette est, QC	0.40	CA-01-013	–
July 9, 2001	Gaspé Bay, QC	0.29	CA-01-014	–
July 16, 2001	Mahone Bay, NS	1.2	CA-01-029	–
July 2, 2002	Gaspé Bay, QC	0.36	CA-02-014	–
July 2, 2003	Chaleur Bay, QC	0.25	CA-03-029	–
July 7, 2003	Magdalen Islands, QC	0.71	CA-03-030	–
July 23, 2003	Ship Harbour, NS	0.64	CA-03-046	–
September 21, 2003	Baie au Saumon, QC	1.80	CA-03-031	–
October 14, 2003	Penouille, QC	5.20	CA-03-028	–
September 12, 2004	Tête-à-la-Baleine, QC	0.25	CA-04-040	–
September 18, 2005	Tête-à-la-Baleine, QC	0.29	CA-05-023	–
June 7, 2011	Rose Bay, Lunenburg Bay, NS	NR	CA-11-020	DFO news release
November 8, 2011	Spring Cove, BC	0.28 ³	CA-11-019	–
September 11, 2012	Effingham Inlet, BC	0.20 ⁴	CA-12-013	–
October 2, 2012	Chaleur Bay, QC	0.22	CA-12-023	–
January 30, 2013	Aimai Inlet, BC	0.37 ⁴	CA-13-019	–
May 21, 2013	Effingham Inlet, BC	0.42 ⁴	CA-13-020	–
June 25, 2013	Magdalen Islands, QC	0.25 ²	CA-13-025	–
July 2, 2013	Chaleur Bay, QC	0.23 ²	CA-13-026	–
October 15, 2013	Lama Passage, BC	0.52 ⁴	CA-13-021	–
December 4, 2013	Amai Inlet, BC	0.24 ⁴	CA-13-029	–
July 8, 2014	Chaleur Bay, QC	0.37 ²	CA-14-013	–
July 12, 2014	Ballet Bay, BC	0.22	CA-14-022	–
August 9, 2014	Magdalen Islands, QC	0.22	CA-14-014	–
September 15, 2014	Chamiss Bay, BC	0.40 ⁴	CA-14-020	–
September 21, 2014	Effingham Inlet, BC	0.20 ⁴	CA-14-021	–
June 29, 2015	Croaker Island, BC	0.18 ⁵	CA-15-014	–
January 4 – June 20, 2016	Croaker Island, BC	0.26 ⁵	CA-16-021	ICES 2017

Date	Location	Max toxin ($\mu\text{g g}^{-1}$)	HAEDAT	Reference
June 7–21, 2016	Toquart Bay, BC	0.33 ³	CA-16-022	ICES 2017
June 22, 2017	Richibucto Harbour, NB	0.12	CA-17-006	–
June 22, 2017	Shediac Bay, NB	0.13	CA-17-006	–
July 6, 2017	Mahone Bay, NS	0.21	CA-17-020	–
July 27, 2017	Ship Harbour, NS	0.21	CA-17-019	–
June 21, 2018	Boughton River, PE	0.32	CA-18-002	–
June 24, 2018	Jervis Inlet, BC	0.12 ⁵	–	–
July 16, 2018	Saanich Inlet, BC	0.12 ⁵	CA-18-001	–
August 27, 2018	Sober Island, NS	2.40	–	–

¹ Blue mussel (*Mytilus edulis*) digestive gland (= 0.8 $\mu\text{g g}^{-1}$ DTX1 in whole tissue)

² Sea scallop (*Placopecten magellanicus*) whole tissue

³ Mussels (*Mytilus* spp.)

⁴ California mussel (*Mytilus californianus*)

⁵ Bay mussels (*Mytilus trossulus*)

Although OA group toxins had already been detected in Canadian waters (from the lower St. Lawrence Estuary and PE; see above), DSP was not officially acknowledged as a problem until early August 1990. Then, the first confirmed case of DSP in North America occurred when 13 people became ill after consuming cultured blue mussels (*M. edulis*) from Indian Point (Mahone Bay, NS) ([Fig. 5B](#)) (Quilliam et al. 1993; HAEDAT [CA-90-006](#); [Table 13](#)). The toxin reported was DTX1, as measured by HPLC analysis and confirmed by LC–MS; no OA was found. Remnants of *D. norvegica* were found in the digestive glands of toxic mussels, but water samples were not available from this site at the time. A month later (September), a bloom of *Dinophysis* (predominantly *D. norvegica*; $\sim 1.8 \times 10^4$ cells L^{-1}) was found but the samples proved nontoxic (Watson-Wright et al. 1993a). The following June (1991), another *D. norvegica* bloom occurred in Mahone Bay. At that time both OA and DTX1 (at a 50:50 ratio), as determined by HPLC analysis, were found in the mussels (Watson-Wright et al. 1992a; HAEDAT [CA-91-003](#)). However, the peak toxicity in the mussels, as determined by mouse bioassay, did not occur until three to four weeks after the concentration of *Dinophysis* had peaked at only 2300 cells L^{-1} . This led to doubt as to the source of the toxicity. The low levels of OA group toxins found did not justify an official closure of mussel harvesting, but the mussel growers in the area voluntarily stopped harvesting for about a two-month period as a precautionary measure. In late September–November (1991), another bloom, composed almost uniquely of *D. norvegica* (1×10^4 cells L^{-1}), exhibited no detectable OA group toxins using ADAM/HPLC analysis (HAEDAT [CA-91-004](#)). In 1992, low levels of DTX1 were again detected, using an immunological test kit and HPLC analysis, in mussels from Indian Point, from late July to mid-October (Gilgan et al. 1995). The highest level ($\sim 0.45 \mu\text{g g}^{-1}$ DTX1 in digestive gland) was during the last week in August, as confirmed by LC–MS, and became undetectable by late September (HAEDAT [CA-92-004](#)). The source organism was not determined, and no toxins were detected in phytoplankton samples. This event resulted in the first official closure in Canada due to the presence of OA group toxins. Similar concentrations of DTX1 were again detected in Mahone Bay during July to October 1993 (Gilgan et al. 1994b), and low levels were found in August 1994 (M.W. Gilgan, DFO, Halifax, NS, 1994 pers. comm.).

Further information about closures due to OA group toxins at Indian Point (Mahone Bay) is available because of routine monitoring at an aquaculture site at that location. Another closure occurred on July 10, 2000, when DTX1 was detected in mussels (*M. edulis*) at $0.58 \mu\text{g g}^{-1}$, with a peak of $0.70 \mu\text{g g}^{-1}$ on July 24 (HAEDAT [CA-00-025](#)). The following year, on August 13, 2001, the peak in mussels was $1.2 \mu\text{g g}^{-1}$ DTX1 (HAEDAT [CA-01-029](#)). Later, on July 6, 2017, the finding of $0.21 \mu\text{g g}^{-1}$ OA equivalent (OA_{eq}) in mussels (*M. edulis*) initiated another closure at Indian Point (HAEDAT [CA-17-020](#)). The peak level in mussels was $0.43 \mu\text{g g}^{-1}$ (July 19) and in scallops (*Placopecten magellanicus*) it was $0.11 \mu\text{g g}^{-1}$ (June 28); scallops never exceeded the action limit. At nearby Snake Island, OA equivalents in scallops peaked at $0.33 \mu\text{g g}^{-1}$ on July 27, 2017. Levels remained elevated at Indian Point. For example, mussels contained $0.29 \mu\text{g g}^{-1}$ on September 27, 2017. Previously, on June 7, 2011, portions of Rose Bay and Lunenburg Bay (also on the southwestern shore of NS) were closed because of unacceptable levels of OA group toxins ([DFO news release](#); HAEDAT [CA-11-020](#)).

Prorocentrum lima, a known producer of OA group toxins (Section [4.2](#)) in Japanese waters (Murakami et al. 1982), was found in association with mussels from the Indian Point (Mahone Bay) aquaculture site implicated in the 1992–1993 DSP episodes (Lawrence et al. 1998a, 2000). Filaments of the phaeophyte *Pilayella littoralis* grew on long lines that held the mussels. *Prorocentrum lima* cells were found growing epiphytically on these fouling filaments (Cembella et al. 1997), and single-cell isolates brought into culture produced OA, DTX1 and DTX4, as confirmed by both LC–MS and specific antibody probes. These results supported earlier reports of toxin production by *P. lima* from that site (Marr et al. 1992a,b; Jackson et al. 1993; McLachlan et al. 1994; Pan et al. 1999). The presence of *P. lima* cells, which may also be found in benthic sediments, may go undetected in plankton samples typically collected in the water column. It has been hypothesized that storms and/or currents may resuspend toxic cells from the benthos and/or dislodge epiphytically growing cells, making them available for filter feeding by the cultured mussels. At the time of sampling, *Dinophysis* cf. *norvegica* was present in the water column at $2800 \text{ cells L}^{-1}$, but LC–MS analysis detected no toxins in the plankton collected at that site (Lawrence et al. 1998a).

On the eastern shore of NS, at Ship Harbour, low levels of DTX1 were first reported in cultured mussels (*M. edulis*) in August 1994 (M.W. Gilgan, DFO, Halifax, NS, 1994 pers. comm). Then, between July 18 and August 25, 2000, total OA toxins in mussels (*M. edulis*) from Ship Harbour were $0.22\text{--}0.76 \mu\text{g g}^{-1}$ whole tissue (HAEDAT [CA-00-024](#); [Table 13](#)). On July 23, 2003, $0.64 \mu\text{g g}^{-1}$ of DTX1 was then found in mussels (*M. edulis*) (HAEDAT [CA-03-046](#)). Later, total OA toxins in mussels (*M. edulis*) were at $0.21 \mu\text{g g}^{-1}$ on July 27, 2017 (HAEDAT [CA-17-019](#)). The following year, DTX1/DTX1 esters were detected at $0.26 \mu\text{g g}^{-1}$ in mussels (*M. edulis*) between July 23 and 27, 2018 (HAEDAT [CA-18-003](#)), and product was withheld from the market. Harvesting was officially closed on August 10 (Prohibition Order [MTN-2018-020](#)), and by August 10–13, levels had increased to $2.6 \mu\text{g g}^{-1}$ (CFIA data). This area was opened on September 25, 2018 (Prohibition Order [REV-MTN-2018-020](#)).

A precautionary closure occurred at nearby Sober Island, based on whole mussel values of $0.16 \mu\text{g g}^{-1}$ DTX1/DTX1 esters on August 8, 2018, and $0.18 \mu\text{g g}^{-1}$ DTX1/DTX1 esters on August 13, 2018. An official closure occurred on August 20, 2018, when the value increased to $0.44 \mu\text{g g}^{-1}$ DTX1/DTX1. The value peaked at $2.4 \mu\text{g g}^{-1}$ DTX1/DTX1 esters on August 27, 2018 (CFIA data).

It is not clear what the biological source of the OA group toxins is in Ship Harbour (NS). However, juvenile bay scallops (*Argopecten irradians*) taken from a hatchery in Ship Harbour rapidly ingested toxic *P. lima* cells in controlled laboratory microcosms, with apparently no adverse physiological effects; the OA group toxin concentration in the viscera exceeded the regulatory limit within one day (Bauder et al. 1996, 2001). However, because most of the OA group toxins are found in the digestive gland, they are also depurated rapidly. A concern is that *P. lima* cells are able to survive passage through the gut of these scallops (Bauder and Cembella 2000). This means that viable cells may be spread to other sites when fecal pellets are released during depuration or when the scallops are transferred to different locations.

Prorocentrum lima was also found in substantial numbers in the water column and attached to vegetation at aquaculture sites in the Miramichi Estuary (NB) ([Fig. 5B](#)) (J.C. Smith, DFO, Moncton, NB, 1995 pers. comm.). Isolates of *P. lima* from the Miramichi Estuary produced OA and DTX1 in culture (J.C. Smith, DFO, Moncton, NB, 1995 pers. comm.). Although the toxin source was not identified, the first report of OA group toxins in NB was on June 22, 2017, when $0.12 \mu\text{g g}^{-1}$ and $0.13 \mu\text{g g}^{-1}$ DTX1 were found in blue mussels (*M. edulis*) from Richibucto Harbour and Shediac Bay, respectively (HAEDAT [CA-17-006](#); [Table 13](#)). Richibucto Harbour was closed on June 22 (Prohibition Order [GTN-2017-001](#)) and opened on July 17, 2017 (Prohibition Order [GTN-2017-004](#)). Shediac Bay was closed June 30 (Prohibition Order [GTN-2017-002](#)) and opened July 7, 2017 (Prohibition Order [GTN-2017-003](#)).

4.4.2 Okadaic acid group toxins in the Bay of Fundy

There have been no closures due to OA group toxins in the Bay of Fundy. However, trace levels of DTX1, as determined by HPLC, were found in mussels (*M. edulis*) on September 14, 1992, at Beaver Harbour on the southwest NB shore (HAEDAT [CA-92-003](#)) ([Fig. 5B](#)). Unusually high numbers of *Dinophysis acuminata* ($7000 \text{ cells L}^{-1}$) were present at the same time, but were not tested for toxins. Other locations include Passamaquoddy Bay (Bay Road), Bocabec Cove, Cheney's Passage (Grand Manan Island), Hills Island, Lepreau Basin, Letete Passage, Red Head Harbour, Simmons Cove, and Stuart Town. On the NS shore of the Bay of Fundy, trace levels were also found at Smith's Cove and in St. Mary's Bay.

Other evidence of OA group toxins in the Bay of Fundy comes from laboratory isolates. Once it was learned that *Dinophysis* species could be grown in culture when fed with prey species (Section [4.2](#)), Tong et al. (2015a) were able to compare the morphology, phylogeny, growth, toxin production, and toxin composition of three isolates from the northeast U.S./Canada region: Eel Pond and Martha's Vineyard (Massachusetts) and Blacks Harbour (Bay of Fundy). Despite observed phenotypic heterogeneity, morphometrics and molecular evidence classified the three northwestern Atlantic isolates as *D. acuminata*. This is in agreement with the observation of this species in the Bay of Fundy in September 1992 (see above paragraph). The Bay of Fundy isolate contained low amounts of DTX1 ($0.26\text{--}0.34 \text{ pg cell}^{-1}$), and no OA was detected. Pectenotoxin-2 (PTX2; Section [5.1](#)) was the dominant toxin (98%; $13.0\text{--}21.8 \text{ pg cell}^{-1}$), in contrast to the other two isolates. This is the first time that a *Dinophysis* species from Canadian waters has been shown to produce OA group toxins. Each of these isolates produced less toxin than those from other parts of the world, and the authors concluded that the risk of significant OA group toxin outbreaks is low to moderate in this region.

Another potential source of OA group toxins, the dinoflagellate *Prorocentrum lima* (Section 4.2), has not been reported in the Bay of Fundy, perhaps because benthic and or/epiphytic sampling is rarely carried out there. Indeed, epiphytic *P. lima* cells were found to contaminate mussels (*M. edulis*) with DTX1 in Bar Harbor, Maine (Morton et al. 1999).

4.4.3 Okadaic acid group toxins in the Gulf of St. Lawrence

In July 1989, OA was reported for the first time in North America in natural phytoplankton assemblages from the lower St. Lawrence Estuary and along the Gaspé coast (Fig. 5B) (Cembella 1989). The toxin, detected with an immunological test kit and confirmed by HPLC, was associated only with samples in which the dinoflagellates *Dinophysis norvegica* and *D. acuminata* were prominent; blooms of the related *D. rotundata* (= *Phalacroma rotundatum*) exhibited no trace of OA group toxins.

In July 1998, 20 people exhibited DSP-like symptoms (diarrhea, vomiting) after consuming cultured blue mussels (*M. edulis*) from the Magdalen Islands, in the Gulf of St. Lawrence. Analyses revealed that the mussels contained DTX1 (but no OA), at a low concentration ($0.1 \mu\text{g g}^{-1}$ digestive gland, below the regulatory limit set in 1994 of $1.0 \mu\text{g g}^{-1}$ digestive gland; the limit is now $0.2 \mu\text{g g}^{-1}$ whole tissue) (Levasseur et al. 2003; van de Riet et al. 2006; HAEDAT CA-98-002; Table 13). This prompted a study to determine the source of DTX1 in the Magdalen Islands, with a focus on *Prorocentrum lima*, which had previously been detected in the water column of the lagoons (Bérard-Therriault et al. 1999). During the summer of 1999, *P. lima* was found growing on epibionts, including the rhodophyte *Trailliella intricata* and the filamentous brown alga *Ectocarpus* sp. It was also present in low numbers in the water column ($<1 \text{ cell L}^{-1}$), as well as in mussel digestive glands (Levasseur et al. 2001, 2003). Interestingly, *P. mexicanum* was also present in their samples; this was the first time that this species was formally identified in eastern Canada. At Havre-aux-Maisons lagoon, cells were in lower abundance on the epibionts, but higher in the water column (but only a maximum of $\sim 10 \text{ cells L}^{-1}$). Similar results were found at Grande Entrée lagoon, 30 km to the northeast. Although empty thecae of *P. lima* and *P. mexicanum* were found in the digestive glands of mussels, Levasseur et al. (2003) report that OA and DTX toxins were below the detection limit of the HPLC-fluorescence method of van de Riet et al. (1995). However, Levasseur et al. (2001) indicate the presence of $0.4 \mu\text{g g}^{-1}$ DTX1 in the digestive gland of mussels in the Havre-aux-Maisons lagoon of the Magdalen Islands, on June 29, 1999. A value of $2.5 \mu\text{g g}^{-1}$ DTX1 (determined by the mouse bioassay) was reported in sea scallops (*Placopecten magellanicus*) from the Magdalen Islands during July–August 1999 (HAEDAT CA-99-002). On July 7, 2003, OA group toxins ($0.71 \mu\text{g g}^{-1}$) were detected in the Havre-aux-Maisons lagoon (HAEDAT CA-03-030). Toxins were present from June 2 to October 14, 2003. Also in the Havre-aux-Maisons lagoon, DTX1 esters ($0.25 \mu\text{g g}^{-1}$) were detected in scallops on June 25, 2013 (HAEDAT CA-13-025). In the Baie de Plaisance (southern Magdalen Islands), mussels (*M. edulis*) contained $0.22 \mu\text{g g}^{-1}$ DTX1 and $0.06 \mu\text{g g}^{-1}$ DTX1 esters, on August 9, 2014 (HAEDAT CA-14-014).

It should be noted that *P. mexicanum* has been confused with *P. rhathymum* in the past, but the two are distinct species (see Cortés-Altamirano and Sierra-Beltrán 2003). Only the latter has been shown to produce OA (An et al. 2010). Furthermore, in all cases where toxicity has been demonstrated this has been due to *P. rhathymum* (IOC-UNESCO Taxonomic Reference List of Harmful Micro Algae), not *P. mexicanum*. During 1994–2008, *P. lima* was present only sporadically in the water

column in the Gulf of St. Lawrence, and only at extremely low abundances (near the detection level of 20 cells L⁻¹), because it is a benthic or epiphytic species (Lessard et al. 2020).

OA group toxins have also been reported in Gaspé Bay. In July 2000, 0.93 µg g⁻¹ DTX1 was found in mussels (*M. edulis*); *P. lima* was also present (HAEDAT [CA-00-002](#)). Further detection of DTX1 in mussels was on June 5, 2001 (0.89 µg g⁻¹; HAEDAT [CA-01-012](#)) and July 9, 2001 (0.29 µg g⁻¹; HAEDAT [CA-01-014](#)). Then, on July 2, 2002, OA was detected in shellfish from Penouille (Gaspé Bay) at 0.36 µg g⁻¹, which caused a closure that lasted until October 7, 2002 (HAEDAT [CA-02-014](#)). *Dinophysis acuminata* was present at 2000 cells L⁻¹. On October 14, 2003, OA group toxins were reported in mussels and scallops at a level of 5.20 µg g⁻¹, at Penouille, where *D. acuminata* was present at 1300 cells L⁻¹ (HAEDAT [CA-03-028](#)). Lessard et al. (2020) describe the abundances of *D. acuminata*, *D. norvegica* and *D. rotundata* (= *Phalacrocoma rotundatum*), and their distribution in relation to temperature and salinity in the Gulf of St. Lawrence during 1994–2008. *Dinophysis acuminata* was most commonly observed, at low concentrations (≥ 20 cells L⁻¹).

At Carleton, on the Chaleur Bay (QC), 0.25 µg g⁻¹ of OA group toxins was detected on July 2, 2003 (HAEDAT [CA-03-029](#)). *Dinophysis acuminata* was present at 920 cells L⁻¹, although *D. acuta* was also present. Also in Chaleur Bay, mussels (*M. edulis*) contained 0.22 µg g⁻¹ DTX1, 0.09 µg g⁻¹ DTX1 esters, and 0.04 µg g⁻¹ OA esters on October 2, 2012 (HAEDAT [CA-12-023](#)). Scallops (*Placopecten magellanicus*) from Newport Point, north of Chaleur Bay, contained 0.23 µg g⁻¹ DTX1 esters on July 2, 2013 (HAEDAT [CA-13-026](#)). The following year, July 8, 2014, 0.37 µg g⁻¹ DTX1 esters were again detected in scallops from that location (HAEDAT [CA-14-013](#)).

On the north shore of the Gulf of St. Lawrence, DTX1 was detected in mussels (*M. edulis*): 0.48 µg g⁻¹ near Tête-à-la-Baleine (HAEDAT [CA-01-007](#)) and 0.40 µg g⁻¹ at Île Brulette (HAEDAT [CA-01-013](#)), both on June 9, 2001. On September 21, 2003, OA group toxins (1.80 µg g⁻¹) were detected at Baie au Saumon (HAEDAT [CA-03-031](#)). *Dinophysis acuminata* was present at 700 cells L⁻¹, although *D. acuta* was also present. On September 12, 2004, OA group toxins (0.25 µg g⁻¹) were found in wild mussels (*M. edulis*) near Tête-à-la-Baleine (HAEDAT [CA-04-040](#)). A year later, on September 18, 2005, scallops from the same location (Baie au Saumon) contained 0.29 µg g⁻¹ DTX1 (HAEDAT [CA-05-023](#)).

4.4.4 Okadaic acid group toxins in Newfoundland and Labrador

In Bonavista Bay (Alexander Bay and Magics Arm) ([Fig. 5B](#)), high levels of DTX1 (4.0 µg g⁻¹ digestive gland; 0.16 µg g⁻¹ whole tissue), but no OA, were found for the first time in blue mussels (*M. edulis*) in late October 1993, as detected by HPLC analysis and confirmed by LC–MS (Gilgan et al. 1994a; van de Riet et al. 2006; HAEDAT [CA-93-004](#); [Table 13](#)). Many of the embayments in the vicinity were contaminated to variable amounts by the toxin. Several persons developed symptoms of what appeared to be DSP after consuming mussels from that area (McKenzie et al. 1994). This resulted in the closure of molluscan shellfish harvesting for the first time ever in Bonavista Bay due to OA group toxins, from October 1993 (HAEDAT [CA-93-004](#)) to August 1994 (HAEDAT [CA-94-001](#)). *Dinophysis norvegica* was the dominant species in the water column (2000 cells L⁻¹) and large numbers (>40,000 cells per mussel) were also present in the gut contents of the mussels during the Bonavista Bay incident, as were *D. acuminata* and *Prorocentrum lima* (C.H. McKenzie unpubl. data). This was the second occurrence of DSP in North America, after the 1990 Mahone Bay (NS) incident (Section [4.4.1](#)).

On October 27, 1993, other mussel samples also contained OA group toxins at: Northwest Arm, Alexander Bay ($1.5\text{--}2.9\ \mu\text{g g}^{-1}$ in wild mussels); Beach Cove, Traytown ($0.09\ \mu\text{g g}^{-1}$ in cultured mussels); and Magic's Arm, Bonavista Bay ($2.4\ \mu\text{g g}^{-1}$ in cultured mussels). Low levels of DTX1 have since been found in bays of northern and southeast NL. For example, in May 2001, mussels collected at an aquaculture site in Fortune Harbour (Notre Dame Bay) contained OA group toxins and were rejected from EU markets; the site was closed from May to June (McKenzie 2006; HAEDAT [CA-01-009](#)). Although the problem did not recur in subsequent shipments, high costs, including shipping, were accrued by the producer and the CFIA now routinely tests for OA group toxins as part of its monitoring program. The probable source of the toxins in the mussels was determined to be *P. lima* growing epiphytically on the mussel ropes (McKenzie 2006).

4.5 Okadaic acid group toxins on the Canadian Pacific coast

Until 2011, there were no reports of DSP on the Canadian Pacific coast, although the presence of several species of *Dinophysis* had been documented previously (Haigh and Taylor 1990; Taylor et al. 1991; Taylor 1993). For example, in Sechart Inlet (BC) ([Fig. 5A](#)), *D. acuminata* peaked in midsummer, and *D. fortii* peaked later in the fall; *D. norvegica* was also present (Taylor et al. 1994). No OA or DTX1 was detected in oysters (*Crassostrea gigas*), blue mussels (*M. edulis*) or Manila clams (*Venerupis philippinarum*) from Storm Bay (Sechart Inlet), in June 1995, at a time when concentrations of *Dinophysis* approached $20\ \text{cells L}^{-1}$ in the nearby waters at 3–4 m depth (J.R. Forbes, M. Ikonou and R.A. Addison, DFO unpubl. data).

Then, between July 28 and August 6, 2011, 62 people became ill after consuming mussels, all reporting symptoms characteristic of DSP (Taylor et al. 2013). The incubation period ranged from 5 to 15 h and symptoms lasted 1 to 3 days. The mussels (*Mytilus galloprovincialis*, *M. edulis*) were harvested from Gorge Harbour (Cortes Island, Upper Strait of Georgia) ([Fig. 5A](#)) (Haigh 2012; Esenkulova and Haigh 2012). The CFIA issued a harvest closure recommendation on August 5, 2011, a health hazard alert on August 6 (CFIA 2011), and recalled affected product. The CFIA determined that one sample contained $0.01\ \mu\text{g g}^{-1}$ OA, $0.23\ \mu\text{g g}^{-1}$ DTX1, and $0.14\ \mu\text{g g}^{-1}$ DTX1 esters; another sample contained $0.72\ \mu\text{g g}^{-1}$ DTX1 esters (Taylor et al. 2013; McCarron et al. 2014a); no DTX2 was detected. Nevertheless, these two samples were above the action level of $0.2\ \mu\text{g g}^{-1}$ for the sum of OA and DTX1. Acyl esters of OA group toxins are of concern because they can be hydrolyzed to their parent molecule (OA or DTXs) in the gastrointestinal tract of humans (García et al. 2005). Because of this event, DTX2, as well as all OA and DTX esters, have been included in the provisional DSP action limit, which is currently under review by Health Canada. This first DSP outbreak in BC prompted a Canadian Diarrhetic Shellfish Poisoning Symposium (McIntyre 2013) to discuss the episode and plan for the future. The event led to an increase in the number of shellfish monitoring sites and in sampling frequency in BC (Taylor et al. 2013). Interestingly, other lipophilic toxins were also detected in these samples: significant amounts of YTXs (Section [5.2](#)) (McCarron et al. 2014a), and trace levels of SPX1 (Section [6.1](#)), PnTX-G (Section [6.2](#)), and GYM (Section [6.3](#)).

The above investigation did not have access to phytoplankton monitoring that could have determined the source organism(s) responsible for the OA group toxins. However, the BC [HAMP](#) was collecting phytoplankton samples for the salmonid aquaculture industry (Section [9.2.1](#)) at Conville Bay (Quadra Island), ~25 km away from the above affected shellfish site at Gorge Harbour (Haigh 2012; Esenkulova and Haigh 2012). The most abundant *Dinophysis* species in these samples, at the time of OA group toxins in Gorge Harbour mussels, was *D. acuminata* (82%), followed by *D.*

acuta (9%), *D. rotundata* (= *Phalacrocoma rotundatum*) (5%) and *D. fortii* (4%), giving a total of 2.4×10^4 *Dinophysis* cells L⁻¹ (HAEDAT [CA-11-002](#); [Table 13](#)). The toxins returned to acceptable levels in Gorge Harbour shellfish samples about two weeks after the highest *Dinophysis* concentrations were found ~25 km away, and the area was re-opened on August 24, 2011.

This event showed the high degree of cooperation between the CFIA (with support from NRC), Health Canada, the British Columbia Centre for Disease Control, the British Columbia Ministry of Health, the British Columbia Public Health Microbiology and Reference Laboratory, the Vancouver Coastal Health Authority, the Fraser Health Authority, and the School of Population and Public Health (University of British Columbia). Fortunately, the [HAMP](#) was sampling at the same time and in the general vicinity of this event, and could gather some information about the suspected source organisms, *Dinophysis* cells.

The above event was likely connected to the one that occurred on June 29, 2011, when three people became ill after ingesting mussels collected from Sequim Bay State Park, WA, on the Strait of Juan de Fuca, bordering BC (Lloyd et al. 2013; Trainer et al. 2015b). Levels of OA group toxins were 37.6–160.3 µg 100 g⁻¹ (0.38–1.6 µg g⁻¹) shellfish tissue. This prompted the [SoundToxins](#) program in Puget Sound and the Olympic Region Harmful Algal Bloom ([ORHAB](#)) partnership on the outer WA coast to monitor for OA group toxins and the source organisms in 2012. The primary toxin isomer in shellfish and plankton samples was DTX1, and *D. acuminata* was the primary *Dinophysis* species (Trainer et al. 2015b). In addition, the lipophilic toxins pectenotoxin-2 (PTX2) and yessotoxin (YTX) were measured in shellfish, and azaspiracid-2 (AZA2) was measured in phytoplankton. Elevated shellfish toxicity in May/June 2012 was correlated with above average flows of the Fraser River. The 2011 event was the first time that a common international toxic episode was demonstrated in WA and BC; the 2015 DA event (Section [3.3](#)) was a second such example.

On the lower west coast of Vancouver Island, DTX1 was detected at 0.28 µg g⁻¹ whole tissue in mussels (*Mytilus* sp.) at Spring Cove (near Ucluelet), on November 8, 2011 (HAEDAT [CA-11-019](#); [Table 13](#)). On September 11, 2012, DTX1 at 0.2 µg g⁻¹ whole tissue was detected in *Mytilus californianus* from Effingham Inlet (Barkley Sound; [Fig. 5A](#)) (HAEDAT [CA-12-013](#)). Just prior and after this, at that same location, DTX1 values were less: 0.03 µg g⁻¹ on September 4 and 0.08 µg g⁻¹ on September 18, 2012. The next year, on May 21, 2013, DTX1 was again detected, at 0.42 µg g⁻¹ in *M. californianus*, from Effingham Inlet (HAEDAT [CA-13-020](#)). The event lasted until May 28, 2013. The following year, on September 21, 2014, DTX1 was again detected in *M. californianus* from this location, at a concentration of 0.20 µg g⁻¹ (HAEDAT [CA-14-021](#)). Between June 7 and 21, 2016, 0.23–0.33 µg g⁻¹ OA_{eq} was detected in *Mytilus* sp. at Toquart Bay (Barkley Sound) (HAEDAT [CA-16-022](#)).

On the central coast of BC, DTX1 reached 0.52 µg g⁻¹ in *Mytilus californianus* from Lama Passage ([Fig. 5A](#)) on October 15, 2013 (HAEDAT [CA-13-021](#)). On the Sunshine Coast, DTX1 was detected in blue mussels (presumably *M. edulis*) from Ballet Bay, at a concentration of 0.22 µg g⁻¹, on July 12, 2014 (HAEDAT [CA-14-022](#)). On Croaker Island southeast, Greater Vancouver Area, 0.18 µg g⁻¹ DTX1 and 0.28 µg g⁻¹ DTX1 esters were detected on June 29, 2015, in blue mussels (presumably *M. edulis*) (HAEDAT [CA-15-014](#)). The following year, during January 4 to June 20, 2016, 0.20–0.26 µg g⁻¹ OA_{eq} was detected in blue mussels (HAEDAT [CA-16-021](#)).

On the upper west coast of Vancouver Island, $0.37 \mu\text{g g}^{-1}$ DTX1 in whole tissue was detected in *Mytilus californianus* in Amai Inlet (Kyuquot Sound) on January 30, 2013 (HAEDAT [CA-13-019](#)). Later that year, on December 4, 2013, $0.24 \mu\text{g g}^{-1}$ DTX1 was detected at the same location and in the same species, and the area remained closed (HAEDAT [CA-13-029](#)). On September 15, 2014, $0.40 \mu\text{g g}^{-1}$ DTX1 was detected in *M. californianus* at Chamiss Bay (HAEDAT [CA-14-020](#)).

4.6 Okadaic acid group toxins in the Canadian Arctic

Several potentially toxic dinoflagellates of the genera *Dinophysis*, as well as *Prorocentrum lima*, have been reported from the five Canadian Arctic regions ([Appendix 1](#)). *Dinophysis acuminata* has been reported in sea ice from the Canadian Archipelago, the Hudson Bay system and the Beaufort Sea (Hsiao et al. 1984; Woods and Smiley 1987; Róžańska et al. 2008, 2009), and in plankton from all Arctic regions (Grøntved and Seidenfaden 1938; Seidenfaden 1947; Bursa 1961a; Foy and Hsiao 1976; Hsiao 1976, 1983; Hsiao et al. 1977, 1984; Anderson et al. 1981; Hsiao and Pinkewycz 1983, 1985a,b; Percy et al. 1992; Simard et al. 1996; Harvey et al. 1997; Lovejoy et al. 2002; Melnikov et al. 2002; Róžańska et al. 2008; McLaughlin et al. 2009; Simo-Matchim et al. 2017; Dhifallah 2019; M. Poulin unpubl. data). *Dinophysis acuta* has been reported in plankton from the Canada Basin, the eastern Arctic and the Hudson Bay system (Grøntved and Seidenfaden 1938; Seidenfaden 1947; Bursa 1961a; MacLaren Marex 1979b; Anderson et al. 1981; Hsiao 1983; Percy et al. 1992; Simard et al. 1996; Melnikov et al. 2002; Simo-Matchim et al. 2017; Dhifallah 2019). *Dinophysis caudata* has been reported in both sea ice and plankton from the Hudson Bay system (Daugbjerg and Vørs 1994), whereas *D. fortii* has been only reported in plankton from the Hudson Bay (Anderson et al. 1981). *Dinophysis norvegica* has been reported only in phytoplankton from all five Arctic regions, except from the Beaufort Sea (Grøntved and Seidenfaden 1938; Seidenfaden 1947; Bursa 1961a,b; MacLaren Marex 1979b; Anderson et al. 1981; Hsiao 1983; Percy et al. 1992; Simard et al. 1996; Melnikov et al. 2002; Poulin et al. 2011; Simo-Matchim et al. 2017; Dhifallah 2019). *Prorocentrum lima* has been reported once in sea ice from the Beaufort Sea (Niemi et al. 2011) and in phytoplankton from the eastern and western Arctic, and the Hudson Bay system (Hsiao and Trucco 1980; Hsiao 1983; Roff and Legendre 1986; Percy et al. 1992; Niemi et al. 2011). However, they have not been tested for their ability to produce OA group toxins.

Phalacroma rotundatum has been reported in phytoplankton from the eastern Arctic, the Canadian Archipelago and the Hudson Bay system (Grøntved and Seidenfaden 1938; Seidenfaden 1947; Bursa 1961a; Anderson et al. 1981; Hsiao 1983; Percy et al. 1992; Simard et al. 1996; McLaughlin et al. 2009; Simo-Matchim et al. 2017; Dhifallah 2019) ([Appendix 1](#)).

4.7 *Dinophysis* spp. and the presence of okadaic acid group toxins

A working group was convened during the Third Canadian Workshop on Harmful Marine Algae (CWHMA), in Mont-Joli (QC) ([Table 5](#)), to discuss the paradox of high numbers of *Dinophysis* species that do not correlate with the presence of OA group toxins in eastern Canadian waters (Cembella et al. 1992). Indeed, in all of the above field studies, it was impossible to attribute the production of OA group toxins unequivocally to a *Dinophysis* species, as the evidence is only circumstantial. The presence of DTX1 in plankton net tow samples (collected in August 1989, in eastern PE), determined by a PPIA (Luu et al. 1993), was not confirmed by any other means and it has not been observed since. Although several *Dinophysis* and *Prorocentrum* species were found in water samples collected at about the same time, there was no apparent correlation between levels of DTX1 and any dinoflagellate species. As in the above Mahone Bay (NS) incident, other examples

illustrate the lack of association between the presence of *Dinophysis* species, reputed to be toxic elsewhere in the world, and the detection of OA group toxins in eastern Canada. In June and July 1990, large populations of *D. norvegica* and *D. acuminata* were observed in Cardigan River (PE) (Smith et al. 1991). However, net tow samples failed to reveal any OA group toxins by LC–MS. In 1991, large populations ($0.5\text{--}1.0 \times 10^4$ cells L^{-1}) of *D. norvegica*, *D. acuta* and *D. acuminata* were reported in St. Georges Bay (NS) and in the Miramichi Estuary (NB) (Fig. 5B) (Smith et al. 1994a). However, OA group toxins were not detected in these *Dinophysis* species when analyzed by HPLC, even after the plankton samples were enriched with nutrients (J.C. Smith, DFO, Moncton, NB, 1994 pers. comm.). Finally, in autumn 1994, over 10,000 L of *D. norvegica*-rich seawater from both Mahone Bay and the southeastern shore of NS were concentrated to yield up to 90 g of algal paste. None of these concentrated samples contained any OA group toxins, as analyzed by LC–MS (D.J. Douglas, NRC, Halifax, NS, 1994 pers. comm.). One caveat, however, is that vigorous pumping may have caused leakage of any toxins from the cells. The closest evidence of a link between *Dinophysis* and OA group toxins in Canadian waters is the detection of trace levels of DTX1 in extracts of 25 hand-picked *D. acuminata* cells from Ship Harbour (NS), as analyzed by a micro-sampling and extraction procedure coupled with micro-column LC–MS/MS (Quilliam et al. 2008).

Interestingly, some of the above discrepancies about the lack of toxicity, and of OA group toxins, in the presence of *D. norvegica* could be explained by recent findings from a bloom of this species in Maine (Deeds et al. 2018). A large monospecific bloom of *D. norvegica* lasted from July 5 to August 29, 2016, and covered much of the central portion of coastal Maine. Cell concentrations were >2000 cells L^{-1} , with a maximum of 5.4×10^4 cells L^{-1} on July 17. A commercial PPIA (manufactured by ZEULAB, Zaragoza, Spain) showed total OA group toxins $>0.16 \mu\text{g g}^{-1}$ OA_{eq} in shellfish, and harvesting was closed on July 20. However, testing by LC–MS/MS confirmed trace concentrations of DTX1 only, with no OA or DTX2 detected. Therefore, the harvesting closure was lifted on August 5, and the PPIA results were considered as false positives. Subsequent testing of shellfish collected during that event, as well as a filtered water sample collected during the bloom, continued to show strong PPIA activity, and results correlated to two different antibody-based commercial kits (a qualitative lateral flow kit and a quantitative ELISA). This suggested that the PPIA activity was due to the *D. norvegica* bloom and that “DSP-like toxins”, likely previously undescribed, were present. Work is in progress to confirm this hypothesis, and the identity of the responsible toxin as 14,15-dihydroDTX1. These findings will have important implications for the management of toxicity where *D. norvegica* occurs, but information about toxicity will need to be determined.

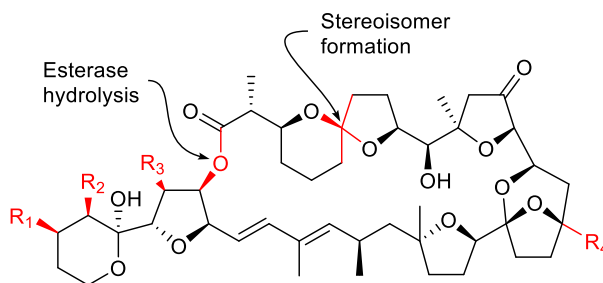
It is known from the physiological literature that some phycotoxins are only produced under certain environmental conditions or during specific phases of growth. As discussed above (Section 4.2), such studies were finally made possible for *Dinophysis* when it was discovered that a ciliate prey was required for its growth in culture (Park et al. 2006). Using this technique, Fux et al. (2011) grew strains of *D. acuminata*, including one isolated from Blacks Harbour (Bay of Fundy) in mixotrophic culture. They showed unambiguously, using LC–MS/MS, that cells contained DTX1 and PTX2. OA was detected in the medium, and they concluded that the OA was produced only in low quantities by *D. acuminata*, and that its presence was not the result of an enzymatic hydrolysis of the longer chained sulfated esters DTX4 and DTX5 (see Hu et al. 1995a,b). Also based on mixotrophic culture studies, Tong et al. (2011) concluded that the paradox of few DSP outbreaks in North American waters compared to Europe and Japan, in spite of the presence of toxigenic and potentially toxigenic *Dinophysis* species, could be explained by a combination of factors: the specific ability of certain

strains within a species to produce OA group toxins based on their genetic makeup and expression, and factors such as diet and environmental variables including temperature, light, salinity and nutrients. Still, more specific conditions must be found to explain this great difference.

5.0 Other Phycotoxins Regulated in Canada or Internationally

5.1 Pectenotoxin group

Because pectenotoxins (PTXs) ([Structure 4](#)) are lipophilic macrocyclic polyethers, they are extracted with OA group toxins from shellfish (Miles 2007). They are also toxic to mice when injected intraperitoneally, so were originally included in the “DSP toxin” group (FAO 2004). However, their mode of action is different; they are not diarrhetic but rather hepatotoxic, and are structurally different. They are therefore now classified together in their own group, the PTX group, on the basis of their chemical and toxicological properties (FAO/IOC/WHO 2005; Toyofuku 2006). It should be noted, however, that they are still grouped with OA group toxins in Chapter 11 (Control of Marine Biotoxins) of the 18/07/2014 version of the CSSP manual, with respect to its closure limit (CSSP 2019).



Structure 4. The structure of pectenotoxin and its analogues. The most common analogue PTX2 has the substituents R₁ = CH₃, R₂ = OH, R₃ = H and R₄ = CH₃.

The PTX group includes at least 15 analogues (EFSA 2009c), among them PTX1, PTX2, PTX3, PTX4, PTX6, and PTX11 (CSSP 2019). Shellfish metabolites include PTX2 seco acid and fatty acid acyl esters (Miles 2007). There are no data indicating adverse effects in humans associated with PTXs in shellfish (EFSA 2009b; Yasumoto et al. 2011). For example neither PTX2 nor PTX2 seco acid elicited signs of toxicity when dosed orally to mice at 5000 µg kg⁻¹ body weight (Miles et al. 2006). Thus, they may be of less concern to public health than was previously supposed. Note that PTX2 seco acid no longer triggers harvesting closures. However, harvesting in Canada is prohibited when PTX levels reach 0.2 µg g⁻¹ edible tissue or 1 µg g⁻¹ digestive tissue (Health Canada 2016; CSSP 2019). Health Canada has calculated the acute human intake of PTXs in Canada to be 0.61 µg kg⁻¹ body weight (Gully and Kuiper-Goodman 2004; Toyofuku 2006). Reviews of the chemistry, metabolism, chemical detection methods, pharmacology, toxicology, and risk assessment of PTXs are included in Munday (2008b), Suzuki (2008, 2014), Vilariño and Espiña (2008), Daneshian et al. (2013), Alfonso et al. (2014), and Farabegoli et al. (2018).

Monitoring of PTXs in Canada has changed over time. The first Canadian regulatory limit for PTXs was introduced by Health Canada in August 2004: $1.0 \mu\text{g g}^{-1}$ in digestive tissue. At that time, the digestive tissue of samples was analysed routinely to increase the sensitivity of the method, because many toxins are more highly concentrated in that tissue. After April 2007, however, results were generally based on whole tissue, because more sensitive instruments were used; the regulatory limit was then expressed as $0.20 \mu\text{g g}^{-1}$ for whole tissue and $1.0 \mu\text{g g}^{-1}$ for digestive tissue. Health Canada clarified the PTX regulatory limit in August 2009, by stating that PTX2 seco acid and 7-*epi*-PTX2 seco acid should both be included. This continued until June 2011, when the two PTX2 seco acid compounds were removed from the Canadian regulation, but other PTXs were added: PTX1, PTX3, PTX4, PTX6, and PTX11.

CFIA data show that PTXs are found at low levels (up to $0.26 \mu\text{g g}^{-1}$ total PTX in the whole tissue) at 119 sites on both coasts: 58 in BC ([Fig. 6A](#)), 9 in NB, 6 in NL, 21 in NS, 6 in PE, and 18 in QC ([Fig. 6B](#)). For example, PTXs were found for the first time in Canada in molluscan shellfish, in June 2000, from NS, NL, and the Magdalen Islands (QC) ([Fig. 6B](#); [Table 1](#)) at levels up to $0.18 \mu\text{g g}^{-1}$ whole tissue. Low levels of PTXs were reported on the south coast of NL in the spring of 2003. In QC, near Penouille (Gaspé Bay), the levels in blue mussels (*M. edulis*) were $0.06 \mu\text{g g}^{-1}$ for PTX2 (HAEDAT [CA-05-062](#)), $0.50 \mu\text{g g}^{-1}$ for PTX2 seco acid (HAEDAT [CA-05-063](#)), and $0.17 \mu\text{g g}^{-1}$ for 7-*epi*-PTX2 seco acid (HAEDAT [CA-05-064](#)) on July 18, 2005. Near Tête-à-la-Baleine (Lower North Shore), the level in sea scallops (*Placopecten magellanicus*) was $0.85 \mu\text{g g}^{-1}$ for PTX2 seco acid on July 22, 2002 (HAEDAT [CA-02-017](#)) and $0.07 \mu\text{g g}^{-1}$ for PTX2 on October 2, 2005 (HAEDAT [CA-05-059](#)).

On the south shore of NS (Ship Harbour and Lunenburg Bay), Solid Phase Adsorption Toxin Tracking (SPATT) bags (Section [9.1](#)), deployed from May to November 2005, detected PTX2 and PTX2 seco acid by LC–MS/MS (Garnett et al. 2006). A harvesting closure was triggered at Ship Harbour when PTX2 reached $0.24 \mu\text{g g}^{-1}$ in blue mussels (*M. edulis*) on November 8, 2010, and the area remained closed until mid-March 2011 (ICES 2011; HAEDAT [CA-10-018](#)). In August 2018, a sample of oysters (*Crassostrea virginica*) from Sober Island (eastern shore of NS) contained $0.26 \mu\text{g g}^{-1}$ PTX2, but the area was already closed on August 20, 2018, because of $0.44 \mu\text{g g}^{-1}$ DTX1/DTX1 esters in whole mussels (Section [4.4.1](#)).

Dinophysis acuminata was identified as the planktonic source of PTX2 that contaminated the mussels in NS (Anderson et al. 2001). Extracts obtained from a pool of 25 hand-picked *D. acuminata* cells from Ship Harbour (NS) showed the presence of PTX2 and PTX2 seco acid (Quilliam et al. 2008). Significantly, the ability to grow *Dinophysis* species in culture (Section [4.2](#)) has allowed the confirmation by LC–MS/MS of PTX2 production by *D. acuminata* from Blacks Harbour (Bay of Fundy) (Fux et al. 2011). Isolates of *D. acuminata* from elsewhere in the world had previously been shown to produce PTX2 (reviewed in Fux et al. 2011). Other evidence that *Dinophysis* species, including *D. acuta*, *D. caudata*, *D. fortii*, *D. norvegica*, and *D. rotundata* (= *Phalacroma rotundatum*), produce PTX2 is circumstantial and comes from the association of these species with the toxin in natural samples (Miles 2007). Some of these species (e.g. *D. acuminata*, *D. acuta*, *D. caudata*, *D. fortii*, *D. norvegica*, *D. rotundata* = *Phalacroma rotundatum*) are present in Canadian waters.

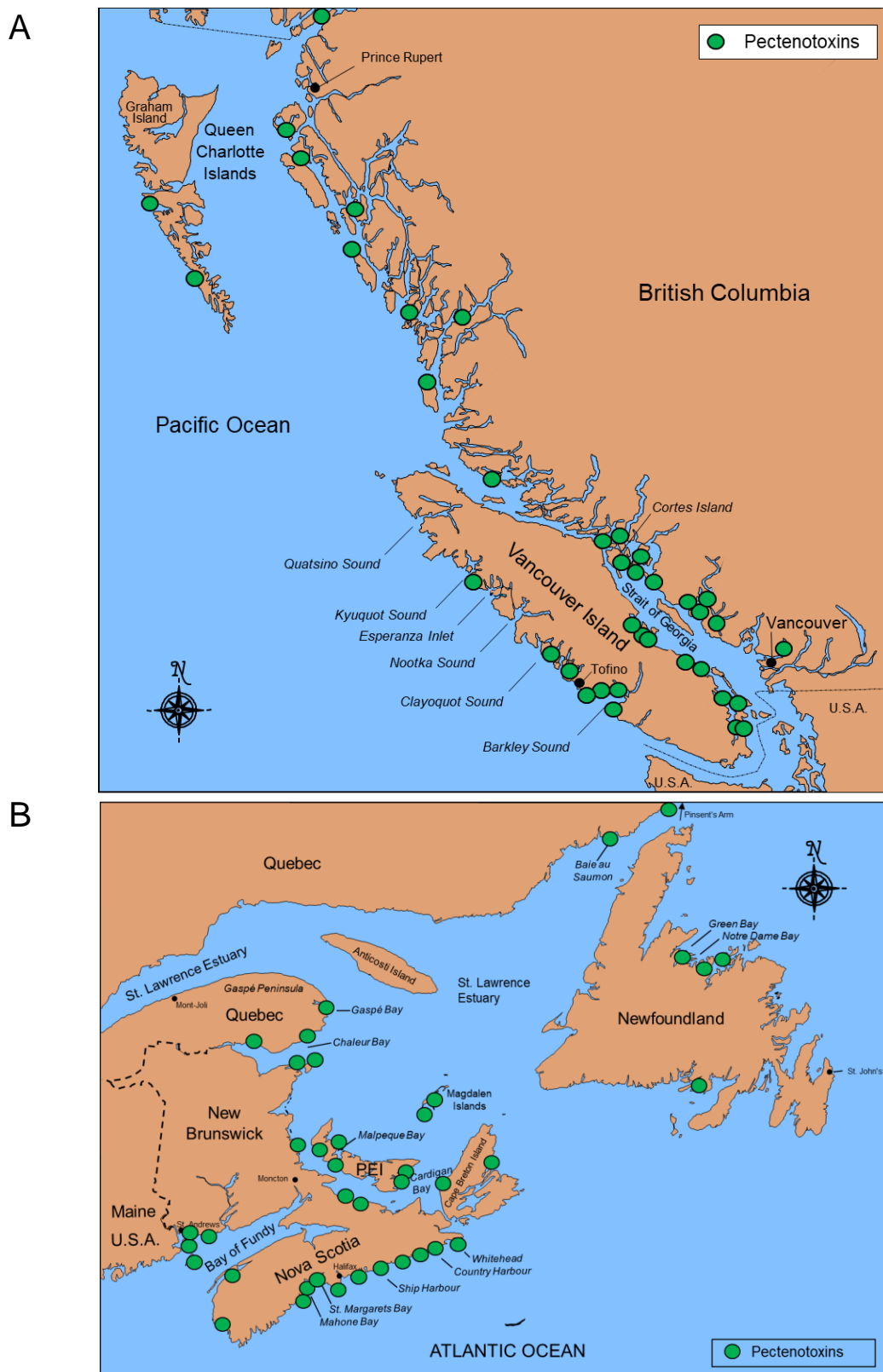


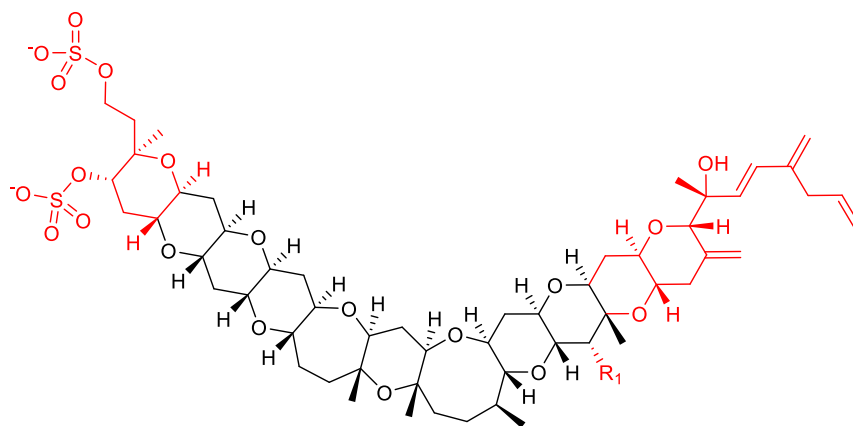
Figure 6. Location of pectenotoxins (up to 0.26 µg g⁻¹ total PTX) along the coast of A) British Columbia; B) Atlantic Canada. The regulatory level for PTXs is 0.2 µg g⁻¹.

In BC, other lipophilic toxins were also found during the first incident of DSP at Cortes Island ([Fig. 6A](#)) in August 2011 (Esenkulova and Haigh 2012; Taylor et al. 2013; [Section 4.5](#)). PTX2 and PTX2 seco acid were detected by LC–MS/MS in mussels (*Mytilus californianus*) (McCarron et al. 2014a). This is not surprising, as the bloom contained a number of *Dinophysis* species: *D. acuminata*, *D. acuta*, *D. fortii*, and *D. rotundata* (= *Phalacroma rotundatum*) (Esenkulova and Haigh 2012).

Analyses for PTXs in Canada are carried out by the CFIA laboratory in Dartmouth (NS), using the ultra performance liquid chromatography-tandem mass spectrometry (UPLC–MS/MS) method ([Appendix 3](#)). The Biotoxin Metrology group of the NRC, in Halifax (NS) (Quilliam 2006), produces a certified calibration solution and matrix reference material for the determination of PTX2 by chemical analysis methods ([NRC Certified Reference Materials](#); [Wikipedia](#); McCarron et al. 2017).

5.2 Yessotoxin group

The yessotoxins (YTXs) ([Structure 5](#)) were originally included with the “DSP toxins” (FAO 2004) largely because of their frequent association and co-extraction with the OA toxin group. However, because YTXs do not induce gastrointestinal symptoms, do not greatly inhibit protein phosphatase 2A (Ogino et al. 1997), and are different structurally, Quilliam (1999) proposed that they be classified as a separate group of toxins. The European authorities now consider them as such (European Commission 2002).



Structure 5. The structure of yessotoxin and its analogues. The most common analogue, YTX, has the substituent $R_1 = OH$. A number of structural variations can occur in the highlighted regions.

YTXs are lipophilic polycyclic ether compounds that were first isolated in 1986, in Mutsu Bay (Japan), from the digestive gland of the scallop *Patinopecten yessoensis*, from which the group got its name (reviewed in Paz et al. 2008; Tubaro et al. 2010; Paredes et al. 2011; Speijers et al. 2011; Rossini and Sala 2014). More than 90 YTXs have been observed (Miles et al. 2005), but only a few dozen have been fully identified (e.g. Finch et al. 2005). They seem to be heat stable in shellfish at temperatures relevant for cooking (EFSA 2008a).

The chemistry, mechanisms of action, metabolism, toxicology, detection, and mammalian cell lines affected by YTXs are reviewed in Toyofuku (2006), Alfonso and Alfonso (2008), Ciminiello and Fattorusso (2008), Munday et al. (2008), Tubaro et al. (2010), Suzuki (2014), and Alfonso et al. (2014, 2016). No human toxicity has been reported as a result of consumption of YTX-contaminated shellfish (Munday et al. 2008). Toxicological information is limited and comprises mostly studies on acute toxicity in mice, wherein oral toxicity is much less than with intraperitoneal injection, with a lethal dose to kill 50% of the test population (LD_{50}) of $286 \mu\text{g kg}^{-1}$ (Alfonso and Alfonso 2008; EFSA 2008a; Speijers et al. 2011). YTXs kill mouse T lymphocyte cells, rat cerebellar neurons, and human intestinal cells; as well, they fragment DNA and have antifungal activities (Ogino et al. 1997; Pérez-Gómez et al. 2006; Martín-López et al. 2012; Ferron et al. 2016). Damage to the cardiac system, but no death, has occurred in mice after repeated oral administration of YTX (Tubaro et al. 2008b). The precise mechanism of action is currently unknown, in spite of an increasing number of molecular studies (Paz et al. 2008), although some progress is being made (Alfonso et al. 2016). For example, a study reported subacute YTX-immunotoxicity in rats, suggesting that repeated exposures to low amounts of YTX might pose a threat to human health, especially in immuno-compromised populations (Ferreiro et al. 2017).

There are no reports of human poisoning induced by YTXs (Finch et al. 2005; Munday et al. 2008; Speijers et al. 2011; Martín-López et al. 2012), including in Canada (Rourke and Haigh 2019). Nevertheless, the positive mouse bioassay responses for lipophilic biotoxins (reported in Finch et al. 2005), along with effects on mammalian cells, raises concern about possible human health risks associated with chronic exposure to low amounts of YTXs (Pérez-Gómez et al. 2006). The EU originally established a maximum level of YTXs in the animals of $1 \text{ mg YTX}_{\text{eq}} \text{ kg}^{-1}$ ($= 1 \mu\text{g g}^{-1}$) of shellfish meat (European Commission 2002); this has now been increased to $3.75 \text{ mg YTX}_{\text{eq}} \text{ kg}^{-1}$ (European Commission 2013). Thus, in order for the CFIA to permit the export of product to EU countries, it now uses this same limit (CFIA 2015a). Note that this compares to $160 \mu\text{g kg}^{-1}$ for the other lipophilic toxins from the OA and PTX groups (Sections 4 and 5.1). There is currently no maximum limit for YTXs in CSSP (Speijers et al. 2011; CSSP 2018), although these toxins have been detected in Canadian waters (see below). In addition to concerns related to its toxicity, YTXs are also of interest because of their potential for pharmacological and therapeutic applications (reviewed in Alfonso et al. 2016). This includes the ability of YTXs to induce the death of some cell tumour lines, its immune-regulatory effect on lymphocyte cells, its improvement of Tau and β -amyloid levels related to Alzheimer's disease, and its effects on glucose and lipid metabolism.

The CFIA has monitored Canadian shellfish samples for YTXs since 2003, in support of export activities and to generate data for a Canadian risk assessment (Rourke and Haigh 2019). To date, YTXs have been found at levels up to $12 \mu\text{g g}^{-1}$ total YTX at 245 sites on both coasts of Canada: 164 in BC (Fig. 7A), 3 in NB, 46 in NL, 13 in NS, 8 in PE, and 11 in QC (Fig. 7B).

YTXs were found, for the first time in Canada, at eight aquaculture sites in eastern NL (e.g. Strong Island Tickle), in blue mussel (*M. edulis*) digestive glands, starting on February 24, 2004 (Table 1), and continuing to May 19, 2004, and again in early June 2004, in Notre Dame Bay and Green Bay (Fig. 7B). Concentrations of YTX ranged between 0.22 and $0.46 \mu\text{g g}^{-1}$ digestive gland (CFIA data), which was below the EU regulatory limit of $1 \mu\text{g g}^{-1}$ of whole animal (European Commission 2003; $\sim 5 \mu\text{g g}^{-1}$ digestive gland). The presence of YTXs (including 45-OHYTX and other YTXs) in these mussels was confirmed by LC-MS (Finch et al. 2005). Although this is the first time that YTXs were found in Canadian shellfish, it is also the first time that samples had

been analyzed for these compounds. Because the YTXs were found during the early spring, it is likely that they were retained in the mussels from a toxic bloom that occurred sometime during the previous year, rather than resulting from the spring bloom.

On September 30, 2005, the CFIA detected YTXs ($0.11 \mu\text{g g}^{-1}$) in blue mussels (*M. edulis*) from a natural harvesting site near Carleton (Chaleur Bay, QC; [Fig. 7B](#)) (HAEDAT [CA-05-080](#)). There were no closures, as there are no Canadian YTX action limits or regulations for product shipped within Canada. From May to November 2005, YTXs were detected in SPATT bags, deployed in Ship Harbour and Lunenburg Bay (NS; [Fig. 7B](#)), by LC–MS/MS (Garnett et al. 2006). YTXs were again detected at Ship Harbour in 2014, in blue mussels (*M. edulis*), starting mid-October ($0.51 \mu\text{g g}^{-1}$) and continuing until December 1 ($0.54 \mu\text{g g}^{-1}$), with a maximum concentration ($1.6 \mu\text{g g}^{-1}$) on November 3 (ICES 2015, 2017). Those toxin results were actually “total YTX”, which is the sum of the four YTX compounds that are regulated in the EU (YTX, homoYTX, 45-OH YTX, and 45-OH homoYTX).

YTXs were first detected in BC during 2004 ([Table 1](#)), when a sample of mussels (*M. californianus*) contained YTX and 45-OH YTX, with a total YTX concentration double the EU limit at the time (CFIA and NRC data). High levels of YTXs continued to be detected from BC shellfish samples. In August 2011, during the first incident of DSP in BC (Esenkulova and Haigh 2012; Taylor et al. 2013; Section [4.5](#)), significant levels of YTX and its metabolites were detected in mussels (*Mytilus californianus*) from Gorge Harbour (Cortes Island; [Fig. 7A](#)) (McCarron et al. 2014a). YTX was the dominant analog, at a concentration of $1.18 \mu\text{g g}^{-1}$, and a significant amount of 45-OHYTX ($0.42 \mu\text{g g}^{-1}$) was also measured.

In addition to the above, Rourke and Haigh (2019) described all the occurrences of YTXs in Canadian shellfish from 2012 to 2018. Mussels (*Mytilus* spp.) were most commonly sampled (73% of all samples and 94% of west coast samples). YTX levels in shellfish were higher and were detected in 65% of west coast samples. In contrast, only 2% of east coast samples had detectable levels of YTXs. During that time period, 3.8% of the samples would have exceeded the EU maximum limit; only one of these samples was from the east coast. The highest total YTX concentration detected was $12 \text{ mg YTX}_{\text{eq}} \text{ kg}^{-1}$ ($= 12 \mu\text{g g}^{-1}$), which is one of the highest shellfish contamination levels reported in the literature to date. YTX was the most dominant analogue, but 45-OHYTX was also detected. There was no relationship between temperature or salinity and the presence of YTXs. No known human illnesses were linked to high YTXs levels during the sampling period.

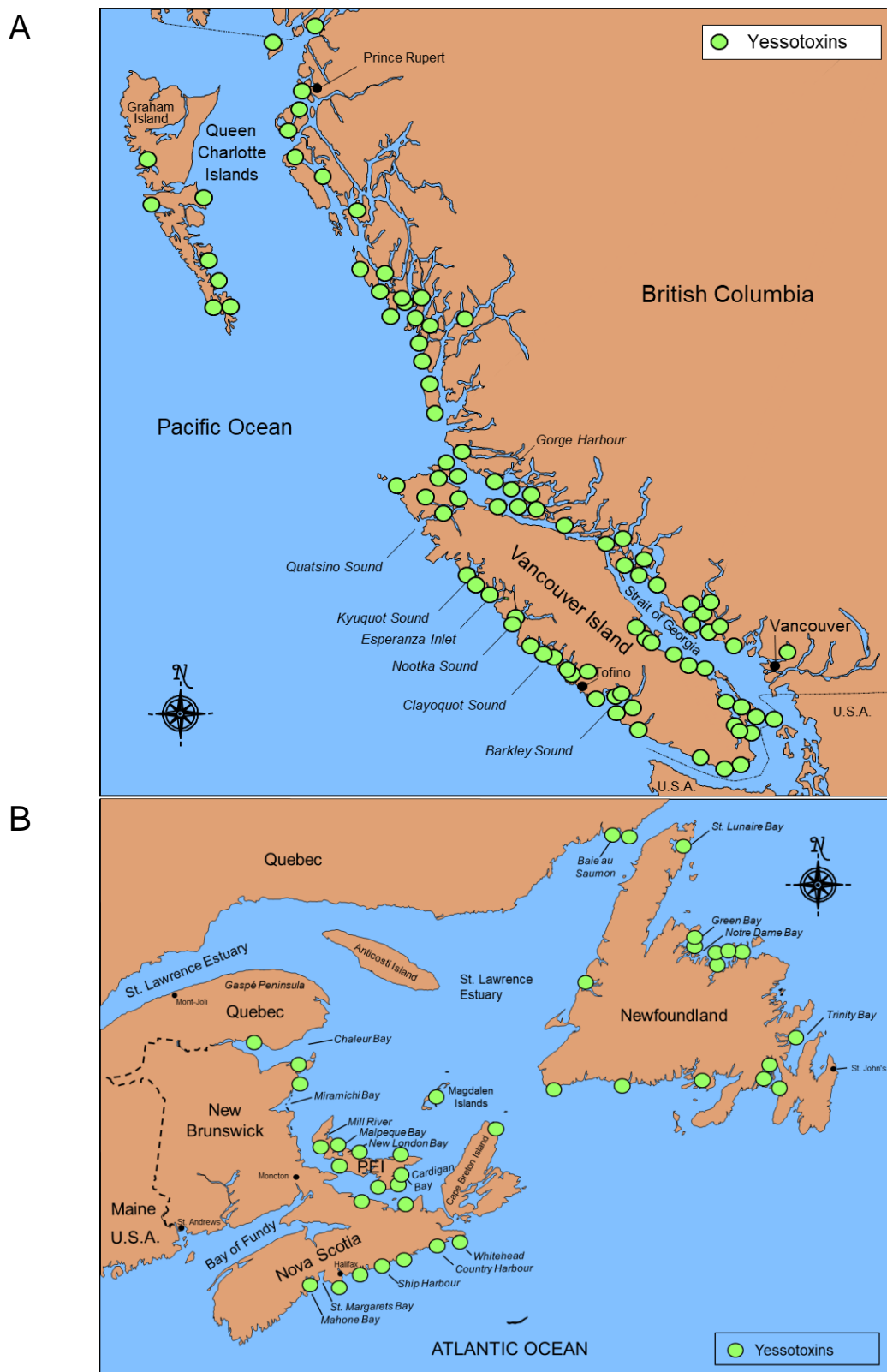


Figure 7. Location of yessotoxins (up to $12 \mu\text{g g}^{-1}$ YTX) along the coast of A) British Columbia; B) Atlantic Canada. There is no regulatory limit for YTXs in Canada.

One of the biogenetic origins of YTXs is the dinoflagellate *Protoceratium reticulatum*, as found in New Zealand, the UK, Norway, Spain, the Adriatic Sea, Chile, Japan, South Africa, Namibia (Ciminiello et al. 2003; Miles et al. 2005; Chikwililwa et al. 2019), as well as the Greenlandic Arctic (Sala-Pérez et al. 2016) and in plankton from the five Canadian Arctic regions (Grøntved and Seidenfaden 1938; Seidenfaden 1947; Bursa 1961a,b; Hsiao 1983; Hsiao et al. 1984; Percy et al. 1991; Simard et al. 1996; Lovejoy et al. 2002; Melnikov et al. 2002; Poulin et al. 2011; Dhifallah 2019; M. Poulin unpubl. data) ([Appendix 1](#)). An isolate of *P. reticulatum* from the Bedford Basin (NS) produced YTX at a concentration of 5 ± 3 pg cell⁻¹, and was confirmed by LC–MS/MS (Stobo et al. 2003; also mentioned in Armstrong and Kudela 2006). This was the first recorded evidence of a YTX-producing dinoflagellate in Canadian waters. Hand-picked single-cell extracts of *P. reticulatum* cells from Ship Harbour (NS) were analyzed by a micro-sampling and extraction procedure coupled with micro-column LC–MS/MS, and 20–40 fmol cell⁻¹ of YTX was detected (Quilliam et al. 2008). This species has also been reported to bloom in BC during July and August (Cassis 2007). In August 2011, it was present in water samples from Conville Bay (BC), 40 km from Cortes Island, where YTX was detected in mussels (see above); the maximum cell count was 7.5×10^4 cells L⁻¹ on August 10 (N. Haigh unpubl. data). Limited monitoring of phytoplankton during 2012 to 2017 revealed that *P. reticulatum* cell concentrations peaked at the same time as shellfish YTX concentrations (Rourke and Haigh 2019). Cysts of *P. reticulatum* were isolated from sediment trap samples in Effingham Inlet (lower west coast of Vancouver Island, BC) (Mertens et al. 2012). Cysts were also found in early Holocene sediments from Saanich Inlet, BC (Mudie et al. 2002). A bloom of *P. reticulatum* was identified as “the main probable cause” of the die-off of juvenile Pacific oysters (*Crassostrea gigas*) during the summer of 2002 in Jarvis Inlet, BC (Cassis 2005). The oyster behavioural response was a strong rejection, complete closure, and feeding cessation when exposed to cultures of *P. reticulatum*, but the possibility of the presence of YTXs was not investigated. Recently, BC isolates of *P. reticulatum* from Oak Bay and Saanich Inlet were reported to contain total YTXs at levels of 47.2 and 5.6 pg cell⁻¹, respectively (Wang et al. 2019a). This dinoflagellate has also been found in NL waters (Notre Dame Bay and Green Bay), although at concentrations too low to test for the presence of YTXs. However, an isolate from Saint-Pierre and Miquelon (France) (coastal NL) contained total YTXs at a level of 63.2 pg cell⁻¹ (Wang et al. 2019a). It should also be pointed out that all Greenlandic isolates of *P. reticulatum* produced compounds (not related to YTXs) that caused cell lysis of the cryptophyte *Rhodomonas salina*, a first report of allelochemical production by *P. reticulatum* (Sala-Pérez et al. 2016).

Other sources of YTXs are the dinoflagellates *Lingulodinium polyedrum* (previously called *Gonyaulax polyedra*) (Stobo et al. 2003; Armstrong and Kudela 2006; Howard et al. 2008) and *Gonyaulax spinifera* (Rhodes et al. 2006). Most Canadian waters are likely too cold to support the growth of *L. polyedrum*, which is usually found in coastal warm temperate and subtropical waters. However, cysts of *L. polyedrum* were found intermittently in sediment cores of Saanich Inlet (southeast Vancouver Island, BC), at sedimentary depths corresponding to the mid-1950s to the mid-1980s (Mudie et al. 2002); cysts of *G. spinifera* were also found in these same cores. *Lingulodinium polyedrum* has been reported in phytoplankton from the eastern Arctic, the Canadian Archipelago, and the Hudson Bay system (Roff and Legendre 1986; Lovejoy et al. 2002; McLaughlin et al. 2009) ([Appendix 1](#)).

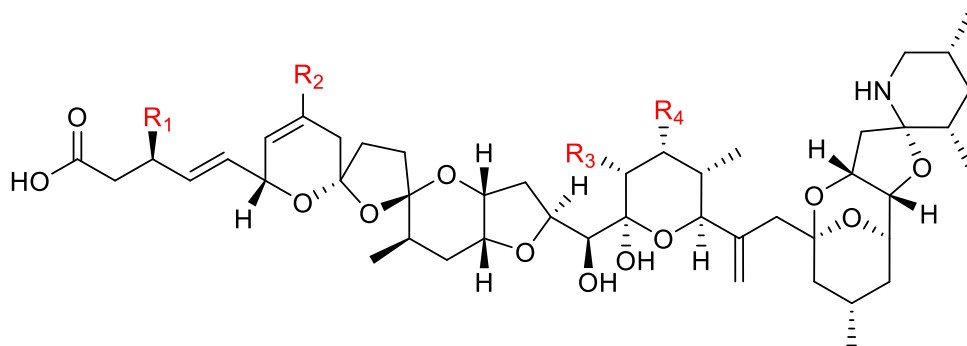
In 1990, a bloom of *G. spinifera* was observed off the southwest coast of Vancouver Island (BC), during August–September. Its bloom extended ~400 km alongshore and as far as 100 km offshore,

reaching up to $\sim 10^7$ cells L^{-1} . Although originally thought to be benign, the bloom was responsible for substantial shellfish mortality, primarily in Barkley Sound (Forbes et al. 1990; Borstad 1991; Forbes 1991b; Gower and Borstad 1991a,b). The mode of action of this mass mortality of clams, mussels and oysters was suggested to be oxygen depletion during the bloom decline (Heath and Lindsay 1993). The potential link between *G. spinifera* and YTX production at that location is still not known. This dinoflagellate is also regularly observed on the Canadian east coast: in the lower St. Lawrence Estuary and Gulf of St. Lawrence (Bérard-Therriault et al. 1999), including Pomquet Harbour (northern NS) (Kim et al. 2004) and inlets of PE (Bates and Strain 2006). In Chaleur Bay (NB), *G. spinifera* was present at a low concentration (20 cells L^{-1}), when YTX was detected at 0.11 $\mu g\ g^{-1}$, but without any harvesting closures (HAEDAT [CA-05-080](#)). This species is also present in Graves Shoal (Mahone Bay) and Ship Harbour, on the Atlantic coast of NS (Cembella et al. 2001b; Rafuse et al. 2007). Interestingly, *G. spinifera* was also found in phytoplankton from the eastern Arctic, the Canadian Archipelago, and the Hudson Bay system (Bursa 1961a; Simard et al. 1996; Harvey et al. 1997; Lovejoy et al. 2002; McLaughlin et al. 2009; Poulin et al. 2011; Dhifallah 2019) ([Appendix 1](#)). Clearly, these observations call for continued vigilance with regard to the possibility of the more wide-spread presence of YTXs in Canadian waters.

Aside from the mouse bioassay, YTXs can be detected by ELISA kits (Armstrong and Kudela 2006), LC–MS and LC–MS/MS (Finch et al. 2005; Stobo et al. 2005; McCarron et al. 2014a; reviewed in Speijers et al. 2011). Analyses for YTXs in Canada are carried out by the CFIA laboratory in Dartmouth (NS), using the UPLC–MS/MS method ([Appendix 3](#)). The Biotoxin Metrology group of the NRC, in Halifax (NS), produces certified calibration solutions for the determination of YTX and homoyessotoxin (hYTX) and a matrix reference material for YTXs by chemical analysis methods ([NRC Certified Reference Materials](#); [Wikipedia](#); Quilliam 2006; McCarron et al. 2017).

5.3 Azaspiracid group

Azaspiracids (AZAs) ([Structure 6](#)) belong to the most recent group of toxins, which were discovered as a result of shellfish consumption (Rossini and Sala 2014). At least 62 analogues have been reported (Krock et al. 2019), some of which are the result of metabolism within the shellfish or of heating during cooking (McCarron et al. 2009; Salas et al. 2011; Jauffrais et al. 2012a; Kilcoyne et al. 2014a,b, 2018; Hess et al. 2016; Ji et al. 2018; Krock et al. 2019). By convention, AZAs are numbered chronologically, depending on their time of discovery. This results in a listing of AZAs that are a mixture of those produced by dinoflagellates plus those that are shellfish metabolites.



Structure 6. The structure of azaspiracid and its analogues. The most common analogue, AZA1, has the substituents R_1 , R_2 , $R_4 = H$, $R_3 = CH_3$.

The first confirmed event occurred in November 1995, when at least eight people in the Netherlands became ill after consuming blue mussels (*M. edulis*) that were cultivated in Killary Harbour, on the west of Ireland (Satake et al. 1998). A new polyether marine toxin, AZA1, that has an unusual azaspiro ring structure (hence the name azaspiracid) was identified as the major causative agent of that incident (Satake et al. 1998), and the new syndrome was named azaspiracid poisoning (AZP). Since then, AZAs have been found throughout Europe, northwest Africa, Chile, Argentina, Mexico, China, Korea, and Japan (Taleb et al. 2006; Furey et al. 2010; Salas et al. 2011; Krock et al. 2012; Gu et al. 2013; García-Mendoza et al. 2014; Tillmann et al. 2016), and in numerous shellfish species (James et al. 2003; Salas et al. 2011), indicating that they are a worldwide phenomenon. However, to date, all AZA poisoning events have been associated with shellfish originating from Ireland (Pelin et al. 2019). In Canada, NRC and CFIA data show that, since 2005 (Table 1), AZAs have so far been found at low levels ($<0.02 \mu\text{g g}^{-1}$ AZA_{eq}) at 13 sites, and only on the east coast: 2 in NL, 3 in NS, 2 in NB, and 6 in QC (4 of which are on the Magdalen Islands) (Fig. 8). It should be noted that on the U.S. west coast and near Naples (Italy), strains of *Azadinium* have been identified that produce AZAs but no AZA1–3 (Krock et al. 2019).

The symptoms of AZP in humans are similar to those described for DSP (Section 4.1), and occur within 3 to 18 h after consuming contaminated shellfish; they include nausea, vomiting, abdominal cramps and diarrhea, which disappear within 2 to 5 days (Ryan et al. 2011). No lethality or long-term effects of AZP have been reported to date, although they have been shown to target the liver, lung, pancreas, thymus, spleen and the small intestine (Furey et al. 2010). However, they are potential tumorigenic compounds (Nzoughet et al. 2009). Pelin et al. (2019) recently determined that AZA1–3 induced a significant concentration-dependent increase of mitochondrial dehydrogenases activity in human hepatocytes.

As phycotoxins may co-occur in shellfish meat, Ferron et al. (2016) evaluated toxicological interactions between the lipophilic toxins OA, AZA1, PTX2, and YTX. In binary mixtures, the presence of both OA and AZA1 resulted in additive toxicity. As well, AZA1/YTX mixtures displayed a synergistic effect. Clearly, tertiary and quaternary combinations of these, and other phycotoxins, would be the next step in phycotoxin mixture studies. The chemistry, ecology, toxicology, pharmacology, risk assessment, genotoxic potential, and health considerations of AZAs have been reviewed (EFSA 2008c; Twiner et al. 2008, 2014; Furey et al. 2010; Ryan et al. 2011; Hess et al. 2014; Kilcoyne et al. 2014c; Doerr et al. 2016).

Because AZAs are a relatively new group of phycotoxins in Canadian waters, a harvesting closure level has yet to be established in Canada, and it is not mentioned in the CSSP Manual of Operations (CSSP 2019). In the EU, the maximum level of AZAs allowed in raw shellfish (although only AZA1, AZA2 and AZA3 are so far included because of their prevalence and toxicity; Pelin et al. 2019) is $160 \mu\text{g AZA}_{\text{eq}} \text{kg}^{-1}$ ($= 0.16 \mu\text{g g}^{-1}$) in “the whole body or any part edible separately” (European Commission 2002). Thus, in order for Canada to export product to the EU, the CFIA specifies that AZA toxins must not exceed $160 \mu\text{g AZA}_{\text{eq}} \text{kg}^{-1}$ (CFIA 2015a). It should be noted that the regulations specify that the shellfish be tested raw. However, AZA3 is only produced in significant quantities when the shellfish are cooked (McCarron et al. 2009). So, consumers are effectively not protected from AZA3. Interestingly, the acceptable limit for AZAs in whelks (*Buccinum undatum*) imported to NL for processing from Saint Pierre and Miquelon (a self-governing territorial overseas collectivity of France, situated at the entrance of Fortune Bay, NL) is the same as for the EU (CFIA 2015b). Tests for AZAs (as well as for DA, STX group toxins, OA group toxins, PTXs, and YTXs) are performed by the laboratory of the Institut Départemental d’Analyses et de Conseil (IDAC), in Nantes (France) and the La Direction des Territoires, de l’Alimentation et de la Mer analytical laboratory in Saint-Pierre and Miquelon (CFIA 2015b).

AZA1 was first detected in Canada by LC–MS/MS in Ship Harbour (NS), from May to November 2005, in blue mussels (*M. edulis*), at $0.007 \mu\text{g g}^{-1}$ (Quilliam et al. unpubl. data), as well as in SPATT bags (Garnett et al. 2006) (Table 1). Trace levels of AZA1, AZA2 and AZA3 have since been detected on both the NB and QC sides of Chaleur Bay, Bonavista Bay (NL), Neguac (Miramichi Bay, NB), and Baie des Belles-Amours (QC) (Fig. 8). No AZAs have yet been detected in waters of BC, although AZA59 was identified in dinoflagellates isolated from Puget Sound, WA (Kim et al. 2017) (see below). CFIA data show that the highest concentrations found to date have been at Country Harbour (NS) (in 2016) and the Magdalen Islands (in 2017) (Fig. 8), but concentrations in these samples have still been $<0.02 \mu\text{g AZA}_{\text{eq}} \text{g}^{-1}$ AZA1_{eq} . The conversion of AZAs in mussels, and subsequent decarboxylation upon heating (McCarron et al. 2009), were investigated for a select number of mussel (*M. edulis*) samples from NS and QC in 2008. This determined the presence of carboxylated AZA1 and AZA2 (AZA17 and AZA19, respectively), which decarboxylated upon heating to form AZA3 and AZA6, respectively (McCarron et al. unpubl. data). The study indicates that the AZA profiles in Atlantic Canada are roughly equivalent to those reported in Ireland.



Figure 8. Location of azaspiracids (up to $0.02 \mu\text{g g}^{-1}$ AZA) along the Canadian Atlantic coast. There is no regulatory limit for AZAs in Canada. No AZAs have been reported for the Canadian Pacific coast.

Of the 62 total analogues (Krock et al. 2019), there are currently 26 AZA variants of dinoflagellate origin, produced by four species in the family Amphidomataceae (Tillmann et al. 2014b, 2018; Kim et al. 2017; Tillmann 2018; Krock et al. 2019). Initially, the dinoflagellate *Protoperidinium crassipes* was proposed as the organism producing AZAs (James et al. 2003). However, it is thought that *P. crassipes*, a predatory alga, is a mere vector for the culprit species upon which it feeds (Tillmann et al. 2009; Furey et al. 2010). *Protoperidinium crassipes* is found in the Bay of Fundy (Martin and LeGresley 2008; Martin et al. 2009), although it has not been tested for the presence of AZAs. It has also been reported in phytoplankton from all Canadian Arctic regions, except from the western Arctic (Bursa 1961a; Sekerak et al. 1976a, 1979; Hsiao 1983; Simard et al. 1996; Melnikov et al. 2002; Dhifallah 2019) ([Appendix 1](#)).

A previously unknown dinoflagellate, *Azadinium spinosum*, was later confirmed as the source of AZAs (Krock et al. 2008; Tillmann et al. 2009, 2014a; Salas et al. 2011; Jauffrais et al. 2012a,b). Species now known to produce AZAs include *A. spinosum*, *A. poporum*, *A. dexteroporum*, and *Amphidoma languida*, although there is a new, nontoxicogenic ribotype of *A. spinosum* (Tillmann et al. 2019). Interestingly, in contrast to other toxigenic phytoplankton, *A. spinosum* strains in culture have shown no obvious change in their toxin profile under the various N- and P-nutrient or other environmental conditions (Li et al. 2016a). A strain of *A. spinosum* from the North Sea was grown in 300-L photobioreactors at the NRC (Halifax, NS) to study the production of AZAs under

different conditions (Salas et al. 2018). Mussels (*M. edulis*) accumulated dissolved AZAs, from lysed *A. spinosum* cells, to greater than the legal limit (Jauffrais et al. 2013).

New AZAs have been found in North Sea, Korean, Vietnamese, Japanese, and Chilean isolates of the dinoflagellate *A. poporum*, and in Irish, Spanish, and Norwegian isolates of *Amphidoma languida* (Krock et al. 2012, 2015, 2019; Tillmann et al. 2014b, 2017, 2018; Uchida et al. 2018b). *Azadinium dexteroporum* is the third *Azadinium* source of AZAs (Tillmann 2018; Krock et al. 2019). Although AZAs have been reported in east coast Canadian waters, the source of the toxins remains elusive. On the west coast of the U.S., AZA59 was identified from *A. poporum* strains isolated from Puget Sound, WA (Kim et al. 2017; Dai et al. 2019). Another potential source of AZAs is *Amphidoma languida* (Krock et al. 2019); the genus *Amphidoma* is closely related to the genus *Azadinium* (Krock et al. 2012; Tillman et al. 2012, 2017). In 2012, *Amphidoma languida* was identified by SEM in seawater samples collected near the islands of Saint-Pierre and Miquelon, in the Gulf of St. Lawrence near NL (Tillmann 2018). Reports of the presence of *A. poporum* and *Amphidoma languida* in nearby waters indicates that vigilance is required in monitoring for these species in Canada.

The most widely available method for AZA analysis is LC–MS/MS (McCarron et al. 2011b), which is the LST multi-class, multi-analyte method used by the CFIA ([Appendix 3](#)), and includes AZA1–3. There are currently no commercially available analytical standards (although non-certified, quantitative reference materials are available) and very limited toxicological data for most of the remaining AZA compounds. A derivatization method to enable fluorescent detection of AZAs is also available for laboratories that do not have LC–MS instrumentation (McCarron et al. 2011d). A methodology was recently described for cleaning up and concentrating AZAs from mussel extracts using polymer-bound boronic acid (Miles et al. 2018b). AZAs can also be quantified by immunological methods (Samdal et al. 2015; Leonardo et al. 2017, 2018b). The Biotoxin Metrology group of the NRC, in Halifax (NS), produces certified calibration solutions and mussel tissue reference materials for the determination of AZAs ([NRC Certified Reference Materials](#); [Wikipedia](#); Quilliam 2006; Perez et al. 2010; McCarron et al. 2015, 2017).

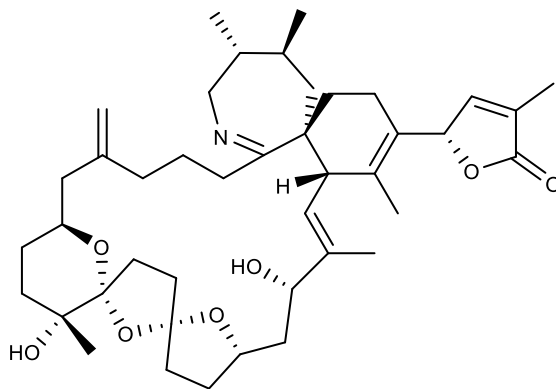
6.0 Cyclic Imine Group Toxins

Cyclic imines are a class of lipophilic phycotoxins that includes spirolides, pinnatoxins, pteriatoxins (not detected in Canadian waters) and gymnodimines (Cembella and Krock 2008; Munday 2014; Mabe and Zakarian 2015; Molgó et al. 2017). They act on neural receptors and bioaccumulate in seafood. This toxin group is noted for its very high acute toxicity in mice upon intraperitoneal injections of lipophilic extracts, and also for how fast-acting the toxins are. No information is available on their possible carcinogenicity, reproductive toxicity or genotoxicity (Pulido et al. 2011). Because they have not yet been linked to human poisoning, they are not regulated in the EU. However, the EFSA still had concerns, and a recent French risk assessment of pinnatoxins reached a similar conclusion that monitoring should continue (ANSES 2019). A resulting study of cyclic imines in European shellfish concluded that although it is unlikely that they pose a potential health risk through shellfish consumption, further information is required about their prevalence and their exposure to humans in order to perform a conclusive risk assessment (Rambla-Alegre et al. 2018). Therefore, cyclic imines should be included in shellfish safety monitoring programs. The global distribution, source organisms, chemistry, metabolism, detection, toxicology, and biosynthesis

of cyclic imine group toxins has been reviewed (Cembella and Krock 2008; Munday 2008a; Chatzianastasiou et al. 2011; Otero et al. 2011; Molgó et al. 2014, 2016, 2017; Davidson et al. 2015; Stivala et al. 2015).

6.1 Spirolide toxins

The world's first discovery of spirolide toxins (SPXs) ([Structure 7](#)) was in 1991 ([Table 1](#)), when the lipophilic fraction of digestive glands of blue mussels (*M. edulis*) and scallops (*Placopecten magellanicus*) from aquaculture sites on the south shore of NS (Graves Shoal [Mahone Bay] and Ship Harbour) ([Fig. 9B](#)) resulted in the rapid death of mice after intraperitoneal injection (Hu et al. 1995c, 1996a; Richard et al. 2001). In BC, SPXs were first detected in 2011, at multiple locations (CFIA and NRC data) ([Table 1](#)). Since then, CFIA data show that SPXs have been detected at low levels (up to $0.12 \mu\text{g g}^{-1}$ SPX1) in Canadian bivalves at 371 locations, on both the east and west coasts, including 214 in BC ([Fig. 9A](#)), 23 in NB, 39 in NL, 49 in NS, 13 in PE, and 33 in QC ([Fig. 9B](#)). Aside from rapid death, the behavioural and neurological symptoms in mice are very different from those associated with other known LSTs, but are still neurotoxic in mammalian systems (Pulido et al. 2001; Richard et al. 2001; Munday et al. 2012; Molgó et al. 2016).



Structure 7. The structure of 13-desmethylspirolide C, one of the most common spirolide toxins. Many structural variations have been observed with different substituents and ring systems.

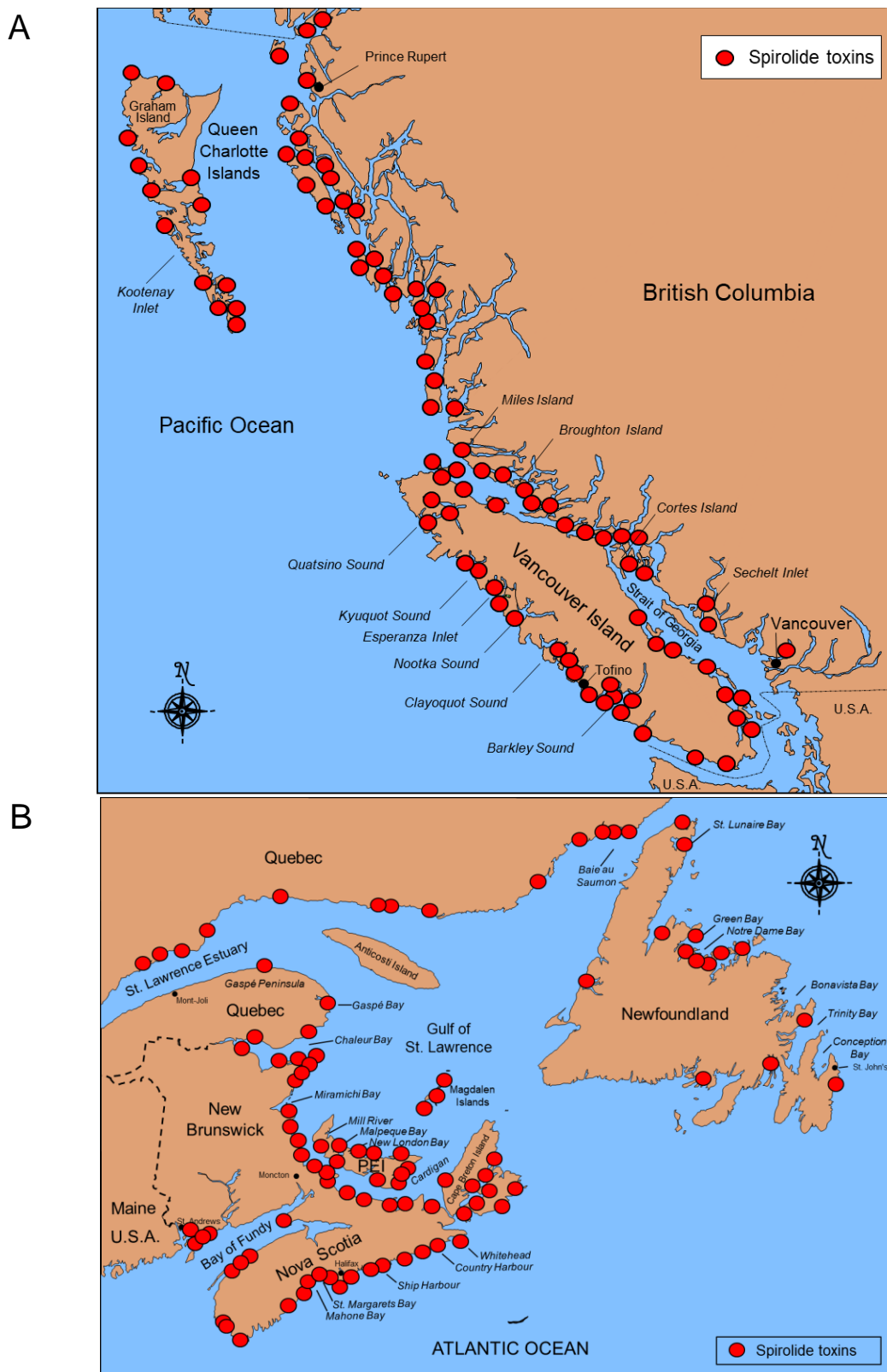


Figure 9. Location of spirolide toxins (up to $0.12 \mu\text{g g}^{-1}$ SPX1) along the coast of A) British Columbia; B) Atlantic Canada. There is no regulatory limit for SPXs in Canada.

These “fast-acting toxins” were named spirolides because their molecular structure contains a spiro ring (Hu et al. 1995c). Some non-specific symptoms, such as gastric distress and tachycardia, were recorded in individuals who consumed shellfish in NS during times when spirolides were known to be present, but these could not definitively be ascribed to SPXs (Richard et al. 2001). Otherwise, there is no conclusive evidence that human health issues have been associated with consumption of shellfish contaminated with SPXs, and there was no cytotoxicity when these toxins were incubated with various cultured mammalian cell lines (Munday et al. 2012). Nevertheless, certain SPX analogues have been shown to target mouse, rat (Pulido et al. 2001, 2011; Richard et al. 2001) and human (Wandscheer et al. 2010) muscarinic acetylcholine receptors in the brain, and they are highly toxic to mice and rats by intraperitoneal injection (Gill et al. 2003; Munday et al. 2012). SPXs also proved toxic to mice through oral ingestion (Richard et al. 2001; Munday et al. 2012). More data are required before regulatory levels of SPXs can be established (FAO/IOC/WHO 2005); they are therefore currently not regulated in the EU (EFSA 2010), U.S. (Gribble et al. 2005), or Canada. However, because of their widespread distribution and potential human health effects, SPXs are monitored by the CFIA to generate data for future risk assessments.

Structural work carried out at the Halifax laboratory of the NRC, using LC–MS/MS and nuclear magnetic resonance (NMR) spectroscopy, initially characterized the macrocycles from NS as existing in six different forms, designated as SPX-A, B, C, D, E and F, including certain desmethyl derivatives of SPX-C and SPX-D (Hu et al. 1995c, 2001). The latter SPXs are toxic due to their cyclic imine ring, whereas SPX-A and SPX-B, without the ring moiety, are less toxic (Munday et al. 2012). Although SPX-H contains the cyclic imine moiety, it does not show toxicity in the mouse assay, suggesting that the presence of this moiety is not the only structural requirement for toxicity (Roach et al. 2009). Two SPX derivatives, SPX-E and SPX-F, were isolated from shellfish extracts and are suspected to be shellfish metabolites of SPX-A and SPX-B (Hu et al. 1996a). They showed no activity in the mouse bioassay (Hu et al. 1996a), which is probably due to the open imine ring. It has also been shown that some SPXs can be metabolized in shellfish to form fatty acid acyl esters (Aasen et al. 2006), which can greatly complicate chemical analysis. The toxicological significance of the esters has not yet been determined. The relative stereochemistries of SPX-B, SPX-D and 13-desmethylSPX-C, from cultures and shellfish extracts, have also been assigned (Falk et al. 2001). This will help with future toxicological studies and comparisons with pinnatoxins, putative toxins that are also of dinoflagellate origin (Section 6.2). MS/MS and NMR studies have yielded structures on 17 SPX analogues, but over 50 different SPXs have been observed (Zurhelle et al. 2018; Quilliam et al. unpubl. data).

The search for the biological origin of SPXs continued after 1991, at various sites in NS. For several years, SPXs were found annually at high concentrations in bulk phytoplankton samples, during early May through July, at Graves Shoal (Mahone Bay) and Ship Harbour (on the eastern shore) (Cembella et al. 1997, 1998). Those studies found an abundance of thecate gonyaulacoid dinoflagellates and non-motile “golden balls” of 42 µm mean diameter in phytoplankton samples collected at multiple sites where SPXs were detected in the water or shellfish, indicating an association with the phytoplankton. The SPX profiles of bulk phytoplankton extracts from nearby locations were different, but reflected the profiles of the *Alexandrium ostenfeldii* cells isolated from those same locations (Cembella et al. 1999, 2001a; Gribble et al. 2005). For example, the toxin profiles from plankton containing *A. ostenfeldii* collected from Ship Harbour were dominated by SPX-A, B, C, and 13-desmethylSPX-C, whereas those collected at Graves Shoal, less than 100 km southwest, were remarkably different, with SPX-B, D, and isomer SPX-D2 as the major analogues

(Cembella et al. 2001a,b). SPXs have also been recovered from SPATT bags deployed at Ship Harbour and Lunenburg Bay (NS), from May to November 2005 (Garnett et al. 2006).

The exact origin of SPXs remained cryptic until SPX production was demonstrated in a unialgal isolate of *A. ostenfeldii* from Ship Harbour, NS (Cembella et al. 2000). Cultured isolates produced SPX congeners C, D, H, 13-desmethylSPX-C, 20-methylspirolide G, and desmethylspirolide D, but not SPX-A and SPX-B (Cembella et al. 1999; Hu et al. 2001; Lewis et al. 2008a; Munday et al. 2012). SPX-H and SPX-I were later isolated from another Ship Harbour culture of *A. ostenfeldii* (Roach et al. 2009). *Alexandrium ostenfeldii* has been identified in Bay of Fundy phytoplankton samples (Martin et al. 2001b, 2014a,b), and isolates from Lime Kiln contained SPX-A and SPX-B (Gribble et al. 2005). Cultured isolates of *A. ostenfeldii* from the Gulf of Maine contained SPX-A and SPX-B (Munday et al. 2012), as well as 13-desmethylSPX-C, SPX-C and SPX-D, plus SPX-C2 and SPX-D2, which are unidentified isomers of SPX-C and SPX-D, respectively (Gribble et al. 2005). Two new SPX-like compounds, SP-1 and SP-2, were discovered in cultured isolates of *A. ostenfeldii* obtained from Limfjorden, Denmark (MacKinnon et al. 2004, 2006); these have not yet been reported for Canadian isolates. No traces of SPX-E or SPX-F were ever found in either bulk phytoplankton from the field or in cultured *A. ostenfeldii* cells (Hu et al. 2001). Rather, they were present only in shellfish extracts, indicating that they are likely chemical or enzymatic hydrolysis products formed in the shellfish.

SPX profiles of nine *A. ostenfeldii* isolates from NS (Shelbourne, Graves Island, Halifax harbour, Ship Harbour) were determined by LC–MS/MS and LC–HRMS (Qiu et al. 2018). All isolates produced high SPX concentrations, but with varied profiles. For example, the predominant analogue in two of the strains was 13-desmethylSPX-C with a high molar percentage (~76%), whereas the dominant analogues in the other strains were SPX-C (~40 mol%) and 20-Me SPX-G (~50 mol%). More than 20 SPX analogues were detected in one of the isolates, demonstrating the potential for high structural diversity of SPX. These isolates also produced trace amounts of STX group toxins. No SPXs were detected in *A. catenella* strains from those locations.

Hand-picked *A. ostenfeldii* cells from Ship Harbour could be analyzed by a micro-sampling and extraction procedure coupled with micro-column LC–MS/MS for the concentration and profile of SPXs (Quilliam et al. 2008). The SPX concentration in cysts isolated from sediment samples in Ship Harbour doubled after a year in storage at 2 °C (Lewis et al. 2008b), suggesting continued production when cells are in the resting stage.

Although *A. ostenfeldii* has not been identified in NL plankton samples thus far, SPXs have been detected in samples collected at Tea Arm, Strong Island Sound in 2010 (McCarron et al. 2012), as well as at 38 other sites ([Fig. 9B](#)), suggesting this species does occur there.

Alexandrium ostenfeldii has been reported in BC waters (Taylor and Horner 1994; Taylor and Haigh 1996), including English Bay, Vancouver (Taylor and Harrison 2002) and Saanich Inlet, Vancouver Island (Kremp et al. 2014), as well as in waters of WA State (Steidinger and Tangen 1996). During the first incident of DSP in BC (Cortes Island) in August 2011 (Haigh 2012; Esenkulova and Haigh 2012; Taylor et al. 2013; [Section 4.5](#)), samples of mussels (*Mytilus californianus*) were also reported to contain trace levels of 13-desmethylSPX-C (McCarron et al. 2014a). This incident occurred during a bloom of various species of *Dinophysis* (Esenkulova and Haigh 2012; Taylor et al. 2013), and no mention was made of the presence of *A. ostenfeldii*, the usual

source of SPXs. However, it should be noted that the source of the low levels of SPXs often found in shellfish samples from both Canadian coasts may be toxic cysts that are resuspended, although this has not been studied. CFIA data show SPXs at 214 sites in BC ([Fig. 9A](#)). Aside from Canadian waters, *A. ostenfeldii* has also been found on the U.S. east coast, and in Iceland, the Faroe Islands, Denmark, Norway, Sweden, Finland, Scotland, Ireland, United Kingdom, Spain, Belgium, Egypt, WA State (U.S.), Russia (Kamchatka Peninsula), Japan, China, New Zealand, and Peru (summarized by Maclean et al. 2003; Gribble et al. 2005; Kremp et al. 2014).

The discovery of *A. ostenfeldii* as a producer of the novel SPXs was surprising, as this species was thought to produce STX group toxins (Section [2.4.2.2](#)). Certain Scandinavian (Cembella et al. 2000) and Dutch (Martens et al. 2017) strains produce both SPXs and STX group toxins. These differences in SPX production by the various *A. ostenfeldii* strains from around the world, as well as the ability of only some strains to also produce STX group toxins, is now better understood by the delineation of six distinct genetic groups in the *A. ostenfeldii* species complex (Kremp et al. 2014). Group 5 represents a monophyletic group of *A. ostenfeldii* strains originating from the NW Atlantic, mainly the Gulf of Maine and Canada, the NW coast of Iceland and the West coast of Norway. Some Group 1 strains were from estuaries at the U.S. east coast (New River, North Carolina and Narragansett Bay, Rhode Island). Confirming previous studies, Group 5 strains produced a mixture of different spirolides, but primarily SPX-A. Interestingly, Group 6 contains an *A. ostenfeldii* strain from Saanich Inlet (southeast Vancouver Island, BC) that did not contain measurable amounts of STX group toxins or SPXs (Kremp et al. 2014). This brings into question the source responsible for the widespread distribution of SPXs in BC waters ([Fig. 9A](#)).

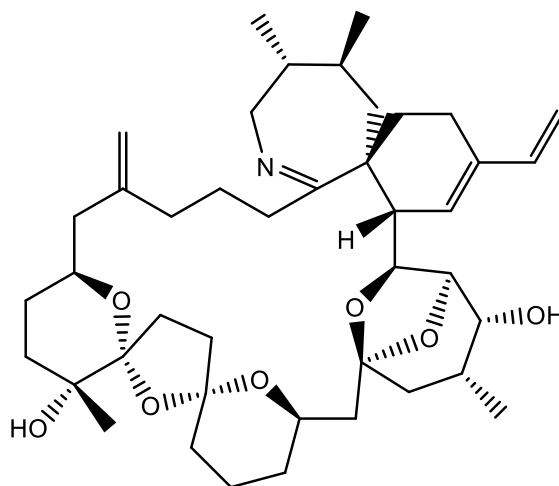
The Kremp et al. (2014) study also included a second *Alexandrium* species, *A. peruvianum*, reported to produce SPXs. This species was found in waters of Ireland (Touzet et al. 2008), North Carolina (Tomas et al. 2012) and Narragansett Bay (Borkman et al. 2012), and was also reported to produce STX group toxins (Section [2.4.2.2](#)) and GYMs (Section [6.3](#)). However, because this species is morphologically similar to *A. ostenfeldii*, the two had sometimes been confused. Kremp et al. (2014) concluded that the present morphological delineation of *A. peruvianum* from *A. ostenfeldii* is not well supported. Furthermore, their phylogenetic and morphological data indicate that *A. peruvianum* should not be considered a distinct species, and that the name should be treated as synonym of *A. ostenfeldii*.

Once the chemical structure of SPXs and the source organism were identified, studies were undertaken to determine the effects of environmental factors on *A. ostenfeldii* growth and SPX production. Using a clonal isolate from Ship Harbour (NS), Maclean et al. (2003) determined that this strain was euryhaline, showing no difference in growth rate over a range of salinities (25–33). It can also grow at a high inorganic nitrogen concentration but it has a narrow optimal light range. In contrast, neither growth rate, SPX production, nor toxin composition (% molar) were affected by environmental factors. Only low concentrations of extracellular SPXs were found. The authors thus concluded that SPXs are neither stress-inducible compounds nor exotoxins.

Analysis for SPXs in Canada is carried out by the Dartmouth (NS) laboratory of the CFIA, using the UPLC–MS/MS method ([Appendix 3](#)). The Biotoxin Metrology group of the NRC, in Halifax (NS) (Quilliam 2006), produces a certified calibration solution and mussel tissue reference material for the determination of SPXs by chemical analysis methods ([NRC Certified Reference Materials](#); [Wikipedia](#); (Quilliam 2006; McCarron et al. 2017).

6.2 Pinnatoxins

Pinnatoxins (PnTXs) ([Structure 8](#)) were initially detected in the pen shell bivalve (*Pinna attenuata*) from China in 1990, and now include eight PnTXs (A to H) and three PnTX metabolites (pteriatoxins A to C), isolated from shellfish and dinoflagellates (Selwood et al. 2010; Mabe and Zakarian 2015; Stivala et al. 2015). They have since been found in shellfish or dinoflagellates from Japan, Australia, New Zealand, Norway, France, the Arabian Gulf, and now Canada (see below). As with the other “fast-acting toxins” in this group, PnTXs cause rapid death in the mouse bioassay for LSTs, and may have a higher oral toxicity than SPXs (Mabe and Zakarian 2015). However, there is thus far no evidence that these toxins have caused any human illnesses or fatalities (Sosa et al. 2020). There is no information on the symptoms of intoxication by PnTXs, nor on the histology of animals dosed with PnTX derivatives (Pulido et al. 2011). Therefore, PnTXs are not yet included in a list of contaminants to be monitored in the international trade (Pulido et al. 2011), nor are they mentioned in the CSSP (2019); however, a recent French risk assessment suggests that PnTXs should be included in regular biotoxin monitoring programs (ANSES 2019). Although PnTXs were previously incorrectly ascribed to act on calcium channels (Araoz et al. 2011), PnTX-A has now been shown to be a potent inhibitor of nicotinic acetylcholine receptors, leading to respiratory paralysis in mice (Mabe and Zakarian 2015). Despite their highly complex molecular structure, several analogues, including PnTXs-A–C and PnTX-G and pteriatoxins A–C, have been synthesized (reviewed in Mabe and Zakarian 2015). Synthetic PnTX-A and PnTX-G blocked, in a reversible manner, nerve-evoked muscle contraction in a mouse muscle preparation (Benoit et al. 2019), and PnTX-G caused death of mice after 30 min (Sosa et al. 2020).



Structure 8. The structure of pinnatoxin-G, one of the most common pinnatoxins. Many structural variations have been observed with different substituents and ring systems.

The dinoflagellate *Vulcanodinium rugosum*, a new genus and species (Nézan and Chomérat 2011), has been identified as the causative organism for PnTXs (Rhodes et al. 2011). It has been recorded in New Zealand, Australia Japan, China, Hawaii, and France (references in Zeng et al. 2012). A strain of *V. rugosum*, isolated from Ingril Lagoon (France), produced PnTX-G in culture, although this species was absent from the water column during a prolonged period of shellfish contamination

(Hess et al. 2013). Studies have described *V. rugosum* growth in culture (Abadie et al. 2015, 2016) and in natural settings (Abadie et al. 2018).

A study was carried out by the NRC and CFIA to examine lipophilic toxins in blue mussels (*M. edulis*) collected by the CFIA for regulatory monitoring purposes from sites in eastern Canada, during June 2008 to September 2011 (McCarron et al. 2012). The earliest detection was from a stored sample, collected on July 7, 2008, at Neguac Bay, NB ([Table 1](#)), in which PnTX-A and PnTX-G were confirmed by LC–MS/MS. The average PnTX-G concentration measured across all sites was 12 $\mu\text{g kg}^{-1}$, and the maximum was 83 $\mu\text{g kg}^{-1}$ for a sample harvested in Dildo Run Provincial Park, on the northern coast of NL ([Fig. 10B](#)), in July 2011. In addition, this study was the first to report fatty acid acyl esters of PnTXs, which may possess toxicities similar to the nonacylated compounds. Hydrolysis of the esters during digestion could potentially result in exposure to significantly higher PnTX levels.

The CFIA has measured PnTXs at low levels (up to 0.71 $\mu\text{g g}^{-1}$ total PnTX-G) from 431 sites on both coasts of Canada: 162 in BC ([Fig. 10A](#)), 26 in NB, 55 in NL, 68 in NS, 25 in PE, and 95 in QC ([Fig. 10B](#)). The PnTX-producing organism in Canadian waters has not yet been identified. However, because PnTXs are so widely distributed, it is important that they be screened as part of routine shellfish monitoring exercises to generate a dataset to be used for risk assessment, as suggested by a recent French risk assessment (ANSES 2019).

In BC ([Fig. 10A](#)), during the first incident of DSP on Cortes Island in August 2011 (Haigh 2012; Esenkulova and Haigh 2012; Taylor et al. 2013; [Section 4.5](#)), samples of mussels (*Mytilus californianus*) were also reported to contain trace levels of PnTX-G (McCarron et al. 2014a). This incident occurred during a bloom of various species of *Dinophysis* (Esenkulova and Haigh 2012; Taylor et al. 2013), although a source organism for PnTXs was not identified.

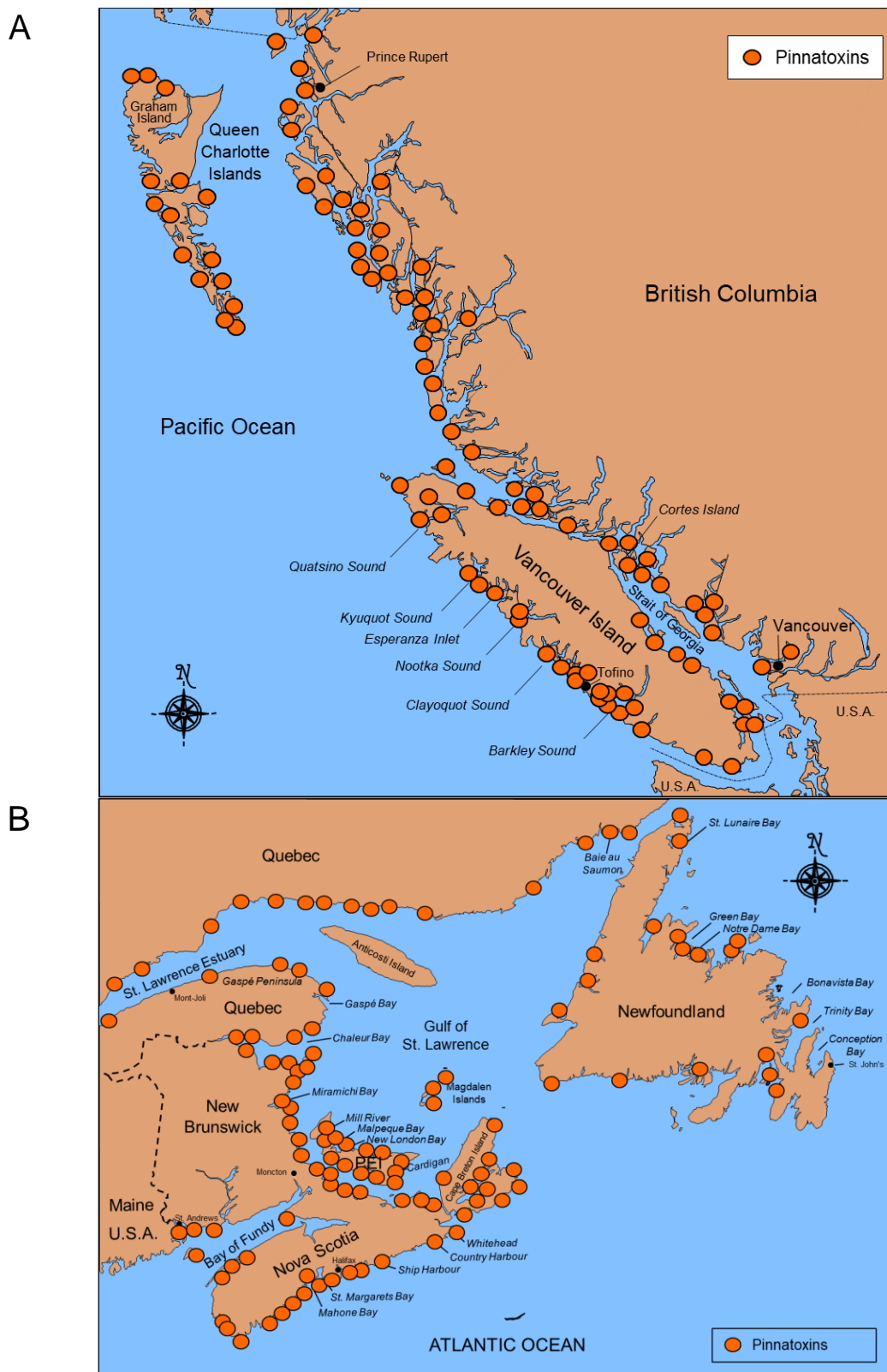
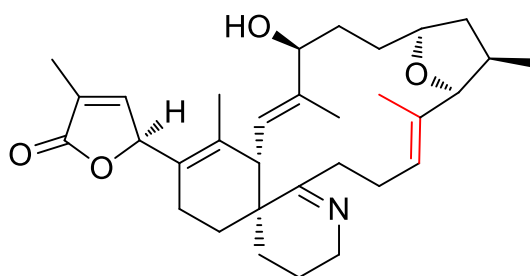


Figure 10. Location of pinnatoxins (up to $0.71 \mu\text{g g}^{-1}$ total PnTX-G) along the coast of A) British Columbia; B) Atlantic Canada. There is no regulatory limit for PnTXs in Canada.

The mouse bioassay and LC–MS analysis are used for the detection of PnTXs (Mabe and Zakarian 2015). In Canada, PnTXs are analyzed at the Dartmouth (NS) laboratory of the CFIA, using the UPLC–MS/MS method ([Appendix 3](#)). The Biotxin Metrology group of the NRC, in Halifax (NS), produces certified calibration solutions for the determination of PnTX-G by chemical analysis methods ([NRC Certified Reference Materials](#); Quilliam 2006).

6.3 Gymnodimines

Gymnodimine (now gymnodimine A; GYM) ([Structure 9](#)) was first isolated in 1994 from oysters (*Tiostrea chilensis*) collected off the coast of New Zealand, where it co-occurred with several incidents of neurotoxic shellfish poisoning (Seki et al. 1996). However, no cases of poisoning have been shown to be associated with the presence of this toxin, and no specific syndrome can be attributed to consumption of GYM by humans. No regulatory levels have been established in Canada or elsewhere. During the mouse bioassay, GYM is more toxic by intraperitoneal injection than when administered orally by gavage (Pulido et al. 2011). Fatty acid acyl esters of GYM have also been reported (de la Iglesia et al. 2013). A review of GYM is included in Stivala et al. (2015).



Structure 9. The structure of GYM, the most common gymnodimine. A number structural analogues have been observed.

The toxin was originally shown to be produced by *Gymnodinium* cells from New Zealand (Seki et al. 1995, 1996), hence the name GYM. However, this dinoflagellate genus has undergone considerable taxonomic revision (Section [2.2](#)). The toxin-producing organism from New Zealand has now been described as a new species, *Karenia selliformis* (Haywood et al. 2004). A strain of *K. selliformis* from Tunisia was also shown to produce GYM (Medhioub et al. 2009). Studies with *K. selliformis* revealed that the total toxicity of GYM and ester derivatives should be considered in order to protect the seafood safety (Li et al. 2018c). Another dinoflagellate, *Alexandrium ostenfeldii*, from Narragansett Bay (Rhode Island, U.S.), was also reported to produce GYM (as *A. peruvianum*; Borkman et al. 2012). As well, a strain from the Baltic Sea produced GYM-D plus other congeners (Harju et al. 2016). A strain of *A. ostenfeldii* from the Netherlands produced GYM-A and 12-methylgymnodimine A (Martens et al. 2017), and two novel GYM: 16-desmethylGYM-D and GYM-E (Zurhelle et al. 2018), bringing the number of GYM derivatives to six. This strain also produced SPXs (Section [6.1](#)). GYM-like compounds have tentatively been identified in *A. ostenfeldii* from NS (Qiu et al. 2018). Canadian phytoplankton monitoring programs should be alert for the presence of *K. selliformis* and the possibility of GYM production by *A. ostenfeldii*.

CFIA data from LC–MS analyses show that, since 2003 ([Table 1](#)), low levels (up to 0.04 µg g⁻¹) of GYMs have been detected in bivalves at 97 locations in Canada, mostly in BC (91 sites), as well as less frequent trace levels in NS (2 sites), QC (3 sites), and NL (1 site) ([Fig. 11](#)). The one published report of GYMs in Canadian waters was when lipophilic toxins were extracted during the first incident of DSP in BC (Cortes Island) in August 2011, associated with various species of *Dinophysis* (Esenkulova and Haigh 2012; Taylor et al. 2013; Section [4.5](#)). In addition to OA group toxins, samples of mussels (*Mytilus californianus*) were also reported to contain trace levels of GYMs (McCarron et al. 2014a). However, *Dinophysis* spp. are not a known source of GYM. Thus, the source of GYMs in BC waters remains unknown, in spite of its widespread distribution there.

The CFIA currently monitors for GYM-A using UPLC–MS/MS ([Appendix 3](#)). The Biotoxin Metrology group of the NRC, in Halifax (NS), produces certified calibration solutions for the determination of GYMs by chemical analysis methods ([NRC Certified Reference Materials; Wikipedia](#); Quilliam 2006).

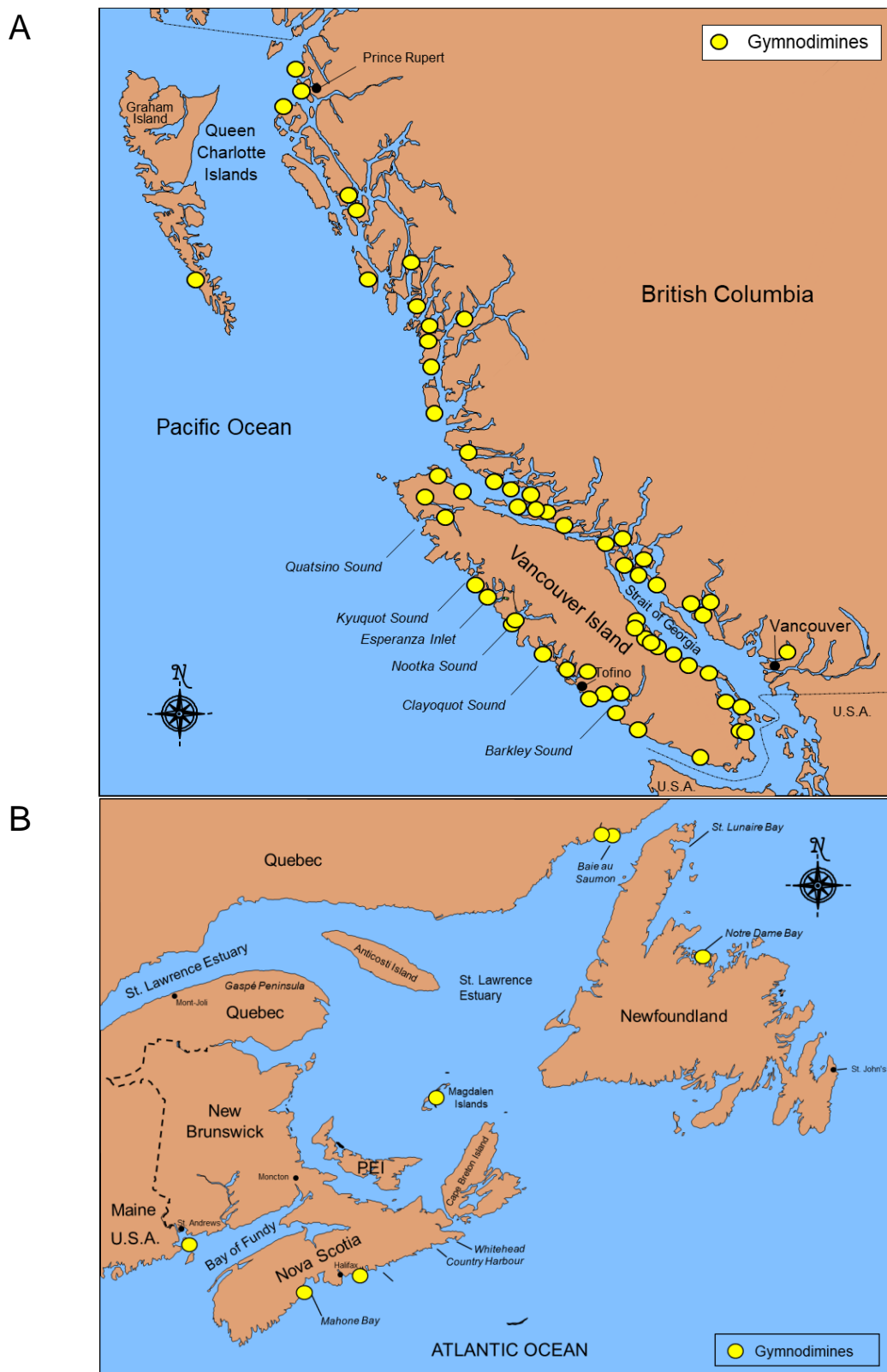


Figure 11. Location of gymnodimines (up to $0.04 \mu\text{g g}^{-1}$ GYM) along the coast of A) British Columbia; B) Atlantic Canada. There is no regulatory limit for GYM in Canada.

7.0 Other Toxic or Harmful Algae of Concern

7.1 Diatoms

Several diatom species are known to be harmful, especially to farmed salmon. Because they have the potential to cause harm, or have already caused harm in Canadian waters, they are discussed in this review. These diatoms include *Chaetoceros* spp., *Skeletonema costatum*, *Thalassiosira aestivalis*, *T. rotula*, *Corethron* spp., *Leptocylindrus minimus*, *Ditylum brightwellii* and *Eucampia zodiacus*.

7.1.1 *Chaetoceros* spp.

Two nontoxic but harmful diatom species, *C. convolutus* and *C. concavicornis* (also known as *C. concavicornis*), are reported to have killed fish in BC since at least 1961 (Bell 1961; Haigh and Taylor 1990; Kent 1992; Taylor 1993). Monetary losses to salmonid aquaculture operations from some events have been similar to those caused by *Heterosigma akashiwo*, e.g. CAN\$3.0–3.9 million in 1987 (Black 1991; Taylor 1993). So far, Canada and the U.S. (in adjacent waters of Puget Sound, WA; Rensel et al. 1989) are the only areas to have reported serious commercial losses due to these *Chaetoceros* species. In Alaska, where commercial salmon farming is not permitted, blooms of *C. convolutus* have hindered efforts at raising Pacific (Chinook and chum) salmon in non-profit hatcheries (Farrington 1988). A small loss of juvenile (<500 g) Atlantic salmon was reported at a farm in Chile in 1991 (Clément and Lembeye 1993).

In 1975, sockeye salmon (*Oncorhynchus nerka*) mortalities were observed in experimental netpens at DFO's Pacific Biological Station (Nanaimo, BC), in association with *Chaetoceros* spp. blooms at a concentration of 1000–5000 cells L⁻¹ (Kennedy et al. 1976). Similar losses due to *C. convolutus* at concentrations of 8000–32,000 cells L⁻¹ were observed in 1977 (Brett et al. 1978). Mortalities of caged salmon (species not indicated) occurred during March–November 1987, in the upper Strait of Georgia (HAEDAT [CA-87-005](#)). The event involved *C. convolutus* (10×10^4 cells L⁻¹) during September–October, and *C. convolutus* (8000 cells L⁻¹) in March.

The seasonal abundance of *C. convolutus* and *C. concavicornis* is reported in Quatsino Sound during 1999–2011 (Haigh and Esenkulova 2012) and at seven stations in the Strait of Georgia during 2015–2017 (Esenkulova and Pearsall 2018; Esenkulova et al. 2018). In Clayoquot Sound, *C. concavicornis* (1.5×10^4 cells L⁻¹) resulted in the mortality of farmed Atlantic salmon (*Salmo salar*) on March 16, 2010 (ICES 2011). A bloom of *C. concavicornis* (1.6×10^4 cells L⁻¹) caused mass mortalities of cultured finfish in Johnstone Strait in mid-July 2015 (ICES 2017). In October 2016, *C. convolutus* (4.7×10^5 cells L⁻¹) and *C. concavicornis* caused mass mortalities in the Esperansa Inlet (ICES 2017). The mortality of thousands of caged salmon was reported on November 15, 2019, caused by a bloom of *C. convolutus* and *C. concavicornis* near Tofino ([The Tyee new report](#); [Clayoquot Action report](#)).

Both *Chaetoceros* species are a normal component of the phytoplankton assemblage, occurring along the entire BC coast (Albright et al. 1992). Highest concentrations in surface waters are found in late spring and autumn, which is when fish kills are typically reported, usually in relatively stratified water with salinities of 26–34 (Albright et al. 1992) and temperatures of 9–10 °C (Cross and Dobbs 1988). During the summer, high concentrations of these taxa were found at the pycnocline, where

light and temperature were lower and nutrients were higher than at the surface (Haigh and Taylor 1990). Field and laboratory observations show that *C. concavicornis* and *C. convolutus* are excluded from waters with salinities <17 (Albright et al. 1992; Harrison et al. 1993); this is useful in helping fish farmers decide which inlets are less prone to losses from blooms of these species.

Both species have been reported in sea ice from the Canada Basin (Melnikov et al. 2002; Mather et al. 2010) and in phytoplankton from the five Canadian Arctic regions (Davidson 1931; Polunin 1934; Grøntved and Seidenfaden 1938; Seidenfaden 1947; Bursa 1961a,b, 1971; Sekerak et al. 1976a,b, 1979; MacLaren Atlantic Limited 1977, 1978; Anderson et al. 1981; Hsiao 1983, 1985; Hsiao and Pinkewycz 1984; Percy et al. 1992; Simard et al. 1996; Harvey et al. 1997; Lovejoy et al. 2002; Melnikov et al. 2002; McLaughlin et al. 2009; Mather et al. 2010; Crawford et al. 2018; M. Poulin unpubl. data) ([Appendix 1](#)).

There is still some disagreement about how the salmonid fish are killed. Both *Chaetoceros* species are characterized by barbed setae (as is *C. danicus*), and three studies showed that the barbs become embedded in the epithelium of fish gills (Bell 1961; Yang and Albright 1992, 1994b). However, this evidence is contradicted by the discovery that the barbs do not penetrate the gill tissue. Rather, diatom cells were shown to lodge between the secondary lamellae of the gills or in the overlying mucus (Rensel 1991, 1992, 1993). Either situation causes coughing, followed by an overproduction and accumulation of mucus. This results in a decrease in the surface area of the primary lamellae, which limits oxygen uptake, so that death could occur via blood hypoxia (Rensel 1993; Yang and Albright 1994a). A secondary effect, even at sub-lethal cell concentrations, is an apparent weakening of the immune system, resulting in increased mortality due, for example, to vibriosis or bacterial kidney disease (Albright et al. 1993). This is supported by the finding of suppression in the neutrophil, lymphocyte and thrombocyte components of the immune system of Chinook salmon (Yang and Albright 1994b). Experiments show that mortality may be reduced by oxygenation (not aeration) within net-pens (Rensel 1991), or by adding a mucolytic agent (e.g. L-cysteine ethyl ester, which suppresses mucus production) to the food of the caged fish when the diatom concentration exceeds critical levels (Yang and Albright 1994a). *Chaetoceros danicus* has been reported in the phytoplankton from the eastern Arctic and the Hudson Bay system (Bursa 1971; Anderson et al. 1981; Hsiao 1983; Percy et al. 1992) ([Appendix 1](#)).

On the Atlantic coast, both *Chaetoceros convolutus* and *C. concavicornis* are regularly observed in the St. Lawrence Estuary, Gulf of St. Lawrence (Roy et al. 1996; Bérard-Therriault et al. 1999), Bay of Fundy (Martin and Wildish 1990; Martin et al. 1995, 2009), Chaleur Bay (NB) (Brunel 1962), and in NL embayments (C.H. McKenzie unpubl. data). *Chaetoceros concavicornis* is present in St. Margarets Bay (NS) (Bugden et al. 1992). In each of these situations, the cell numbers have been too low to observe any effect on fish.

Chaetoceros gelidus (formerly *C. socialis*) can produce haemolytic substances that can cause gill damage or lesions by physical irritation to the epithelium, resulting in excessive mucus production, which consequently leads to asphyxiation (Burridge et al. 2010). However, bioassays exposing salmon smolts from the Bay of Fundy to 4.0×10^6 *C. gelidus* (as *C. socialis*) cells L⁻¹ for 24 h had no apparent deleterious effects. Although *C. gelidus* may affect gills of exposed Atlantic salmon (Landsberg 2002), there was no evidence of cells on gill filaments of the salmon studied by Burridge et al. (2010). The other harmful *Chaetoceros* species from that location should be tested. There is laboratory evidence, however, that Atlantic salmon (*Salmo salar*) respond with a rapid increase in

mucus discharge on the gills, and experience severe hypoxia and elevated blood carbon dioxide, when exposed short-term to *C. socialis* concentrations as low as 10 cells mL⁻¹ (Rensel 1995; text reproduced in Rensel 2007).

Chaetoceros gelidus (as *C. socialis*) has been reported in sea ice from the Canada Basin and the eastern Arctic (Melnikov et al. 2002; Simard 2003) and in the phytoplankton from the five Canadian Arctic regions (Polunin 1934; Grøntved and Seidenfaden 1938; Seidenfaden 1947; Bursa 1961a,b, 1971; Foy and Hsiao 1976; Hsiao 1976, 1983, 1985; Sekerak et al. 1976a,b, 1979; Bain et al. 1977; Hsiao et al. 1977; MacLaren Marex 1979a,b; Hsiao and Trucco 1980; Grainger and Hsiao 1982; Hsiao and Pinkewycz 1983, 1984; Percy et al. 1992; Simard et al. 1996; Harvey et al. 1997; von Quillfeldt 2000; Lovejoy et al. 2002; Melnikov et al. 2002; Riedel et al. 2003; McLaughlin et al. 2009; Mundy et al. 2011; Mather et al. 2010; Joo et al. 2012; Simo-Matchim et al. 2017; Crawford et al. 2018; Joli et al. 2018; M. Poulin unpubl. data) ([Appendix 1](#)).

Marine microalgal extracts obtained by sonicating cells from unialgal cultures or from plankton net hauls were tested for their effects on the heartbeat of Atlantic salmon (*Salmo salar*) smolts from the Bay of Fundy (Wildish et al. 1991). Extracts of *C. debilis* placed directly on the heart produced bradycardia, according to electrocardiograms, indicating that this is a potentially harmful diatom species. *Chaetoceros debilis* has been reported in the phytoplankton from all Canadian Arctic regions, except from the Canada Basin (Polunin 1934; Grøntved and Seidenfaden 1938; Seidenfaden 1947; Bursa 1961a,b, 1971; Thomson et al. 1975; Sekerak et al. 1976a, 1979; MacLaren Atlantic Limited 1977; Hsiao and Trucco 1980; Anderson et al. 1981; Hsiao 1983, 1985; Hsiao and Pinkewycz 1984, 1985b; Pinkewycz et al. 1987; Percy et al. 1992; Simard et al. 1996; Harvey et al. 1997; von Quillfeldt 2000; Lovejoy et al. 2002; McLaughlin et al. 2009; Mather et al. 2010; Simo-Matchim et al. 2017; Crawford et al. 2018; M. Poulin unpubl. data) ([Appendix 1](#)). However, it should be noted that extracts of the diatoms *Pseudo-nitzschia multiseriata*, *P. pseudodelicatissima*, and of the dinoflagellate *Alexandrium catenella* (as *A. fundyense*), caused similar results. No effects on heartbeat were found with extracts of the dinoflagellate *Scrippsiella trochoidea* and the diatom *Thalassiosira gravida* (Wildish et al. 1991).

The occurrence of *C. convolutus* is monitored in the Bay of Fundy because of the salmon aquaculture industry (e.g. Chang et al. 2006, 2007b; Martin et al. 2006a). The presence of *Chaetoceros* species in NL is important to monitor, as the development of a salmon aquaculture industry continues (C.H. McKenzie unpubl. data). Several salmon aquaculture sites have expressed concern for blooms of harmful algae, and there is interest in developing a harmful phytoplankton monitoring program similar to that in BC (Section [9.2.1](#)).

7.1.2 *Skeletonema costatum*, *Thalassiosira aestivalis*, *T. rotula*

In addition to the above *Chaetoceros* species, other diatoms have been associated with salmon mortalities in BC. A dense algal bloom started on April 4, 1994, at a netpen farm near Quadra Island, and lasted for 14 days (Kent et al. 1995). A dramatic increase in mortality of Atlantic salmon (*Salmo salar*) post-smolts at the farm was noted by the farmer during the first day of the bloom, and mortalities continued for one week. A loss of ~16,000 fish (4%) was attributed to the bloom. Many moribund fish exhibited excessive mucus secretion on the surface of the gills. The bloom consisted predominantly of the diatoms *S. costatum*, *T. aestivalis* and *T. rotula*. Although these diatoms do not possess spines on their setae, unlike the *Chaetoceros* species, Kent et al.

(1995) indicate that they do have long siliceous setae that may damage the gill surface in a similar way. However, it must be pointed out that the *Thalassiosira* processes are short and frequently have long organic threads extruded from them (Hoppenrath et al. 2009). Thus, the cause of future such mortalities must be investigated more thoroughly.

Skeletonema costatum has been reported in sea ice from the Hudson Bay system (Poulin et al. 1983) and in the phytoplankton from the five Canadian Arctic regions, except from the Canada Basin (Bursa 1961a; Foy and Hsiao 1976; Hsiao 1976, 1983; Hsiao et al. 1977; MacLaren Atlantic Limited 1977, 1978; Sekerak et al. 1979; Anderson et al. 1981; Percy et al. 1992; Simard et al. 1996; Harvey et al. 1997; Lovejoy et al. 2002; Mather et al. 2010; Simo-Matchim et al. 2017; M. Poulin unpubl. data). *Thalassiosira aestivalis* has been reported in sea ice from the eastern Arctic (Hsiao and Grainger 1982) and in the phytoplankton from the eastern Arctic and the Hudson Bay system (Bursa 1961b; MacLaren Atlantic Limited 1978; Hsiao and Grainger 1982; Hsiao 1983), whereas *T. rotula* has been only reported in the phytoplankton from the five Canadian Arctic regions, except from the Canada Basin (Polunin 1934; Seidenfaden 1947; Bursa 1961a,b, 1971; MacLaren Atlantic Limited 1977, 1978; Sekerak et al. 1979; Hsiao 1983; Hsiao et al. 1984; Mather et al. 2010) ([Appendix 1](#)).

7.1.3 *Corethron pennatum*

A previously healthy group of 100,000 coho salmon (*Oncorhynchus kisutch*) smolts suffered 60% mortality within one week of being transferred from fresh water into salt water netpens in October 1987, in coastal BC (Speare et al. 1989). Clinical symptoms included fish hanging listlessly near the top and sides of the netpen, and rapid, laboured respiration with constant opercular flaring. In addition, some fish had petechial haemorrhages mid-way along their fin rays. Subsequent investigation revealed that oxygen levels were normal, but Secchi disk readings indicated turbidity caused by a phytoplankton bloom during the first week. The bloom abated, but fish continued to die at a rate of ~10% per week through November and early December 1987. Speare et al. (1989) showed that extensive entrapment of diatoms in interlamellar spaces was associated with dramatic suppurative bronchitis and extensive fusion of gill lamellae. The fish were also infected with a microsporidian parasite (*Loma* sp.) that produced multifocal intracellular cysts in endothelial cells of the gill vasculature and in the endothelial-like pillar cells of the lamellae. This infection was believed to have occurred prior to the transfer of the fish to salt water. Mortality was considered most likely a response to the diatoms, compounded by the osmotic stress from transfer to salt water.

The above diatom resembled *Corethron* sp., a solitary cell with a circumferential corona of non-branching spiny setae. The setae were located at only one end of the cells, in contrast to the normal arrangement in which both valves have a circlet of spines directed toward the same pole (Hendey 1964). Because *Corethron* sp. is very plastic in response to variable environmental conditions, Hendey (1937) merged all known types into a single species, *C. pennatum*. It should be noted that Crawford et al. (1998) discarded the name *C. criophilum*, in favour of *C. pennatum*, which is currently regarded as a synonym ([AlgaeBase](#)). A second species that also occurs as single cells and is found in BC is *C. hystrix* (Stockner and Cliff 1975; DFO 2015; Haigh 2017). Hendey (1937) noted that in some waters (e.g. Antarctic, but not European), *C. pennatum* produces a central corona of short spines tipped with small claws. Although such a feature is not visible in the SEM of Speare et al. (1989), it is possible that spiny setae may contribute to salmon mortality, as is the case for some *Chaetoceros* species. Although *Corethron* sp. is oceanic and rarely found in great numbers along the

west coast (Cupp 1943; Forbes and Waters 1993a,b), *C. pennatum* dominated the early autumn diatom peak in Sechart Inlet during three years of observations (Haigh et al. 1992). *Corethron pennatum* has also been reported in the Bay of Fundy (Martin et al. 2009), and in ballast water ([Appendix 4](#)).

7.1.4 *Leptocylindrus minimus*

The chain-forming estuarine centric diatom *L. minimus* has been implicated in mortalities of cultured salmon and trout in southern Chile, but nowhere else in the world (Clément and Lembeye 1993; Clément 1994). Mechanisms of salmon mortality have not yet been elucidated. Because of the importance of salmon aquaculture in the Bay of Fundy, this diatom has been monitored there since the beginning of the phytoplankton monitoring program in 1987 (Martin et al. 1995, 2006a, 2009, 2010b, 2014b,c; Martin and LeGresley 2014). The highest concentration recorded during a 1987–2004 survey was 3.56×10^5 cells L⁻¹, at Brandy Cove (Passamaquoddy Bay) on July 31, 1990 (Martin et al. 2010b). This is still considerably lower than the $>1.0 \times 10^7$ cells L⁻¹ that caused the salmonid mortalities in Chile.

On the east coast, *L. minimus* has also been reported in Conception Bay (NL), at concentrations of 5×10^4 cells L⁻¹ (C.H. McKenzie unpubl. data), although there is no fish aquaculture there. Aquaculturists should nevertheless be aware of the presence of *L. minimus* because of the potential for development of salmon aquaculture in NL. This species was also reported near Rimouski (QC), in the Gulf of St. Lawrence, in 2002 (Starr et al. 2003), as well as from all sectors of the Estuary and Gulf of St. Lawrence (Bérard-Therriault et al. 1999).

This diatom is also common in BC coastal waters. Forbes and Waters (1993a,b) reported a maximum concentration of 2700 cells L⁻¹ in Barkley Sound (Vancouver Island) in June 1989. Samples from salmon aquaculture sites on the west coast of Vancouver Island frequently contain *L. minimus* at $>1 \times 10^6$ cells L⁻¹ in late August and September, with a maximum reported concentration of 4.6×10^7 cells L⁻¹ in northern Clayoquot Sound in August 2001. No mortalities have been recorded associated with these blooms (Haigh et al. 2004a,b,c,d,e).

Leptocylindrus minimus has been reported in sea ice from the Canada Basin (Melnikov et al. 2002) and in the phytoplankton from the five Canadian Arctic regions (Polunin 1934; Seidenfaden 1947; Bursa 1961a; Sekerak et al. 1976b; MacLaren Atlantic Limited 1978; Hsiao 1983; Anderson et al. 1981; Simard et al. 1996; Harvey et al. 1997; Melnikov et al. 2002; Simo-Matchim et al. 2017; M. Poulin unpubl. data) ([Appendix 1](#)).

7.1.5 *Ditylum brightwellii*

This cosmopolitan diatom has not been documented to have caused problems with aquaculture operations anywhere in the world. However, high concentrations of *D. brightwellii* (1.9×10^5 cells L⁻¹) were observed during an *Alexandrium catenella* (as *A. fundyense*) bloom that was associated with salmon mortalities near Grand Manan Island (Bay of Fundy) in September 2003 (Martin et al. 2006c, 2008a, 2009). As a potentially harmful diatom, it was therefore included in the Bay of Fundy phytoplankton monitoring program (Section [9.2.2.3](#)). Preliminary experiments exposing salmon to high concentrations of *D. brightwellii* (1.0×10^6 cells L⁻¹) for 24 h did not result in any fish mortalities (Burridge et al. 2010), although further research is warranted. It has been reported in the

phytoplankton from the eastern Arctic (Hsiao and Pinkewycz 1985a; Lovejoy et al. 2002; Mather et al. 2010) ([Appendix 1](#)).

7.1.6 *Eucampia zodiacus*

Eucampia zodiacus is another cosmopolitan diatom endemic to the Bay of Fundy since at least the early 1930s (Martin et al. 2007a, 2009). An extended *E. zodiacus* bloom (9.6×10^5 cells L⁻¹) was associated with low levels of salmon mortalities in the Bay of Fundy in 2002 (ICES 2003). However, there was no mortality in laboratory experiments when salmon smolts were exposed to 9.0×10^5 cells L⁻¹ for 24 h (Burridge et al. 2010).

7.2 Other dinoflagellates

7.2.1 *Alexandrium pseudogonyaulax*

Another toxic species of the dinoflagellate genus *Alexandrium*, *A. pseudogonyaulax* (previously known as *Goniodoma pseudogonyaulax*), was observed in the Strait of Georgia (BC) in 1991 (Taylor and Haigh 1993), in the Bay of Fundy since 2001 (Martin and LeGresley 2008; Martin et al. 2009; Klein et al. 2010), and in the Gulf of St. Lawrence, also since 2001 (Dufour et al. 2010). After its first occurrence at Havre-aux-Maisons and Tête-à-la-Baleine (QC), *A. pseudogonyaulax* has been detected regularly at the former location on the Magdalen Islands, but only one subsequent time at Tête-à-la-Baleine, in 2003 (Lessard et al. 2020). It was observed once at Sept-Îles, in 2004, and at Mont-Louis, in 2008, at the detection level (20 cells L⁻¹).

This species produces the phycotoxin goniodomin A, a polyether macrolide that has antifungal properties (Murakami et al. 1988). It also targets the liver and thymus in mice (Terao et al. 1989) and affects muscle contraction in rabbits by modulating the actomyosin system (Furukawa et al. 1993). Goniodomin A has also been isolated from *Alexandrium monilatum*, from the northern Gulf of Mexico (Hsia et al. 2006). This species has been responsible for fish kills in the Gulf of Mexico (references in Hsia et al. 2006) and was associated with whelk mortality in the U.S. (Harding et al. 2009). Goniodomin A has thus far not been implicated in any human health issues. However, the bioactive properties of this phycotoxin and the presence of *A. pseudogonyaulax* in Canadian waters make it a topic of interest. *Alexandrium monilatum* has been reported in sea ice from the Hudson Bay system (Hsiao et al. 1984) and in the phytoplankton from the eastern Arctic and the Hudson Bay system (Hsiao and Pinkewycz 1983, 1984, 1985a,b; Hsiao et al. 1984; Pinkewycz et al. 1987) ([Appendix 1](#)).

7.2.2 *Gyrodinium*, *Karenia*

The identification of athecate (i.e. without thecal plates) dinoflagellates is difficult, and their taxonomy has also undergone considerable change. Thus, the names used below are those reported in the original publications. Dinoflagellates of the genera *Gyrodinium* and *Gymnodinium* are difficult to distinguish from each other, leading to taxonomic confusion (Taylor 1985).

A *Gyrodinium* dinoflagellate species was observed in the southeastern Gulf of St. Lawrence (Bugden et al. 1992; J.C. Smith, DFO, Moncton, NB, 1994 pers. comm.) and along the Pacific coast (Forbes and Waters 1993a,b), where *Gyrodinium* cf. *aureolum* (also referred to as *Gymnodinium* cf.

aureolum) has also been recorded. A dense bloom of *Gyrodinium aureolum* (6×10^5 cells L⁻¹) was reported for the first time in the Gaspé current of the Gulf of St. Lawrence in September 1993 (Blasco et al. 1994). An immunological probe indicated that the organism was phenotypically identical to the species found in northern European waters. It was also found at low concentrations (up to 3.8×10^4 cells L⁻¹) in the Bay of Fundy (Martin and Wildish 1990; Martin et al. 1995), and it has been reported in the phytoplankton from the eastern Arctic (MacLaren Atlantic Limited 1978; Hsiao 1983; Lovejoy et al. 2002) ([Appendix 1](#)). *Gyrodinium galatheanum* was isolated near Lummi Island (WA), adjacent to Canadian waters, in 1992. This species has caused fish kills in Norway, and the isolate proved toxic to sticklebacks (Taylor and Horner 1994).

During a comprehensive study of harmful algal species in inlets of PE during late summer to fall of 2001–2003, the athecate dinoflagellate *Karenia mikimotoi* (synonyms: *Gymnodinium mikimotoi*, *Gyrodinium nagasakiense* and *Gymnodinium nagasakiense*) was identified (Bates and Strain 2006). It was the fourth most abundant phytoplankter, but found in substantial numbers only in the Cardigan River, and only in 2001 and 2003. In 2001, peak numbers reached 1.5×10^6 cells L⁻¹ on October 31, and in 2003, the peak was at 1.3×10^6 cells L⁻¹ on October 20. In spite of these high concentrations, no harmful effects were observed. *Karenia mikimotoi* was reported from the south and north shores of the Gulf of St. Lawrence during 1994–2008, from May 1 to October 31, at a maximum concentration of 2.6×10^5 cells L⁻¹ (Lessard et al. 2020). This species was also identified at salmon aquaculture sites in Quatsino Sound, on the northwest coast of Vancouver Island, BC, but without any fish mortalities (Haigh and Esenkulova 2012; Haigh 2017). It has been also reported in the phytoplankton from the western Arctic (McLaughlin et al. 2009) ([Appendix 1](#)).

The presence of *K. mikimotoi* should be considered as a potential threat to benthic and pelagic fauna, including molluscan shellfish used in the aquaculture industry. It has caused mortalities of wild and farmed fish in European coastal waters. For example, there was an 80% mortality of farmed clams (*Tapes semidecussata*) during a 1992 bloom of *K. mikimotoi* in Ireland (O’Boyle et al. 2001). An even larger bloom in 2005 resulted in mass mortalities of cultured oysters (spat and adult), clams, scallops and abalone, as well as other benthic fauna (Silke et al. 2005). In 1995, a *K. mikimotoi* bloom caused the mortality of 800–900 tonnes of blue mussels (*M. edulis*) along the French Atlantic coast (Gentien 1998). Other blooms of *K. mikimotoi* in France caused a mass mortality of post-larval stages of the king scallop (*Pecten maximus*) and affected the growth and reproduction of adult scallops (Erard-Le Denn et al. 1990). It has also resulted in the mortalities of fish and benthic fauna in Scottish waters (Smayda 2006). Cytotoxic polyethers have been extracted from cultures of *K. mikimotoi* (Silke et al. 2005) and it generates reactive oxygen species (Yamasaki et al. 2004; Diaz and Plummer 2018; Diaz et al. 2018). Fish mortality may also result from hypoxia during a *K. mikimotoi* bloom (reviewed in Silke et al. 2005; Smayda 2006).

Specific taxonomic problems are related to these particular taxa (Taylor 1985; Partensky et al. 1988; Matsuoka et al. 1989). While phenotypically identical or very similar, there appear to be physiological and genetic differences. Blooms of *Gyrodinium aureolum* (in some cases identified as *Gymnodinium nagasakiense*) have not been associated with human illness. However, at high cell numbers, *G. aureolum* has killed farmed and wild Atlantic salmon and rainbow trout in Norway, Scotland and Ireland, and quinnat (Chinook) salmon in New Zealand (Tangen 1977; Dahl and Tangen 1993; Rhodes et al. 1993) and, from 1980 to 1990, caused losses of ¥6.7 billion (~CAN\$77 million) to the fish farm industry in Japan (Honjo 1994). *Gyrodinium aureolum* also represents a potential threat to larval oysters, mussels, sea scallops and other benthic invertebrates (Shumway 1990; Lesser

and Shumway 1993; Smolowitz and Shumway 1997). A massive shellfish kill was associated with the presence of *G. aureolum* just south of Atlantic Canada, in Maine, U.S. (Heinig and Campbell 1992), and similar kills have been observed elsewhere, including Ireland (Ottway et al. 1979). The chemical identity of the toxin is still being debated, but *G. aureolum* negatively affects marine organisms by consuming oxygen, stimulating mucus production, and producing a weak toxin(s) having haemolytic, neurotoxic, and cytotoxic components (Turner et al. 1987; Partensky et al. 1989).

7.2.3 *Akashiwo sanguinea*

Although not toxic to humans, the HAB-forming unarmoured dinoflagellate *A. sanguinea* (previously called *Gymnodinium sanguineum* and *G. splendens*) has been implicated in the mortality of Olympia oyster (*Ostrea lurida*) larvae in WA (Shumway 1990) and juvenile eastern oysters (*Crassostrea virginica*) in Long Island Sound, New York (Bricelj et al. 1992). *Akashiwo sanguinea* is capable of producing large quantities of mycosporine-like amino acids (MAAs), which are water-soluble and serve as powerful surfactants. This property resulted in the first documented case of marine bird mortalities (mostly western grebes and northern fulmars), in Monterey Bay, CA, in November–December 2007 (Jessup et al. 2009). A “red tide” of *A. sanguinea* occurred coincident with the mortalities, and senescent cultures of *A. sanguinea* produced the same surfactant as found in the seawater.

Akashiwo sanguinea is a component of the BC coastal flora, although uncommon and normally at low concentrations in offshore waters (Forbes and Waters 1993a,b). However, it frequently blooms in the inlets on the west coast of Vancouver Island (Haigh et al. 2004a,b,c,d,e), and is responsible for recurring blooms in Esquimalt Lagoon, on the southern tip of Vancouver Island (Watanabe and Robinson 1979; Robinson and Brown 1983), where it forms as yet unidentified overwintering cysts (VOLTOLINA 1993).

Although the growth rate of *A. sanguinea* is low, it grows well, even under mild turbulence and under seasonally varying water temperatures (Menden-Deuer and Montalbano 2015). It is therefore able to persist over long periods, thus permitting a very slow accumulation to form observable blooms. Given the ubiquitous nature of *A. sanguinea* HABs and their threat to fish and the shellfish industry, Tang and Gobler (2015) studied the mechanisms regulating its distribution and bloom formation. They provided the first convincing evidence that *A. sanguinea* produces sexual resting cysts. This may be one of the key mechanisms that accounts for its ubiquitous distribution, expansion and frequent recurrence around the world, including its possible transport in ballast water (Smayda 2007).

7.2.4 *Margalefidinium polykrikoides*, *M. fulvescens*

The athecate (unarmoured) dinoflagellate *M. polykrikoides* (previously known as *Cochlodinium polykrikoides*; Gómez et al. 2017) is well known for causing fish kills worldwide, but especially in Korea, Japan and China (reviewed in Kudela et al. 2008), and is associated with the mortality of North Atlantic bivalve molluscs (Griffith et al. 2019a). Cells are oval and slightly flattened dorso-ventrally, 30–40 µm long, 20–30 µm wide, and they form short chains. A new species, *M. fulvescens*, was distinguished from *M. polykrikoides* by several morphological characteristics, i.e. cell size, shape of chloroplasts and the position of a narrow sulcus situated on the cell surface (Iwataki et al. 2007). Molecular analysis later confirmed these two species were genetically distinct (Iwataki et al. 2008). Molecular phylogenetic studies have since showed that isolates of *M. polykrikoides* (as

Cochlodinium polykrikoides) and *M. fulvescens* (as *Cochlodinium fulvescens*) (including one from Coal Harbour, BC, near Quatsino, northern Vancouver Island) differed significantly from the type species *C. strangulatum* (Gómez et al. 2017). These two species, and all other similar photosynthetic species with an eyespot in the episome and an anterior nucleus that were previously in the genus *Cochlodinium*, were therefore placed in the new genus *Margalefidinium*. The presence of *M. fulvescens* (as *Cochlodinium fulvescens*) was documented during 2015–2017, in the Strait of Georgia, BC (Haigh 2007; Esenkulova and Pearsall 2018; Esenkulova et al. 2018).

Blooms of cells that matched the original description of *M. polykrikoides*, but with chains comprised of two cells, were reported on the west coast of Vancouver Island, from August to October 1999, when the BC [HAMP](#) first started operating (Whyte et al. 2001a,b). The bloom in Quatsino Sound then migrated eastwards and killed farmed salmon (*Salmo salar*) in Holberg Inlet; this is the first report of a fish kill caused by *Margalefidinium* on the Canadian west coast, and it resulted in a loss of about CAN\$2 million. The highest cell concentrations (up to 60,000 cells mL⁻¹ at the surface) were in early September in Kyuquot and Quatsino Sounds; no blooms persisted after October 8. Based on morphological and phylogenetic analysis of three isolates from the Pacific (Japan, CA [U.S.], Coal Harbour [BC]), Iwataki et al. (2008) conclude that the species present in these areas is *M. fulvescens*.

Bioassays carried out in the field and laboratory confirmed that *Margalefidinium* (as *Cochlodinium*) killed the salmon in BC (Whyte et al. 2001a,b). Cultured juvenile smolted Atlantic salmon were confined in a flow-through container and exposed to a natural bloom of *Margalefidinium* sp. in Quatsino Sound, in September 1999. As the bloom passed through the test site, the cell concentration decreased from 10,800 to 2700 cells mL⁻¹, after 500 min. Total mortality occurred after 480 min. In the laboratory experiments, *Salmo salar* smolts died within 27 min of exposure to 7200 cells mL⁻¹ and 55 min with 3400 cells mL⁻¹. Juvenile coho salmon (*Oncorhynchus kisutch*) exposed to different concentrations of *Margalefidinium* sp. (as *Cochlodinium* sp.) did not survive beyond 212 min. Interestingly, the time of death of both salmon species was accelerated in both the field and laboratory bioassays by oxygenation or aeration of the *Margalefidinium* sp. The dead fish exhibited no external pathogenic abnormalities, and no excess mucus production was evident in the gill region. However, histological examination of gill samples showed oedema and separation of the lamellar epithelium, with clubbing and curling of the epithelial filaments (Whyte et al. 2001b).

Margalefidinium fulvescens (as *Cochlodinium fulvescens*) was quantified in Quatsino Sound, from February to November, during 1999 to 2011 (Haigh and Esenkulova 2012). Although cells of the *Margalefidinium* genus may burst or become deformed during preservation with formalin, they are not difficult to identify in Lugol's iodine-preserved samples using the light microscope. Perceived difficulties in identification and enumeration, however, prompted Howard et al. (2012) to develop a successful qPCR assay to quantify cell numbers of *M. fulvescens* in Monterey Bay (CA). This could also be used for *M. fulvescens* in BC waters.

On the Canadian east coast, *Margalefidinium* sp. is reported from the Bay of Fundy (Van Guelpen and Pohle 2019), but has otherwise not been studied. On the U.S. east coast, isolates and natural blooms of *M. polykrikoides* caused the mortality of eastern oysters (*Crassostrea virginica*), bay scallops (*Argopecten irradians*), bay quahaugs (*Mercenaria mercenaria*) (Tang and Gobler 2009; Griffith et al. 2019a), Atlantic silverside (*Menidia menidia*) at different life stages, inland silverside

(*M. beryllina*), sheepshead minnow (*Cyprinodon variegatus*), and mummichog (*Fundulus heteroclitus*) (Gobler et al. 2008; Rountos et al. 2014).

Understanding the mechanism of toxicity of *Margalefidinium* to fish has been challenging because no signs of pathology are obvious. It may involve a combination of unrelated factors, including: 1) production of ichthyotoxic substances (neurotoxic, hemolytic and hemagglutinating fractions, and STX group toxins); 2) production of reactive oxygen species; and 3) production of extracellular mucoid polysaccharide substances (Richlen et al. 2010; Diaz and Plummer 2018).

7.2.5 *Coolia monotis*

Benthic dinoflagellates of the genus *Coolia* are globally distributed in temperate coastal and estuarine waters, but most species are found in tropical regions (Leaw et al. 2016). Starting in August 2012, *Coolia* spp. were observed in the Bras d'Or Lakes of Cape Breton Island, NS (at Johnston Harbour, Ben Eoin, Whycocomagh Bay, Boisdale, and Grand Narrows), as well as on the northwestern tip of Cape Breton Island (Point Edward), and on the eastern shore of NS (West Jeddore) (Lewis et al. 2018). An isolate, identified as *C. monotis* by molecular and morphometric means, was isolated from Johnston Harbour, when the water temperature was 12 °C, and exponential growth was observed from 15–25 °C. This is the first record of this species in NS, although it has been reported in sea ice and in the plankton from the Hudson Bay system (Hsiao et al. 1984) ([Appendix 1](#)). Toxicity, however, was not observed in two zebrafish bioassays (Lewis et al. 2018). Likewise, no sulphated analogues with similarities to YTXs (see references in Lewis et al. 2018) were detected by LC–MS. Nevertheless, this genus is considered harmful because it produces mucilage aggregates or biofilms that can degrade water quality, accumulate and disperse other harmful microorganisms, and cause benthic fauna mortalities (Lewis et al. 2018).

7.3 Other toxic or harmful phytoplankton

7.3.1 Raphidophyte flagellate *Heterosigma akashiwo*

Heterosigma akashiwo (previously also referred to as *Heterosigma carterae* and misidentified as *Olisthodiscus luteus*; Engesmo et al. 2016) is a chloromonad flagellate (Raphidophyceae). Cells are 10–25 µm long and 8–15 µm wide and “potato”-shaped, varying from ovoid to oblong. They are characterized by numerous golden chloroplasts and two clearly distinct flagella: a hairy flagellum directed forward and a smooth flagellum that trails the cell (Engesmo et al. 2016; Haigh 2017). Cells are fragile, having a thin membrane covering, and may disintegrate if certain preservatives are used (e.g. formalin-acetic acid, glutaraldehyde). However, Lugol's iodine preservative allows the cells to remain mostly intact, and has been successfully used in BC for over 20 years to identify this species morphologically in routine sample analysis, using the light microscope (N. Haigh unpubl. data). Since the inception of salmonid aquaculture in BC in the 1970s, this species has been responsible for the greatest number of farmed salmon mortalities due to harmful algae, particularly within the Strait of Georgia (Haigh and Taylor 1990; Taylor and Haigh 1993), Broughton Island (Whyte et al. 1999), Barkley Sound (Taylor 1993; Taylor and Haigh 1993), Quatsino Sound (Haigh and Esenkulova 2012), Kyuquot Sound, and Sechart Inlet (Taylor et al. 1991, 1994) ([Fig. 1](#)). It was also present at low concentrations on the continental shelf off Vancouver Island (Forbes 1993). Although this organism is found worldwide, including the Russian Arctic (Engesmo et al. 2016) and the Canadian Arctic (Poulin et al. 2011), it has never been reported on the east coast of Canada.

In the Strait of Georgia (which is now considered part of the Salish Sea that also includes the U.S. waters of the Strait of Juan de Fuca and Puget Sound), *H. akashiwo* blooms are generally seeded from shallow areas in the vicinity of the Fraser River plume, and advected via circulation in a roughly counter-clockwise direction around the Strait. Freshwater input from the Strait appears to be important both in causing the necessary stratification and in providing a source of micronutrients (Taylor and Haigh 1993). This agrees with evidence of an enhanced requirement for iron, manganese and vitamin B₁₂ for growth (Honjo 1993). Blooms are typically initiated when water temperature rises above 15 °C and salinity drops to below 15. However, in experimental conditions *H. akashiwo* demonstrated tolerance or acclimation to a wide range of temperatures (15–30 °C) and salinities (2–50) (Honjo 1993; Smayda 2006; Martínez et al. 2010; Butrón et al. 2012). The ability to grow at these wide ranges may be attributed in part to physiological and genetic interstrain variability (Fredrickson et al. 2011), which may also account for its ecological success in different environments.

Although not associated with human illness, blooms of *H. akashiwo* that can turn the water brown have killed fish in many other regions of the world, including Chile, Bermuda, New Zealand, Japan, China, Korea, Malaysia, Spain, Scotland, and waters of WA (U.S.) (Chang et al. 1990; Clément and Lembeye 1993; Honjo 1993; Smayda 1998, 2006; Whyte et al. 1999; Rensel and Whyte 2003; Rensel 2007; Rensel et al. 2010). In BC ([Fig. 1A](#)), the first netpen-reared mortalities were reported at Nanoose Bay, on the west coast of Vancouver Island, in 1976 (Gaines and Taylor 1986). This organism then resulted in the mortality of ~30% of coho salmon (*Oncorhynchus kisutch*) and Chinook salmon (*Oncorhynchus tshawytscha*) in Sechelt Inlet, in 1986, resulting in a loss of US\$2.5 million in revenue (reported in Lewitus et al. 2012). Subsequent losses were CAN\$4 million in both 1989 and 1990 (Black 1991; Taylor 1993), and a staggering CAN\$10–20 million from July to October, 1997 alone (Whyte et al. 1999). In adjacent Puget Sound, WA, *H. akashiwo* blooms have resulted in an estimated US\$4–5 million annual loss to fish aquaculturists (Horner et al. 1991). Additional losses of revenue result from the accompanying downturn in the activity of the aquaculture sector.

Caged salmon are killed when *H. akashiwo* concentrations exceed several millions of cells per litre (Taylor and Haigh 1993), although cell concentration is often not correlated with mortality rates (e.g. Kennedy and Kreiberg 1999). A detailed description of Atlantic and Chinook salmon mortalities at the DFO Experimental Mariculture Facility of the Pacific Biological Station (Departure Bay), during 1993 and 1997, is given in Kennedy and Kreiberg (1999).

In 2010, Atlantic salmon mortalities occurred as a result of *H. akashiwo* at BC aquaculture operations at: Quatsino Sound (2×10^6 cells L⁻¹; March 16); Bedwell Sound, Clayoquot Sound (39×10^6 cells L⁻¹; July 15); Klemtu, Finlayson Sound (26×10^6 cells L⁻¹; September 3); Clayoquot Sound (6×10^6 cells L⁻¹; mid-September); Sechelt Inlet (25×10^6 cells L⁻¹; September 30) (ICES 2011). *Heterosigma* was also responsible for salmon mortalities in 2011, but at far lower cell concentrations: in the Upper Strait of Georgia (4×10^5 cells L⁻¹; June 23); Clayoquot Sound (3×10^5 cells L⁻¹; September 12) (ICES 2012). Interestingly, a cell concentration comparable to that found in 2010 (22×10^6 cells L⁻¹) had no ill effects on the farmed salmon at Jervis Island, on the Lower Strait of Georgia (ICES 2012), which suggests different potencies of *H. akashiwo* (see below).

In late September 2014, Marine Harvest Canada reported an estimated loss of up to 280,000 fish (4.1 kg average weight) due to a bloom of *H. akashiwo* at Marsh Bay, near Port Hardy, BC ([Marine Harvest news release](#)). The concentration of *H. akashiwo* reached 40×10^6 cells L⁻¹ on

September 2, prior to the mortalities (ICES 2017). This bloom was preceded by increased water temperatures and three months of sunny days. The normal mitigation systems, which include aeration, not feeding fish during blooms, pumping water from deeper depth, and encircling the pens with a plastic curtain to impede water movement (Kent 1992), were unable to overcome this extreme bloom.

In late June 2015, mass mortalities of fish occurred at Clayoquot Sound, when a *H. akashiwo* bloom lasted three weeks (ICES 2017). Then, another bloom during September to October lasted four weeks. In the Broughton Archipelago, a bloom of *H. akashiwo* started in mid-July and lasted six weeks. Sechelt Inlet experienced a *H. akashiwo* bloom in October that lasted two weeks (ICES 2017).

Two salmon farms in Jervis Inlet lost ~250,000 fish (~1000 tonnes), about half of the farmed fish, because of blooms of *H. akashiwo* during the first week of June 2018 (Robinson 2018). The alga was at an “extraordinarily high concentration” and was spread throughout the water, so that protective measures (e.g. aeration) could not prevent the extensive kill. Because the fish were not toxic, they could be sent to composting and rendering companies.

Wild salmon are also vulnerable to blooms of *H. akashiwo*, and this topic has been understudied. As part of the five-year multi-partnership [Salish Sea Marine Survival Project](#) to understand the causes of wild salmon mortality, a pilot study was initiated to monitor HABs and investigate their possible effects on juvenile salmon (wild and hatchery) in the Cowichan Bay estuary and outer bay, on the southeast coast of Vancouver Island (Esenkulova et al. 2015; Esenkulova and Pearsall 2018). A *H. akashiwo* bloom occurred during that study. Cells first appeared on May 26, 2014, in very low numbers (<10 cells mL^{-1}) in offshore waters. The bloom reached maximum concentrations of 4×10^4 cells mL^{-1} near shore (June 11) and 1.2×10^4 cells mL^{-1} offshore (June 16). High concentrations of *H. akashiwo* at open water sites were associated with lethargic swimming behavior of chum, Chinook, coho, and sockeye salmon, although Chinook salmon appeared to be the most affected. The north side of Cowichan Bay appeared to be a “hot spot”, with consistently higher concentrations of *H. akashiwo* than other areas. It will thus continue to be monitored.

There is no evidence that aquaculture activities in BC have promoted *H. akashiwo* blooms. In support of this, it has been observed that some of the most frequent and intense blooms in BC occur in the southern Strait of Georgia, where there are no commercial fish farms (Rensel et al. 2010). Rather, the Fraser River peak discharge during the spring/early summer is believed to have a greater influence by creating ideal growth conditions for this alga in that area (Lewitus et al. 2012). As well, *H. akashiwo* cysts first appeared in large numbers in Saanich Inlet sediments just prior to 9000 years ago (Mudie et al. 2002). It has been also reported in the phytoplankton from the eastern and the western Arctic, and the Canadian Archipelago (Riedel et al. 2003; McLaughlin et al. 2009; Simo-Matchim et al. 2017; M. Poulin unpubl. data) ([Appendix 1](#)).

From the opposite point of view, the Cohen Commission Report (Cohen 2012b) pointed out HABs, specifically those caused by *H. akashiwo*, as one of the numerous cumulative stressors that contributed to the 1992–2009 decline in the Fraser River sockeye salmon (*Oncorhynchus nerka*), historically the most valuable salmon fishery on the west coast. This conclusion was based in part on evidence, provided by Rensel et al. (2010), showing that the marine survival of the Chilko River salmon stock averaged 2.7% in years when the seawater migration of juvenile sockeye salmon in the

Strait of Georgia coincided with major *Heterosigma* blooms, compared to 10.9% in years with no or only minor blooms. It should be noted, however, that mortalities of sockeye salmon have not been directly attributed to *Heterosigma*, although there is no reason to preclude this possibility (Peterman et al. 2010). A further link between *Heterosigma* blooms and young-of-the-year Pacific herring (*Clupea pallasii*) abundance is given in Lewitus et al. (2012). Nevertheless, Dr. Rensel's testimony during the Commission (Cohen 2012b) suggested that "exposure of juvenile Fraser River sockeye to *Heterosigma* blooms could result in direct, acute effects or in chronic effects such as infections, making the fish more susceptible to poor food supply conditions and predation". Thus, *Heterosigma* blooms could have an impact on salmon when combined with other stressors, e.g. diseases and contaminants, making them more susceptible to predation and low food availability.

Recommendation 65 of the Commission Report (Cohen 2012c) was that "the Department of Fisheries and Oceans should undertake or commission research, in collaboration with academic researchers...[to] examine ... biological, chemical, and physical oceanographic variables, including water temperature, the presence or absence of harmful algal blooms..." (p. 59–60). It further pointed out that "because DFO is no longer involved in the harmful algae monitoring program (HAMP)... and is not doing any research or monitoring in this area, pertinent information and advice about harmful algal blooms might not be available to DFO fisheries managers or scientists. To the extent that DFO requires this information for the management and control of the fishery, it could work with the salmon-farming industry and HAMP as well as with non-DFO scientists to obtain it" (p. 99). The HAMP is discussed in Section [9.2.1](#).

Despite over 35 years of research, there is still controversy surrounding the mode of action of *Heterosigma* blooms that lead to fish kills, and this remains mostly unresolved. Moribund fish do not exhibit distinctive pathological changes (Kent 1992). Several toxicity mechanisms have been proposed, e.g. production of extracellular reactive oxygen species (such as hydrogen peroxide), intracellular brevetoxin-like compounds, or excessive mucus secretion (Cochlan et al. 2014; Diaz and Plummer 2018). However, none of these mechanisms has been supported by sufficient evidence.

The cause of death of salmon in Scotland exposed to "flagellate X", which may be *H. akashiwo*, was attributed to cardiac arrest (Gowen et al. 1982). Two of the first blooms of *H. akashiwo* to be studied in BC were at San Mateo Bay, in Barkley Sound (95% *H. akashiwo*; 200×10^6 cells L⁻¹) and at Blind Bay, in the Strait of Georgia (>95% *H. akashiwo*; 793×10^6 cells L⁻¹), during the summer of 1990 (Whyte 1994); note the order of magnitude higher cell concentrations compared to the blooms mentioned above. Experiments using water samples from these blooms showed that rates of mortality of juvenile Chinook salmon (*Oncorhynchus tshawytscha*) were independent of acclimation of the fish to seawater or ambient oxygen levels, but were dependent on concentration of the algae and ambient water temperature. These results, combined with the apparent lack of pathological abnormality to gills or other internal organs in the fish and lack of impact of aeration or oxygenation on mortality, suggested that the cause of death was due to a labile ichthyotoxic agent (Black et al. 1991; Whyte 1994). Aeration or oxygenation of fish cages to dilute or disperse the *H. akashiwo* bloom did not enhance or inhibit fish survival (Black et al. 1991).

Other experimental evidence suggested that the fish are killed when *H. akashiwo* produces superoxide radicals and hydrogen peroxide (Yang et al. 1995). Production of such reactive oxygen species by *H. akashiwo* was later confirmed (Twiner and Trick 2000). However, the production of

free radicals as a mechanism of mortality is only partially consistent with the findings of Black et al. (1991) and Whyte (1994), i.e. the lack of tissue breakdown, the absence of increased mortality when additional oxygen was supplied, and the absence of an attempt by the fish to avoid exposure to the material. In addition, it would be difficult to generate the required elevated levels of free radicals in the organic-rich environment of a bloom. This was confirmed when Twiner et al. (2001) concluded that the concentration of hydrogen peroxide produced by *H. akashiwo* is orders of magnitude less than that required to kill two vertebrate cell lines or the invertebrate brine shrimp (*Artemia salina*). It should be noted that no salmon were included in these tests.

Aside from reactive oxygen species, other modes of toxicity have also been proposed and investigated in laboratory settings, including production of brevetoxin-like compounds (Khan et al. 1997) and mucus or lectin-like polysaccharides (Chang et al. 1990). The reports on brevetoxin-like components are based on HPLC analysis, and not sodium channel inhibition, so this toxin was ruled out. However, an organic compound with a molecular weight similar to that of brevetoxin does dramatically alter calcium homeostasis, indicating the substance as a unique bioactive metabolite (Twiner et al. 2004, 2005). These compound(s) may play a significant role in the ichthyotoxic and allelopathic behaviour of *H. akashiwo*, yet there is still no molecular structure of the organics, nor any consensus about what the mortality mechanism is. Differences in toxicity of various natural blooms (see above cell numbers) and laboratory isolates (Fredrickson et al. 2011; Cochlan et al. 2014) complicate investigations. Other studies have shown that aqueous extracts of *H. akashiwo* cells decreased viability of the rainbow trout cell line RTgill-W1, derived from gill filaments (Cochlan et al. 2014). Furthermore, toxicity was greatest during the stationary phase of *H. akashiwo* cell growth, following the depletion of external nitrate. Until modes of toxicity are determined with more certainty, it will be difficult to implement mitigation strategies.

Heterosigma akashiwo has been shown to have harmful effects on organisms other than fish. For example, Whyte et al. (1999) reported that it was responsible for the mortality of oyster spat in BC. In Japan, it has been shown to have allelopathic effects against the harmful dinoflagellate *Akashiwo sanguinea* (present in Canadian waters; Section 7.2.3) (Qiu et al. 2012). It can also be toxic to some, but not all, ciliates (Clough and Strom 2005; Fredrickson et al. 2011).

Slow progress is being made on how viruses may affect the bloom dynamics of HAB species. In the case of *H. akashiwo* isolated from English Bay (Vancouver, BC), three viruses were shown to impair its photosynthetic activity prior to lysis (Juneau et al. 2003). This occurred even in darkness, showing that viral production can take place below the photic zone.

Early research on the fish-killing *H. akashiwo* was conducted by DFO and University of BC scientists prior to their retirements in the 1990s and early 2000s. Collaborative research on this species is now being carried out with U.S. scientists in Puget Sound, WA, by a Canadian scientist (Dr. Charles Trick; University of Western Ontario) ([NOAA news release](#)). This is also reflected in Cochlan et al. (2014). The presence of *H. akashiwo* in the Canadian Arctic (Riedel et al. 2003; McLaughlin et al. 2009; Poulin et al. 2011; Simo-Matchim et al. 2017) must also be noted.

Another ichthyotoxic raphidophyte species, *Chattonella* cf. *marina*, was responsible for the mortality of aquacultured fish in Esperanza Inlet, on the upper west coast of Vancouver Island, in September 2002 (Haigh 2017). Mortality occurred when the *Chattonella* concentration was 5.0×10^4 cells L⁻¹.

7.3.2 Prymnesiophyte *Chrysochromulina* spp.

The prymnesiophycean flagellate *C. birgeri*, originally identified during a bloom under the ice on the southern coast of Finland (Hällfors and Niemi 1974), was associated with a massive kill of farmed Atlantic salmon in March 1996, in the brackish waters of the Bras d'Or Lakes, on Cape Breton Island, NS (C. Carver, Mallet Research Services, Dartmouth, NS, 1996 pers. comm.). This was near the same location as the fish kills in March 1994, which were associated with the chrysophycean flagellate *Mallomonopsis* or *Mallomonas* (Section 7.3.6). It has been reported in the sea ice from the Hudson Bay system (Daugbjerg and Vørs 1994) and in the phytoplankton from the Canadian Archipelago, the western Arctic and the Hudson Bay system (Daugbjerg and Vørs 1994; M. Poulin unpubl. data) ([Appendix 1](#)).

The toxic prymnesiophycean flagellate *C. polylepis* (synonym = *Prymnesium polylepis*), responsible for massive fish kills in Scandinavia in 1988 (Granéli et al. 1993), has been recorded in BC waters (Taylor et al. 1991; Taylor 1993; Haigh 2017) and the St. Lawrence Estuary (Bérard-Therriault et al. 1999), as has *Chrysochromulina* sp. (Roy et al. 1996). Other species of *Chrysochromulina* are also common and periodically abundant. A flagellate that resembles *C. polylepis* was found in Ship Harbour, on the south shore of NS (J.C. Smith, DFO, Moncton, NB, 1994 pers. comm.). At least five species of *Chrysochromulina* have also been found in abundance in the northeastern Gulf of St. Lawrence (Levasseur et al. 1994). There are no records of fish mortalities caused by this species in Canada.

In BC, species of *Chrysochromulina*, including *C. polylepis*, have been reported in routine phytoplankton sampling (Forbes and Waters 1993b; Taylor and Haigh 1996). From 1999 to 2011, the highest abundances of *Chrysochromulina* sp. were recorded in Quatsino Sound (Haigh and Esenkulova 2012). Blooms of *Chrysochromulina* sp. have been associated with salmon kills at farm sites. These include *Chrysochromulina* cf. *hirta* (now known as *Haptolina hirta*) in Knight Inlet in September 2001 (Haigh 2017; HAEDAT [CA-01-005](#)) and *C. ericina* (synonym = *Haptolina ericina*) in Knight Inlet in May 2014 (Haigh 2017; HAEDAT [CA-14-002](#)). The latter bloom lasted ~2 weeks and affected most of Knight Inlet. Farmed salmon mortalities were seen at mostly one site, although three were affected. Maximum cell numbers were 16×10^6 cells L⁻¹ in May 2014 (ICES 2017). In Clayoquot Sound, there was a *Chrysochromulina* sp. bloom in May 2016 (6×10^6 cells L⁻¹), and again in December (6×10^5 cells L⁻¹) (ICES 2017; HAEDAT [CA-16-001](#)), with low but significant fish mortalities. *Chrysochromulina* sp. caused mortalities in Quatsino Sound, Quadra Island area, Sechelt Inlet, and Clayoquot Sound (Haigh 2017). Another *Chrysochromulina* sp. bloom occurred in 2017 (N. Haigh unpubl. data). *Chrysochromulina ericina* (synonym = *Haptolina ericina*) has been reported in the phytoplankton from the Canadian Archipelago and the western Arctic (M. Poulin unpubl. data) ([Appendix 1](#)).

A toxic compound was isolated from waters of a 1988 Scandinavian bloom of *C. polylepis*. It had haemolytic effects, and was identified as digalactosyl glycerol, esterified with polyunsaturated fatty acids (see Granéli et al. 1993). So far, there has been no chemical characterization of toxins produced in cultures. This is a knowledge gap that requires research, given the importance of the farmed salmon industry in BC.

7.3.3 Silicoflagellates (dictyochophytes)

7.3.3.1 *Octactis speculum*, *Dictyocha fibula*

Silicoflagellates (also known as dictyochophytes) are planktonic, exclusively marine, unicellular heterokont algae. They have a complex life cycle, with planktonic and benthic forms, and many species have both a naked and a skeleton-bearing planktonic stage (van Valkenburg and Norris 1970; Moestrup and Thomsen 1990; Henriksen et al. 1993; Chang et al. 2017). During the skeleton-bearing stage, the silicoflagellate produces an internal siliceous skeleton that looks like a basket composed of a network of hollow bars and spikes. The skeleton-bearing genus *Dictyocha*, had been reported to have three or four species, including *D. speculum* (also known as *Distephanus speculum*; Erard-Le Denn and Ryckaert 1990) and *D. fibula*. Based on morphological and molecular data, Chang et al. (2017) reassigned *D. speculum* and the similar species *D. octonaria* to the genus *Octactis*, as *O. speculum* and *O. octonaria*, respectively; this nomenclature is used in this review.

Both the naked (Thomsen and Moestrup 1985) and skeletal (Erard-Le Denn and Ryckaert 1990) forms of *Dictyocha/Octactis* have killed fish in Denmark and France. In France, the gills of affected farmed sea trout (*Salmo gairdneri*) were clogged by mucus in which many *O. speculum* cells were present. It is not known if this organism produces a toxin, as the naked stage did not kill *S. gairdneri* nor mice (Henriksen et al. 1993). However, it appears that its siliceous skeleton irritated the gills of *S. gairdneri*, possibly leading to its mortality (Erard-Le Denn and Ryckaert 1990).

Low numbers of *O. speculum* are commonly found in the southeastern Gulf of St. Lawrence (Bugden et al. 1992; J.C. Smith, DFO, Halifax, NS, 1996 pers. comm.), including Chaleur Bay (Brunel 1962), and in the lower St. Lawrence Estuary and central Gulf of St. Lawrence (L. Bérard-Therriault and E. Bonneau, DFO, Mont-Joli, QC, 2000 pers. comm.). It has also been reported in the Bay of Fundy (Martin et al. 2014c) and NL (C.H. McKenzie unpubl. data). *Dictyocha fibula* is also reported in the Bay of Fundy (Van Guelpen and Pohle 2019). *Octactis speculum* has been reported in the sea ice from the eastern and western Arctic, and the Canadian Archipelago (MacLaren Marex 1979a,b; Grainger and Hsiao 1982; Booth 1984; Róžańska et al. 2008, 2009; Niemi et al. 2011) and in the phytoplankton from the five Canadian Arctic regions (Bursa 1961a; Adams 1975; Thomson et al. 1975; Foy and Hsiao 1976; Hsiao 1976, 1983; Sekerak et al. 1976a,b, 1979; Hsiao et al. 1977, 1984; MacLaren Atlantic Limited 1978; MacLaren Marex 1979b; Hsiao and Pinkewycz 1985b; Percy et al. 1992; Simard et al. 1996; Gosselin et al. 1997; Harvey et al. 1997; von Quillfeldt 2000; Lovejoy et al. 2002; Riedel et al. 2003; Róžańska et al. 2008; Niemi et al. 2011; Joo et al. 2012; Simo-Matchim et al. 2017; Crawford et al. 2018; M. Poulin unpubl. data) ([Appendix 1](#)).

In BC, both *O. speculum* and *D. fibula* are regular components of the phytoplankton community, with *O. speculum* being particularly prevalent throughout the year (Haigh and Taylor 1990; Taylor et al. 1991, 1994; Haigh et al. 2004a,b,c,d,e, 2018; Haigh and Esenkulova 2012, 2014; Haigh 2017). Data from the BC salmon aquaculture industry's HAMP indicate that blooms of both *O. speculum* and *D. fibula* cause mortalities of farmed salmon, including Atlantic (*Salmo salar*) and Chinook (*Oncorhynchus tshawytscha*). Fish kills during 1997–2017 (Haigh et al. 2018) occurred with cell counts of 300–500 cells mL⁻¹ of either silicoflagellate species. Blooms of *O. speculum* were more limited in duration and extent than those of *D. fibula*, and were most often observed in the spring (April–June), whereas those of *D. fibula* occurred in summer to early autumn (July–September).

Octactis speculum was generally observed in the skeletal form, whereas *D. fibula* cells were predominantly the non-skeletal form, with mixed naked, skeletal cells and intermediate forms more common later in the blooms. Dense blooms were not always associated with fish kills; mortalities were more often observed when significant concentrations of the algae were seen throughout the water column (surface to 15 m depth), rather than at the surface only. Significant bloom years during this study were 2004, 2005, 2012 and 2013 for *O. speculum*, and 2001, 2003, 2016 and 2017 for *D. fibula*. *Dictyocha* spp. were documented at seven stations in the Strait of Georgia during 2015–2017 (Esenkulova and Pearsall 2018; Esenkulova et al. 2018). Other salmon kills were seen during blooms of *O. speculum* in 1997 (Quatsino Sound) and 1999 (Sooke Basin). Dissolved oxygen levels were not limiting during these fish kill events, and there was no obvious mechanical gill damage, although gills appeared to have increased mucus. Pathological signs in salmon mortalities were consistent with a toxic mechanism. This suggested that *Dictyocha* and *Octactis* species produce a toxin similar to that of the related species *Pseudochattonella verruculosa* (see the following section).

7.3.3.2 *Pseudochattonella verruculosa*

The silicoflagellate (Dictyochophyceae) *P. verruculosa* (formerly classified as the raphidophyte *Chattonella verruculosa*) has been reported as ichthyotoxic in Asia, New Zealand, and Europe (Yamaguchi et al. 1997; Backe-Hansen et al. 2001; Hosoi-Tanabe et al. 2007; MacKenzie et al. 2011; Chang et al. 2014). In BC waters, *P. verruculosa* (9×10^4 cells L⁻¹) killed farmed Atlantic salmon (*Salmo salar*) in Esperanza Inlet, in September 2007, and in Quatsino Sound (7.5×10^5 cells L⁻¹) in September 2008 (Haigh 2017). In late September 2013, another bloom in Quatsino Sound was again associated with a significant mortality of farmed salmon. Cells from that bloom were identified as *P. verruculosa* using molecular techniques, and were shown to cause extensive gill damage in the salmon (Haigh et al. 2014; Jones 2017).

7.3.4 Protozoan ciliate *Mesodinium rubrum*

Mesodinium rubrum (formerly known as *Myrionecta rubra*, a name that is now invalid; Garcia-Cuetos et al. 2012) is a planktonic, nontoxic protozoan ciliate known to cause brick-red blooms; hence it is often associated with the name “red tide” when it aggregates near the surface. It is found in many coastal areas of the world, including the east and west coasts of Canada (Taylor et al. 1971; White 1977; Bérard-Therriault et al. 1999; Martin et al. 2007b). The cells are roughly oval, 25–45 µm long and 18–35 µm wide, with two lobes, and are surrounded by an equatorial ciliary belt (ECB) that allows them to dart rapidly in different directions (Taylor et al. 1969). Curiously, this protozoan was found to be capable of photosynthesis when it was discovered that its plastids were derived from its food source, the cryptophyte *Teleaulax amphioxeia* (Park et al. 2006) (Section 4.2), although a different endosymbiont may be present (Hansen and Fenchel 2006). It has been further suggested that cells of this endosymbiotic ciliate were intact and expressed genes involved in all major metabolic pathways (Qiu et al. 2016), although this has been disputed (Johnson et al. 2017) and then rebutted (Qiu et al. 2017). The importance here is that the phycoerythrin-rich chloroplasts of the cryptophyte are responsible for the red colour. This organism has not proven to be toxic, but can nevertheless cause harm when the blooms decay and deplete the water column of oxygen (see below).

In BC, *Mesodinium* blooms have been recorded since the 1800s (Quayle 1969; Taylor and Horner 1994; Taylor and Haigh 1996; Taylor and Harrison 2002). In August 1967, *M. rubrum* bloomed to a concentration of 2×10^6 cells L⁻¹ in Departure Bay, Nanaimo (Parsons and Blackburn 1968; Taylor

et al. 1971). In Saanich Inlet, Vancouver Island, this species was present throughout the winter of 1975–1976, and represented up to 85% of the total carbon biomass of the protozoans (Takahashi and Hoskins 1978). In 1978, *M. rubrum* was a significant component of the plankton in the pioneering work in Saanich Inlet with large mesocosms, in the Controlled Ecosystem Pollution Experiment (CEPEX) (Grice et al. 1980).

On the east coast, *M. rubrum* has been reported in the Bay of Fundy and Gulf of Maine since 1931, based on records summarized by Martin et al. (2007b, 2009). Red water caused by this organism has been sighted in the Passamaquoddy Bay region of NB in 1989, 1993, 1998, 1999, 2000, 2002 and 2003 (Martin et al. 2007b). This number of recent observations does not necessarily indicate that these red tides are increasing in frequency, but rather it reflects the increased awareness of them due to intensified use of these coastal waters.

In 1977, a dense bloom of *M. rubrum* became trapped in a cove at Oven Head, in the Bay of Fundy (Martin and Wildish 1990). This caused oxygen depletion in the water where herring were being held captive in a weir, resulting in a major fish mortality. A similar bloom was observed in the Bay of Fundy at Passamaquoddy Bay (NB), in 1975, but with no reports of mortality (White et al. 1977). Results of a monitoring program from 1987 to 2004 indicate that *M. rubrum* was present at all stations 94% of the time (Martin et al. 2007b). Maximum concentrations (up to 7.0×10^4 cells L⁻¹) occurred yearly between May and October. Conditions that promote these blooms are apparently still elusive. Phytoplankton monitoring programs may miss seeing *M. rubrum* because the fragile organism is destroyed in conventional preservatives and in freshly collected samples.

As part of a long-term study in the Bedford Basin (NS), Kyewalyanga et al. (2002) reported on a bloom of *M. rubrum* in October 1994, which substantially changed the shapes and amplitudes of both the action and absorption spectra of the water column. This was caused by the phycoerythrin pigments in the chloroplasts of this organism's endosymbionts. It was concluded that this changes the quality of light available for photosynthesis and therefore of the photosynthetic properties of the phytoplankton. The bloom was also associated with a decline in dissolved oxygen.

Mesodinium rubrum was implicated in an unusual event in NS. In the spring of 1991, consumers of cultured blue mussels (*M. edulis*) from Ship Harbour complained about a peppery taste, a sulphur-like smell, and a deep red-brown colour in the molluscs (Watson-Wright et al. 1993a). Analyses for STX group toxins, DA and OA group toxins proved negative. After the red mussels reappeared in April 1992, a collaborative effort culminated in the discovery that they had grazed on *M. rubrum* cells (Carver et al. 1996). The colouration originated from the red phycoerythrin of the photosynthetic symbionts living inside of the *M. rubrum* cells. This is the first time that *M. rubrum* was known to colour mussels. In depuration experiments, macroscopic evidence of red colouration disappeared within 48 h, but epifluorescence microscopy indicated that phycoerythrin persisted in the digestive gland for as long as 3 weeks. The *M. rubrum* bloom declined from 3×10^4 cells L⁻¹ in early May to 1×10^4 cells L⁻¹ in early June, coincident with an increase in water temperature and the disappearance of the red colouration in the mussels. Although the mussels are not toxic, their uncharacteristic colour and taste tend to decrease their appeal to consumers, causing concern to the aquaculture industry. Similar events have occurred in NL periodically, where harvested mussels have been removed from display when the red colouration caused by *M. rubrum* in the mantle water and mussels drained onto the ice (C.H. McKenzie unpubl. data). Parrish et al. (1995) reported *M. rubrum* in South Broad Cove (Terra Nova National Park), northeast NL.

In BC, oysters growing in areas where *Mesodinium* had bloomed had also become pink (Quayle 1969). When oysters feed on these cells, their gills may become slightly pink or red, and are said to “bleed” when the oyster is opened and the red stomach contents run out of the mouth. Oysters in this condition are nevertheless quite edible, although not attractive (Quayle 1988).

7.3.5 *Hematodinium*-like protozoan parasite

Hematodinium is a dinoflagellate-like parasite that causes bitter crab disease (Bower 2013). It has infected the spot prawn (*Pandalus platyceros*) and pink shrimp (*P. borealis*) in Alaska (Meyers et al. 1994). Low rates of infection of spot prawn have also been found in localized areas in BC, in the area of Malaspina Channel in the Strait of Georgia (Bower et al. 1994). Infected prawns have an unusual orange-pink colour and accumulate a milky-like fluid in the cephalothorax. The parasite is undescribed, but ultrastructurally appears to be a close relative of the causative agent of the bitter-crab syndrome, described for Alaskan tanner crabs (*Chionoecetes bairdi*), and of *Hematodinium perezii*, described from various crab species in European and southeastern U.S. waters (Meyers et al. 1994). Rates of infection are typically <1% in affected populations in BC, although fishermen have reported rates of up to 10% (Bower et al. 1994), and affected prawns are easily sorted and discarded. Whether parasitization has an impact on stocks is still an open question, although it does affect the survival of individuals (S. Bower, DFO, Nanaimo, BC, 1995 pers. comm.).

Hematodinium sp. has also been implicated, for the first time in Atlantic Canada, in “bitter crab disease” of snow crabs (*Chionoecetes opilio*) in NL (Taylor and Khan 1995; Taylor et al. 2002; Bower 2013). As in BC, the incidence of disease was low (<0.11%) off the eastern and northeastern NL coasts in 1992–1993. However, 3.7% of the crabs examined in Conception Bay during that period were infected. Again, the significance of this dinoflagellate parasite to the snow crab fishery remains to be determined.

7.3.6 Chrysophytes *Mallomonopsis*, *Mallomonas*

In March 1994, there was a massive kill of farmed Atlantic salmon in the brackish waters of the Bras d’Or Lakes, on Cape Breton Island, NS (P. Keizer, DFO, Dartmouth, NS, 1994 pers. comm.). Analysis of the water samples revealed the presence of a Chrysomonadales flagellate, tentatively identified as either two species of *Mallomonopsis elliptica* (a synonym of *Mallomonas matvienkoae*; [AlgaeBase](#)) or *Mallomonas acaroides* (now called *Mallomonas ploesslii*; [WoRMS](#)) (4×10^5 cells L⁻¹) (C. Carver, Mallet Research Services, Dartmouth, NS, 1994 pers. comm.). These chrysophytes are characterized as having siliceous scales, each provided with a long siliceous bristle (Nicholls 1987). Although no literature was found relating this freshwater genus to fish mortality, these bristles may have damaged the gills of fish in a manner analogous to *Chaetoceros convolutus* (Section [7.1.1](#)). The dead salmon were described as having “moderate” damage to their gills. However, as this damage may not be sufficient to account for the deaths, the possibility of a toxin must still be investigated.

7.3.7 Exceptional blooms of other nontoxic phytoplankton

Given appropriate oceanographic conditions, extremely dense blooms of any nontoxic phytoplankton species may cause anoxia and subsequent mortality of benthic organisms or fish when the algae decay (e.g. Subramanian 1985). For example, the nontoxic dinoflagellate *Gonyaulax digitale* bloomed to high concentrations ($\sim 4.3 \times 10^5$ cells L⁻¹) in Bedford Basin (NS), in 1989 (Amadi

et al. 1992). This is the first time that this organism has been reported on the Atlantic coast of North America, although it is also present in the Bay of Fundy (Van Guelpen and Pohle 2019).

In BC, other problematic nontoxic organisms include *Bodo* sp., *Mesodinium rubrum* (Section [7.3.4](#)), and *Noctiluca scintillans* (Quayle 1969; Taylor et al. 1971; Haigh and Taylor 1990; Gower 1994; Taylor and Horner 1994). Blooms of the latter heterotrophic, non-photosynthetic dinoflagellate can occasionally create a dramatic tomato soup-like “red tide” colour along the coast of BC (Beairsto 2018). In other parts of the world, it can produce spectacular bioluminescence displays (hence the species name *scintillans*) in the surf zone. The dinoflagellate *N. scintillans* was originally not thought to cause deleterious effects in BC (Quayle 1969; Haigh and Taylor 1990). However, as part of the [Salish Sea Marine Survival Project](#) to determine the cause of salmon mortalities (Section [7.3.1](#)), two localized blooms of *Noctiluca* were detected in Cowichan Bay: the first in early May, and then in mid-June (Chittenden et al. 2018). These coincided with the release of hatchery-reared Chinook salmon smolts. The study determined that the *Noctiluca* blooms appeared to repel the salmon smolts from a critical early marine food source, thus reducing their survival. Although Chittenden et al. (2018) suggested that *Noctiluca* may have produced toxins, there is no evidence of phycotoxin production by this organism. However, in other parts of the world *Noctiluca* has been shown to produce enough ammonia to be toxic to fish (Okaichi and Nishio 1976). Laboratory tests that expose salmon smolts to varying cell concentrations of *Noctiluca* would improve our understanding of field results, such as in Cowichan Bay.

8.0 Introduction and Spreading of Harmful Algae via Ballast Water

8.1 Background

The introduction of harmful phytoplankton species via ships’ ballast water and ballast sediment is an increasing concern around the world (Hallegraeff et al. 1990; Kelly 1993; Murray and Hallegraeff 2018), including Canada (Bates and Keizer 1996; Claudi and Ravishankar 2006; Scriven 2014; Scriven et al. 2015) and the Arctic (Chan et al. 2018). The pioneering work of Medcof (1975) suggested the transport of organisms in ballast water. Studies have therefore been carried out to determine the presence of potentially harmful algal species in the ballast tanks of ships (Gauthier and Steel 1996). In order for the exotic organism to survive release in a new location, however, environmental conditions (e.g. nutrient availability, salinity, temperature), as well as biological factors (inter-specific competition, grazers), must be conducive (cf. Mather et al. 2010). Furthermore, before becoming established, non-native species must pass through several phases, i.e. pioneering, persistence, and community entry (Smayda 2007). Unfortunately, increasing water temperatures due to climate change (Section [10.1](#)) are increasing the potential for tropical species, including toxic dinoflagellates, to invade coastal waters of Canada (Modler 2017).

Managing the risk of introduction of HAB-forming algae to Canadian marine waters via ballast tanks has been a concern of DFO since at least the 1980s, when it was recognized that such introductions might increase the risk of a toxic algal episode affecting mussel aquaculture in the Grande Entrée lagoon of the Magdalen Islands (Gulf of St. Lawrence), an area historically free of phycotoxins. Consequently, in 1982, the Canadian Coast Guard issued *Notice to Mariners #995*, prohibiting ballast water discharge within 10 nautical miles of the Magdalen Islands, except for water

taken on in designated areas at least five nautical miles from a coast. The prohibition remains in effect (Transport Canada 2018).

Subsequent Canadian initiatives to prevent introductions of HAB-forming algae have focused on ballast water and sediment transported from outside of Canada's Exclusive Economic Zone (EEZ). Voluntary Canadian Ballast Water Guidelines were introduced in 1989, for vessels bound for the Great Lakes Basin (Locke et al. 1993). Vessels were encouraged to exchange their ballast water (which was usually of coastal or freshwater origin) for high-salinity, mid-ocean water before entering the EEZ. The Laurentian Channel was established as an alternative exchange zone. However, no ballast water management guidance was provided for vessels bound for Canadian marine ports. In 1997, the Port of Vancouver imposed its own ballast water management system under the *Vancouver Port Authority Harbour Master's Standing Orders* (Levings 1999). Fraser Port (New Westminster) and Nanaimo also adopted the protocol. Ballast water exchange was mandatory for ships arriving at these ports, with exemptions for vessels travelling from north of Cape Mendocino (CA), and for vessels carrying <1000 tonnes of ballast water. Federal regulations made ballast water management mandatory for any Canadian destination in September 2006. Acceptable management methods included exchange in mid-ocean or a designated alternative exchange zone, or an equally effective ballast water treatment. Vessels originating from points north of specified coastal biogeographic barriers (Cape Cod, on the east coast and Cape Mendocino, on the west coast) were exempt from these requirements. The disposal of ballast tank sediments (i.e. sediments that have settled out of the ballast water) was required to take place in a shore-based reception facility (Transport Canada 2018).

Canadian regulations have now been amended to align with the *International Convention for the Control and Management of Ships' Ballast Water and Sediments, 2004*, which came into force in 2017 (Transport Canada 2018). Most existing vessels may continue to exchange ballast as previously required, but over time (and no later than 2024), all vessels will be required to meet specific performance standards of the International Maritime Organization (IMO-D2 standards). For phytoplankton in the size range $\geq 10 \mu\text{m}$ and $< 50 \mu\text{m}$, the IMO-D2 discharge standard is $< 10 \text{ cells mL}^{-1}$. Note that some harmful and toxic cells that are $< 10 \mu\text{m}$ and $\geq 50 \mu\text{m}$ are unregulated in the IMO and U.S. Coast Guard protocols (Soler-Figueroa et al. 2020). It is expected that most vessels will use ballast treatment rather than ballast exchange to achieve the required performance (Transport Canada 2018).

8.2 Risk assessments of ballast water introductions

Risk assessments have consistently identified a high risk that harmful algae will be transported in ballast tanks to Canadian marine waters. A 1991 workshop of ballast water and harmful algae experts concluded that introductions of exotic phytoplankton via ballast water posed a significant and immediate threat to the marine ecosystems of Canada (Smith and Kerr 1992). Evidence was presented indicating that both the Atlantic and Pacific coasts of Canada should be considered at risk. In the absence of evidence, it was suggested that similar considerations may apply to the Arctic.

Field sampling in ballast tanks, conducted during the decade preceding the application of ballast water regulations to Canadian marine waters (1992–2002), confirmed that numerous potentially harmful marine phytoplankton taxa were being transported to marine waters of Canada in vessels entering Canada from outside the EEZ ([Appendix 3](#)). During the period of mandatory ballast water exchange (2006–2009), numerous harmful/toxic species nevertheless remained. The voluntary

guidelines designed to protect the Great Lakes Basin may have increased the risk of invasions to the Atlantic coast, as only 37 of the 336 vessels (entering the EEZ and bound for the Great Lakes Basin) for which records were available in 1990–1991, released ballast water in the Laurentian Channel or St. Lawrence River to comply with the guidelines (Locke et al. 1993). Phytoplankton samples from that study contained 32 potentially bloom-forming, red tide and/or toxigenic algal species, several of which would have been new to Atlantic Canadian waters had they been released there. Cultures of the diatoms *Leptocylindrus danicus*, *Cylindrotheca closterium*, *Pseudo-nitzschia pungens*, *Skeletonema costatum*, *Thalassiosira* spp., several microflagellates, and picoplankton (phytoplankton cells between 0.2 and 2 µm diameter), were established from unpreserved samples (Subba Rao et al. 1994).

Phytoplankton samples collected from the ballast tanks of vessels arriving in ports of the Atlantic and Pacific coasts during 1992–2002 reinforced the concern that non-indigenous and potentially harmful algal taxa were arriving by this vector in ballast water and sediments originating outside the EEZ ([Appendix 3](#)). Ballast water exchange, the only practical method available for ballast water management at that time, was demonstrably ineffective for protecting marine coastal receiving waters (Roy et al. 2012). For example, ballast water exchanges conducted during voyages from Maryland and Virginia (U.S.) to NS reduced the diversity of phytoplankton species taken on at the original port (Carver and Mallet 2004). However, 31–61% of the original taxa remained following ballast water exchange and the total abundance of phytoplankton increased rather than decreased in ballast tanks following exchange. This may have been due to an improvement in water quality in the ballast tanks when water was taken on during the exchange and/or an influx of phytoplankton with the new ballast water.

During the period of mandatory ballast water exchange, which started in 2006 (Roy et al. 2012), numerous harmful/toxic species nevertheless remained in the ballast water arriving at Canadian marine ports. Field investigations of ballast waters and sediments conducted by the Canadian Aquatic Invasive Species Network (CAISN) between 2007 and 2009 illustrated ongoing ballast water and sediment transport of potentially harmful phytoplankton ([Appendix 3](#)). For example, a large cyst bed of the potentially toxic and bloom-forming dinoflagellate *Alexandrium catenella* (as *tamarense*) (Section [2.2](#)) was reported in bottom sediments near a container ship terminal in the Bedford Basin (Halifax, NS) (Lacasse et al. 2013). Although this species is endemic to this region, their work strongly suggested that some of those cysts were introduced through discharged ballast water and sediments. While we make no attempt here to quantitatively compare the propagule pressure (abundance) of potentially harmful algal species before and after the regulatory change in 2006, colonization pressure (taxon presence) of potentially harmful algae (shown in [Appendix 3](#)) was similar during the two time periods. Sampling conducted during 2007–2009 determined that:

- Living diatoms were present in 43% of the ballast sediment samples collected after vessels arrived in Canadian Atlantic and Pacific ports (Villac and Kaczmarek 2011). If all the diatoms found in ballast sediments carried by these vessels had been released into the environment, it could have resulted in 60 new records of diatom species for the Atlantic coast and 70 for the Pacific coast (Villac et al. 2013).
- There was at least one potentially harmful dinoflagellate species present in 81% of the vessels examined on the Atlantic coast, and in 41% of vessels on the Pacific coast. Of the 156

dinoflagellate species identified from ships arriving at Atlantic ports, 15 were potentially harmful, and 46 would have been new to the ports if released (Roy et al. 2012, 2014).

- Ballast water exchange had no effect on diversity or abundance of dinoflagellates or diatoms. Abundances of diatoms and dinoflagellates in coastal exchanged and non-exchanged ships were similar to abundance ranges in natural coastal areas. In transoceanic exchanged ships (i.e. ships verified to have conducted exchange in mid-ocean), diatom and dinoflagellate abundances were occasionally three and four orders of magnitude higher, respectively, than abundances reported in mid-ocean (Briski et al. 2012, 2013).

Risks of introducing harmful algae to the Canadian Arctic must also be considered, as global warming opens up northern waterways (Section [10.1](#)). The distance traveled by ships in Arctic Canada nearly tripled during 1990 to 2015, due to increased ship traffic from general cargo vessels, government icebreakers (including research ships), and pleasure craft (Dawson et al. 2018). There is concern that introductions of exotic or harmful algal species will increase because of this augmented vessel traffic (Niimi 2007; Chan et al. 2013; DFO 2014). To date, no studies of ballast water or sediment transfer of harmful algae to the Arctic have been published. However, an investigation of biofouling assemblages on six Canadian naval vessels making voyages between Halifax (NS) and ports in the eastern Canadian Arctic during 2009–2012 detected algal taxa with potential for harmful effects, including: *Amphora* sp., *Fragilaria/Synedra* sp., *Dinophysis* sp., and *Prorocentrum minutum* (Chan et al. 2016).

Relative risks of ballast-mediated non-indigenous phytoplankton introductions are evaluated in DFO (2014). Risks of introductions under Canadian ballast management regulations in force in 2013, using vessel traffic data from 2005–2008, were ranked by pathway (receiving region/route), relative to Great Lakes Basin/international transoceanic ballast exchange (considered the pathway of lowest risk), as follows:

- 1) Annual risk of non-indigenous phytoplankton introduction (i.e. taking into account the numbers of vessels in each pathway):
 - Lowest: Great Lakes Basin/international transoceanic; Arctic / coastal domestic
 - Intermediate: Arctic / international transoceanic; eastern / coastal domestic
 - Highest: all Atlantic and Pacific categories (international coastal U.S., international exempt, international transoceanic).
- 2) Per discharge risk of non-indigenous phytoplankton introduction:
 - Lowest: Great Lakes Basin / international transoceanic; Arctic / coastal domestic
 - Intermediate: none
 - Highest: Arctic / international transoceanic; eastern / coastal domestic, all Atlantic and Pacific categories (international coastal U.S., international exempt, international transoceanic).

Future risk projections suggest that the relative risks of ballast-mediated non-indigenous phytoplankton introductions, when conducted so as to meet the new international performance standards IMO-D2, will change very little from the present-day relative risks (DFO 2014). Relative annual and per discharge risk under IMO-D2 remained the same for all pathways except for

“eastern/coastal domestic”, for which annual and per-discharge-event risk were moved to the ‘Lowest’ category. Absolute risk has not been assessed for either the present-day or future scenarios.

8.3 Examples of introductions via ballast transport

It is difficult to unequivocally state that a given taxon was introduced via ballast transport as opposed to introduction by other vectors. However, invasive species experts consider ballast water and sediment to be the most common vectors of phytoplankton (e.g. Ruiz et al. 2015). Several examples strongly support a ballast vector for the introduction of potentially harmful algal taxa into Canadian marine waters. Despite the requirement for ballast water exchange intended to protect the Grande Entrée lagoon of the Magdalen Islands from harmful algae (*Notice to Mariners* #995), the potentially harmful dinoflagellates *Alexandrium* and *Dinophysis* were reported there for the first time in 1989 (Larocque and Cembella 1991). The ballast water of vessels docked in Grande Entrée lagoon, in 1992, contained vegetative cells of *A. catenella* (as *A. tamarense*), *D. acuminata*, *D. rotundata* (= *Phalacroma rotundatum*), and/or *D. norvegica*, and ballast sediments contained cysts of *A. catenella* (Gosselin et al. 1995). In this case, the species were probably carried to the Grande Entrée lagoon from elsewhere in the Gulf of St. Lawrence or waters south of NL, where the sampled vessels had exchanged their ballast water while travelling from domestic ports to the Grande Entrée lagoon.

An example of a species apparently introduced to the Gulf of St. Lawrence in ballast water is the diatom *Pseudo-nitzschia fraudulenta*, some strains of which produce DA (Bates et al. 2018). It had not been observed in the Gulf of St. Lawrence before 2001 (Carver and Martin 2003), but was observed in four locations in northern NS (Caribou Harbour, Wallace Harbour, Little Harbour, and Pugwash Harbour) on December 20, 2001, just before the December 22 shellfish harvesting closures of this area (Section [3.2.1](#); HAEDAT [CA-02-001](#); J. Ehrman and S.S. Bates unpubl. data). In the month before this observation, *P. fraudulenta* was frequently sampled in the ballast water of vessels arriving in Atlantic Canada, including ports on the northern coast of NS, after having taken on ballast water from the northeastern U.S. and the Atlantic coast of NS (Carver and Martin 2003). Ballast water transport may also have introduced *P. fraudulenta* to the Bay of Fundy, where it was first reported by Martin and LeGresley (2008). It has been also reported in the phytoplankton from the eastern Arctic (Mather et al. 2010) ([Appendix 1](#)).

Several other potential harmful species reported for the first time in Atlantic Canada have also been detected in ballast water arriving in the region. Examples include *Pseudo-nitzschia subpacifica* and *Alexandrium pseudogonyaulax*, observed in sampled ballast water by Carver and Mallet (2001, 2002, 2003) and Roy et al. (2012), respectively. Both species were first recorded in the Bay of Fundy by Martin and LeGresley (2008). *Alexandrium pseudogonyaulax* was subsequently reported from the St. Lawrence Estuary by Dufour et al. (2010). The diatom *Trieres chinensis* (as *Odontella sinensis*) was detected in the Bay of Fundy in 2000 (Martin and LeGresley 2008). Although a possible mode of transfer could have been through natural dispersion via currents, *T. chinensis* is thought to have come through ships’ ballast because the first documented observation in the phytoplankton monitoring program in 2000 coincided with its observation from a ballast water sample taken from a ship at the port of Bayside.

The following harmful/toxic species have been detected in ballast water tanks of ships in the Canadian Arctic: *Alexandrium ostenfeldii*, *Dinophysis caudata*, *D. norvegica*, *Phalacroma*

rotundatum, *Gonyaulax spinifera*, *Karenia mikimotoi*, *Heterocapsa triquetra*, *Prorocentrum minimum* (= *P. cordatum*), *Protoceratium reticulatum*, *Protoperidinium crassipes* (Dhifallah 2019).

The preceding examples demonstrate that it is not only possible, but probable, that ballast waters inoculate Canadian waters with non-indigenous harmful/toxic algae. It is therefore essential to: 1) document the phytoplankton already present in Canadian waters; 2) identify harmful algae that have the potential to be introduced via ballast waters; and 3) study the conditions that would favour the development of the harmful blooms (Taylor 1995).

9.0 Monitoring Programs for Phycotoxins and Harmful/Toxic Phytoplankton

9.1 Monitoring approaches

Monitoring the causative species and/or the presence of phycotoxins in seafood in real time is currently the only effective way to protect human health, as early warning prevents contaminated shellfish from reaching the markets (Wright 1995; Berdalet et al. 2016; Reguera et al. 2016; Langlois and Morton 2018). Such monitoring programs have proven effective in reducing human exposure to phycotoxins in many parts of the world, including Canada (Anderson et al. 2001), for example in the Bay of Fundy (Martin et al. 1999, 2001b, 2004b, 2006a; Sephton et al. 2003; Chang et al. 2005, 2006, 2007b) and PE (Smith et al. 1990a; Bouchard-Steeves et al. 1993a). As well, they provide information that is useful for aquaculture site selection in Canada and for its management to reduce potential effects on the industry from HABs (McKenzie et al. 1998; Keizer and Zurbrigg 2007; F. Page, DFO, St. Andrews, NB unpubl. data). Anderson et al. (2019) summarize regional efforts worldwide to create state-of-the-art HAB monitoring and forecasting tools, vulnerability assessments, and observing networks, which are also relevant to Canada.

Current regulation for several of the marine phycotoxins requires a monitoring method based on mass spectrometric analysis. However, Botana et al. (2016) discuss several limitations of this analytical chemistry approach: novel toxins will not be detected, because the monitoring protocol is pre-targeted; there is a poor comprehension of the toxicity of some toxin groups detected; and there is a lack of sufficient calibrants for some phycotoxins. Nevertheless, recent advances in approaches for non-targeted monitoring mean that it is now possible to detect and identify novel toxins by LC-MS and in the future/near future this may be an approach that is more widely used. On the other hand, live animal bioassays, which have been touted by Botana et al. (2016) as a “universal” detector of toxins, have so many problems that they have been banned from routine monitoring in most countries. Besides the animal rights issues, there are other serious concerns, e.g. the assays are not validated and suffer from both false negatives and false positives. Biochemical and immunological assays are lacking for some toxin groups and they can also have significant problems with both false negatives and false positives. In recent years, mass spectrometry and other chemical analytical methods have proven to be very effective for protecting public health and addressing the needs of international trade, which require validated quantitative analyses of specific toxins. The combination of multiple detection methods can provide additional information, such as the combination of the PPIA and LC-MS/MS methods used to detect a OA group toxin, 14,15-dihydroDTX1, novel in shellfish in coastal Maine (Deeds et al. 2018) (Section [4.7](#)).

Unfortunately, phytoplankton monitoring programs have been reduced in Canada during a period of budgetary cutbacks (Martin et al. 1995, 1999, 2001b, 2006, 2014c; Bates and Keizer 2006; Martin and LeGresley 2014). In some cases, industry has stepped in to provide at least a minimal program to serve their specific requirements. Some of the programs presented below no longer exist, but will nevertheless be described in the event that they may once again be implemented.

Until April 1997, the monitoring of shellfish toxins to ensure the safety of seafood products was the responsibility of the Inspection Branch of DFO. Stephen (1991) described the east coast Inspection Branch monitoring program. The responsibilities were then transferred to the newly created CFIA, although most methodologies and personnel were retained. The CFIA headquarters is in Ottawa, with regional laboratories in Atlantic Canada (Dartmouth, NS), Quebec (Longueuil, QC), and the west coast (Burnaby, BC) conducting analyses for biotoxins ([Appendix 3](#)). The history and evolution of monitoring activities for phycotoxins in bivalve shellfish by the CFIA's Dartmouth Laboratory, up to 2004, is reviewed by van de Riet et al. (2006).

The CFIA is responsible for arranging the collection of shellfish samples from specific monitoring sites, commercial shellfish shipments, and shellfish growing areas on a regular basis. The CFIA analyzes shellfish tissue for the presence of phycotoxins, and then recommends the closure of shellfish harvesting sites if the phycotoxin levels approach or exceed the guidelines of the Canadian Shellfish Sanitation Program (CSSP 2019) ([Table 3](#)). DFO then has the legal responsibility for issuing harvest Prohibition Orders based on the CFIA recommendations, and for posting closure information at the harvesting sites and with the media. These Prohibition Orders are enforced by DFO Conservation and Protection staff, who conduct patrols in areas closed to shellfish harvesting (Section [1.1](#)). As detailed elsewhere in this review, harvesting areas are reopened only after three samples over a 14-day period are found to contain less than the action level specified for a given phycotoxin (CSSP 2019).

Several alternative monitoring approaches may be used to give a “heads up” about an impending toxic event that could lead to harvesting closures, or to mortalities of finfish. First, water samples may be taken to determine the presence and abundance of toxic, or potentially toxic, phytoplankton. However, there is presently no general consensus on action limits for algal cell concentrations, only recommendations in different jurisdictions (e.g. Mexico, U.S., EU) that should be used to manage potentially toxic shellfish (Anderson et al. 2001). Depending on the cell toxicity, each species (and even strain) of toxigenic phytoplankton would have a different cell concentration at which a bloom could result in a toxin concentration over the limit in shellfish. Cell concentrations of HAB species that trigger the implementation of restrictions on shellfish harvesting for various countries are given in Anderson et al. (2001). Although this method can give a good indication of the presence of toxigenic phytoplankton, and therefore of the potential for toxicity in shellfish, it has some disadvantages. For example, there are uncertainties when deciding on the location and frequency of sampling for toxic phytoplankton. Monitoring sites should be located near harvesting sites, but toxic blooms may also originate offshore. Sampling should be frequent enough not to miss a bloom, but this adds to the cost of monitoring. There are problems in detecting *Prorocentrum lima* cells, which are not always in the water column, but sometimes grow epiphytically on mussel ropes or filamentous seaweeds, or on the sediments (Section [4.2](#)). Cells can be stirred up from sediments or released from epibionts during storms and then be consumed by shellfish, resulting in the accumulation of OA group toxins. Although using a light microscope may be faster than applying molecular methods or using instruments such as the FlowCam to identify toxic phytoplankton, and it also has this advantage of providing cell

concentrations, this approach can still be time-consuming and requires taxonomic expertise. For some phytoplankton (e.g. *Pseudo-nitzschia*), transmission or scanning electron microscopy is required to identify which species is present, as some species of a genus are not toxigenic (Bates et al. 2018, 2019). Molecular methods are being used increasingly to identify and even quantify toxigenic species. These rely on studying rDNA in the small subunit (SSU), large subunit (LSU) and the internal transcript spacer (ITS) regions (reviewed in Bates et al. 2018, 2019 for *Pseudo-nitzschia* species). A molecular method can be used to detect and enumerate cysts of *Alexandrium catenella* (as *A. tamarense*) in sediments (Erdner et al. 2010). Another molecular approach detects *Alexandrium* cells, but only to the genus level (Hatfield et al. 2019). Microarray technology is also being used to detect multiple HAB species simultaneously (e.g. Medlin and Orozco 2017). Another approach is to detect targeted HAB species and the phycotoxins they produce by use of automated *in situ* methods (Wang et al. 2019b), e.g. moored Environmental Sample Processor (e.g. Bowers et al. 2016, 2017, 2018; Scholin et al. 2017), the [Imaging FlowCytot](#), or portable imaging flow cytometers (Göröcs et al. 2018), techniques that could be used to quantitate particles globally (Lombard et al. 2019).

An additional factor to consider is that phytoplankton monitoring conveys only that there is a risk that phycotoxins are present, not that shellfish are unsafe for consumption. Nevertheless, high phytoplankton counts could be used as a trigger to sample shellfish for phycotoxins (potentially in conjunction with a precautionary closure while waiting for shellfish test results) to determine the actual risk in order to minimize the chance of closing harvest areas unnecessarily. Phytoplankton monitoring can also provide an indication that cultured finfish may be at risk. Aquaculturists can take precautionary measures in order to reduce harmful effects.

Airborne remote sensing, using the Compact Airborne Spectrographic Imager (CASI) mounted in a single engine float plane, has been used to detect surface phytoplankton blooms in BC (Borstad 1991). A trial use of aircraft for remote sensing of toxic *Pseudo-nitzschia* blooms in Cardigan Bay (PE) was carried out in October–November 1989 (Sathyendranath et al. 1997). Although algorithms were developed to successfully map the local distribution and development of a bloom, laboratory measurements showed no unique optical signatures specific to *Pseudo-nitzschia*. Thus, until such a “toxic-diatom-bloom signal” can be developed, it was recommended that *in situ* sampling should remain an integral part of monitoring programs. Nevertheless, satellite imagery was used to provide information on toxic *Pseudo-nitzschia* blooms in the *rias* of Spain (Torres Palenzuela et al. 2019).

Satellite imagery, using ocean colour radiometry, is another approach for detecting the development of HABs (Bernard 2016; Doucette et al. 2018). For Canadian or adjacent waters, this has been used for the Canadian east coast and the North Atlantic (Sathyendranath et al. 2004), in the Bay of Fundy (Harrison et al. 2007; Devred et al. 2018), Gulf of Maine/Bay of Fundy (McGillicuddy et al. 2014), the BC coast (Gower and Borstad 1991a,b; Gower 1994), and the Oregon coast (McKibben et al. 2012). However, as discussed in these papers, challenges include not being able to discriminate among the different groups of toxic/harmful phytoplankton, interference from sediments in coastal waters, and the inability to detect low concentrations of *A. catenella* (200 cells L⁻¹ can still be problematic; Page et al. 2004). Nevertheless, satellite imagery can be combined with other seawater measurements as an early warning and to optimize an *in situ* monitoring system (Harrison et al. 2007; Devred et al. 2018; Torres Palenzuela et al. 2019).

Canada does not rely on toxic phytoplankton cell counts to regulate harvesting closures because only the phycotoxin level measured directly in molluscan shellfish tissue is accepted by the CSSP. Nevertheless, the concentration of *Pseudo-nitzschia* cells has been used by the Prince Edward Island Department of Aquaculture and Fisheries to provide alerts to the mussel-culture industry (Section [9.2.2.1](#)). In the Bay of Fundy, the DFO phytoplankton monitoring program alerted the CFIA when concentrations of *Pseudo-nitzschia* cells increased. This information was used to determine when and where increased monitoring for the presence of DA in shellfish tissue should occur. Similarly, increases in the presence and concentration of *A. catenella* served as an alert of impending STX group shellfish toxicity for the CFIA as well as for the salmon aquaculture industry (Section [2.4.1.1](#)).

A second approach is to monitor phycotoxins dissolved in seawater, using passive samplers, referred to as Solid Phase Adsorption Toxin Tracking (SPATT) technology (MacKenzie et al. 2004; Robertson et al. 2006). This uses porous synthetic resins that passively adsorb dissolved phycotoxins (reviewed in Kudela 2017; Roué et al. 2018). Although SPATT bags have been used in research to trap PTX2 and PTX2 seco acid (Section [5.1](#)), YTXs (Section [5.2](#)), AZAs (Section [5.3](#)), and SPXs (Section [6.1](#)) in the waters of NS, they are not certified for use as a regulatory tool. The SPATT technology has also been used to adsorb OA group toxins (OA, DTX), other cyclic imines (PnTXs, GYMs), DA, STX group toxins, and microcystins (Zendong et al. 2015; Roué et al. 2018). Passive samplers have the advantage that, unlike in mussels, the adsorbed phycotoxins do not undergo biotransformation (Zendong et al. 2015). Once adsorbed, the toxins must be extracted and analyzed, e.g. by LC–MS/MS. These passive samplers have been proposed to support a more holistic regulatory approach, since SPATT resins adsorb both lipophilic and hydrophilic toxins. SPATT devices have the advantage of providing a spatially and temporally integrated response for the detection and quantification of multiple phycotoxins, to better assess and manage the risks posed by HABs. However, they have disadvantages (Roué et al. 2018), for example: 1) there is yet no general consensus about the optimal deployment time for SPATT devices in the field, and this can vary from a few hours, to weekly, or monthly. Thus, the saturation limit of a given toxin may or may not be exceeded. 2) The adsorption efficiencies for each toxin and resin may vary with environmental conditions. 3) Toxin levels retained on SPATT devices are reported in relative units (e.g. ng toxin g⁻¹ of resin) and cannot be readily converted into quantitative toxin concentrations in the environment. More studies are needed to standardize the methodology for inter-laboratory comparisons. However, the use of SPATT technology alone would not be acceptable for regulatory monitoring of shellfish toxicity since it does not conform to CSSP guidelines. Nevertheless, this technology may be useful as a supplementary technique because it indicates the potential presence of phycotoxins in the environment.

Monitoring the behaviour of molluscan bivalves is another potential “heads up” approach for signalling an impending toxic event. The published literature contains sparse, but growing, experimental evidence that certain bivalves respond rapidly (~1 h) to toxic microalgae by modifying their gaping behaviour. Such behavioural responses to toxic microalgae have been reported for at least six bivalve species (references in Comeau et al. 2019). For example, relatively low concentrations of toxic dinoflagellates (500–1200 cells L⁻¹) increased the frequency of valve microclosures in *Mytilus galloprovincialis* (Comeau et al. 2019), *Ruditapes philippinarum* (Basti et al. 2009), and *Crassostrea gigas* (Haberkorn et al. 2011). In parallel, new engineering initiatives are transforming laboratory biosensors into field-deployable instruments (Garcia-March et al. 2016; Ballesta-Artero et al. 2017; Miller and Dowd 2017), with some devices integrating real-time

monitoring capabilities in conformity with the concept of biological early-warning systems (Kramer and Foekema 2001; Borcharding 2006; Andrewartha et al. 2015). The MolluSCAN eye, for instance, is a highly specialized valvometry system that automatically transfers real-time data on valve movements of sentinel bivalves through a mobile network (Andrade et al. 2016), allowing land-based servers to scan for abnormal behaviour. Such systems could potentially provide an early warning by alerting stakeholders on the timing and location of developing HABs. While *in situ* biosensors might not replace conventional field sampling and toxin analyses, they could optimize the allocation of costly diagnostic resources.

One monitoring approach that several states in the U.S. have taken is for state and federal regulatory agencies to partner with trained volunteer citizen groups. A similar citizen program has recently been implemented in France (Siano et al. 2020). These programs allow greater coverage in time and space, and improves the early warning of HABs (e.g. Bean et al. 2005; Shuler et al. 2012; Trainer et al. 2014, 2015a).

In Canada, a [Citizen Science Program](#) is operated by [Salish Sea Marine Survival Project](#), in BC (Esenkulova and Pearsall 2019). Started in February 2015, the program uses fishing vessels to collect oceanographic data during the spring and neap tides at 11 locations in coastal waters of the Strait of Georgia. One component collects water samples to identify phytoplankton, as part of an examination of the spatial and temporal prevalence of harmful algae throughout the Strait of Georgia. Data on the prevalence of *Heterosigma akashiwo*, *Chaetoceros convolutus*, *C. concavicornis*, and *Dictyocha* spp. from this program are presented by Esenkulova and Pearsall (2018) and Esenkulova et al. (2018). Such citizen science programs could be expanded to other Canadian coasts.

9.2 Monitoring in Canadian coastal waters

9.2.1 Pacific coast

The history of the monitoring program for BC, up to 1994, is reviewed in Taylor and Horner (1994), and up to 2012 by Haigh and Esenkulova (2012, 2014). Shellfish have been monitored for STX group toxins every year on the BC coast since an outbreak in 1942 (Taylor and Horner 1994). Finnis et al. (2017) used data from 49 monitoring sites during 2002–2012 to review the spatiotemporal patterns of STX group toxins and their relationships with environmental variables in BC. The authors concluded that their analysis “has the potential to optimize biotoxin monitoring, improve public health surveillance, and engage the shellfish industry in helping to reduce the risk of PSP”. Forbes (1991b) listed the advantages and disadvantages of different phytoplankton monitoring approaches (direct water monitoring, airborne monitoring, satellite monitoring, and predictive monitoring) in BC waters.

For management and control purposes, the BC coast is divided into 48 fisheries management areas, each of which is in turn divided into a variable number of subareas. Unfortunately, because of the length and complexity of the coastline (~18,000 km), it is impossible to cover the whole region adequately. Consequently, large areas of the coast are permanently closed to private shellfish collection. Although many locations are sampled from the whole coast, only the Strait of Georgia, southern Queen Charlotte Sound, Juan de Fuca Strait, and Barkley Sound have been monitored regularly (Taylor and Horner 1994). With the exception of First Nations subsistence fisheries, the northern waters have fewer shellfish harvesting activities. Therefore, samples are generally provided only to monitor specific fisheries such as razor clams in the Haida Gwaii archipelago, geoduck clams

(*Panopea generosa*) throughout this area and littleneck clams on the central coast. The monitoring includes samples of both mussels and the commercial species being harvested. Monitoring for STX group toxins was initiated in 1942, and testing for DA was added in 1992 (Taylor and Horner 1994).

Shellfish toxicity records are summarized for 1982 to 2001 (DFO 1982, 1983, 1984, 1985, 1986, 1987, 1988, 1989, 1990, 1991, 1992, 1993, 1994, 1995, 1996; CFIA 1997, 1998, 1999, 2000, 2001). These reports indicate that some management subareas require closure for the harvesting of some species of shellfish on an annual basis. The prevalence of STX group toxins along the Pacific coast is illustrated by the fact that all subareas monitored during 1982 to 2001 required at least one temporary closure due to samples exceeding the action limit. There are indications of a seven to eight year periodicity in STX group toxins in molluscan shellfish on the Pacific coast (Gaines and Taylor 1985).

For the marine biotoxin monitoring program in BC, large mussels (as well as a lesser number of commercial samples of all species) are hung in plastic mesh sacks in shellfish growing areas. Shellfish are sampled weekly or biweekly and shipped to CFIA laboratories for analysis ([Appendix 3](#)). Because mussels tend to be indiscriminate feeders of phytoplankton, they accumulate phycotoxins more quickly than many other shellfish species. Their levels of phycotoxins are also frequently up to 10 times higher than in oysters, manila clams or littleneck clams growing in the same area. This allows the CFIA to recommend harvest closures to DFO while species other than mussels, e.g. clams, oysters and scallops, are still safe. Mussels also tend to rid their tissues of biotoxins faster than other species. Therefore, although harvest restrictions are usually based on mussel analysis results, samples of the other species in the area are tested before the harvest restrictions are lifted.

Contamination by STX group toxins has been regularly assessed by various government agencies in BC since 1942 (Quayle 1969; Bond 1974; Chiang 1985, 1988, 1991; Bugden et al. 1992; DFO 1994; Taylor and Horner 1994). Initially, there was considerable variation in the extent of the program. This resulted both from changing perceptions of risk and from divided authority and responsibility for control (Quayle 1969). Since 1963, a comprehensive shellfish monitoring program was operated by DFO, until it was taken over by the newly created CFIA in 1997.

Schallié (2001) and Schallié and Kelly (2008) described the BC biotoxin monitoring as of November 2008. The CFIA has entered into partnerships with various sectors of the shellfish industry and First Nations groups, which are responsible for providing the shellfish samples. CFIA staff administer the program and analyze the shellfish tissue samples. In the southern waters of BC, where most of the shellfish are grown and harvested, partners chose ~70 sites to also provide information regarding the occurrence of toxic phytoplankton blooms in shellfish growing and harvesting areas.

Mesh sacks of mussels are suspended in the water at these sites and samples are withdrawn from the sacks once per week during the high risk part of the year (May to October) and once every two weeks for the low risk period (November to April). These samples are shipped to the CFIA laboratory in Burnaby (BC), where they are analyzed for STX group toxins and DA, and to the CFIA laboratory in Dartmouth (NS), where they are analyzed for OA group toxins and other lipophilic toxins (AZA, PTX, YTX, and cyclic imines) ([Appendix 3](#)).

An effective program must be able to detect and respond to toxic blooms in a timely manner in order to minimize risk to public health. This risk is a function of the extent of the monitoring program

and the response time required to implement closure actions (Chiang 1988). The present monitoring program in BC encompasses 73 sites on the south and central coasts where mussels (*Mytilus californianus* or *M. edulis*) are sampled on a weekly or biweekly basis, with a further 15 sites to support the geoduck clam (*Panopea generosa*) fishery on the Haida Gwaii archipelago (DFO 1994). Additional samples are collected by fishery officers and patrol crews, commercial geoduck clam and scallop fishermen, oyster growers, and other commercial harvest sectors, to supplement the mussel monitoring sites. Samples of commercial bivalves are also periodically taken at federally registered processing plants as verification of the monitoring program. Because of the remoteness of the northern coast and of the Haida Gwaii archipelago, virtually all of those coastlines have remained closed to molluscan shellfish harvesting.

From 1986 to 1992, information about *Heterosigma akashiwo* (Section 7.3.1) and other phytoplankton harmful to farmed salmonids was reported to a “plankton watch” coordinator, who compiled the information and made it available via a recorded telephone message (Stockner 1991). The program was funded by the provincial government and BC salmon farmers (Black 1989). As a result of phytoplankton monitoring, the loss of finfish in cages decreased from 38% of all stock due to *Heterosigma* blooms in 1986, to 4% in 1990 (Black 1991). After this program ceased, most BC salmon farming companies continued some sort of on-site monitoring for the presence of harmful phytoplankton taxa. This independent effort helps to protect their investment, and is required for insurance purposes. Spence (1991) outlined the importance of a phytoplankton monitoring program to protect the salmon farming industry. A volunteer Citizen Science Program now operates at several stations in the Strait of Georgia (Section 9.1).

Striving for efficient HAB management and mitigation, a consortium of BC salmon fish farmers, in association with DFO, initiated the Harmful Algae Monitoring Program (HAMP) in 1999 (Haigh and Esenkulova 2011, 2012, 2014; Haigh 2012). Funding was provided by the aquaculture industry, and DFO Science provided in-kind funding through expertise and laboratory and office space. Since 2004, HAMP has been fully supported by the aquaculture industry and run by Nicola Haigh, through her company Microthalassia Consultants. Housed at Vancouver Island University from 2004 to 2016, HAMP is now based off campus and is operated by Jay Pudota. HAMP personnel have monitored 72 sites (to 2017) since 1999, on or near fish farm sites around Vancouver Island and in the BC Central Coast. Of these, 10 have been monitored for over 10 years, 12 have been monitored for 5–9 years, and 10 have been monitored for 3–4 years. The rest were monitored for one year only. Samples are taken by fish farm personnel every week at 1, 5, and 10 m depths, during the spring to fall, and sent to Nanaimo for analysis. The samples are analyzed for the presence and concentration of at least 11 possible harmful algae, which has culminated in warnings of potentially harmful events. This monitoring program augments the routine (usually twice-daily) plankton sampling and analysis that is done at all BC salmon farm sites by industry personnel. As a result of this program, the first incidence of DSP in BC was well documented (Section 4.5), and novel fish-killing species of algae found in BC waters since 1999 have been identified (Section 7.3.1). Detailed information on the history of phytoplankton monitoring to assist the salmon farming industry, along with sampling sites and data analysis, can be found in Haigh et al. (2004a,b,c,d,e).

Modelling is one approach for forecasting HABs (Anderson et al. 2015; Flynn et al. 2018). For example, isolates of *A. catenella* (as *A. fundyense*) from the Salish Sea were grown in culture, and the effects of temperature and salinity on the dinoflagellate’s growth were studied (Bill et al. 2016). The

data resulted in an empirical equation that has the potential for identifying periods when Salish Sea *Alexandrium* are at the highest risk of blooming. Likewise, a “deep learning” neural network algorithm has been used to provide weekly site-specific forecasts of STX group toxin levels in coastal Maine, with a high forecast accuracy, generally >95% (Grasso et al. 2019).

In BC, there is no monitoring of phytoplankton to provide an alert for DA in support of CFIA’s regular shellfish monitoring program. However, while mussels appear to take up only modest quantities of DA and depurate rapidly, an assessment has indicated that the sampling frequency and extent of the mussel monitoring program in BC is adequate to detect toxic events (Whyte et al. 1995). Barth et al. (2019) provide an example of cooperation between Canada and the U.S. with respect to sharing data from a wide range of multidisciplinary ocean sensors along the Northeast Pacific. These have been used to track HABs, ocean acidification, hypoxia, ocean productivity, and upwelling.

9.2.2 Atlantic coast

After the 1987 DA crisis in eastern PE, DFO developed an expanded phytoplankton monitoring program over the next three years throughout the Atlantic zone (Bugden et al. 1992). The initial goal of this program was to improve scientific understanding of phytoplankton ecology. Its objectives were also to: 1) determine what areas and times are favourable or unfavourable for shellfish or finfish aquaculture with regards to the presence of toxic phytoplankton; 2) indicate times when screening for phycotoxins should be more or less frequent if a consistent species succession can be established; and 3) provide background information for gauging whether observed phytoplankton events are normal or whether changes in biomass and species diversity may be related to exceptional meteorological events or anthropogenic activity. Variables measured included phytoplankton species enumeration, taxonomy and quantitative abundance, chlorophyll, temperature, salinity, and inorganic nutrients.

Observations of these toxic phytoplankton and the physical parameters affecting their growth have proven to be important for describing and predicting bloom events and for identifying potentially toxic events and subsequent shellfish contamination. However, various budget cutbacks at DFO due to the Program Review in the 1990s ([DFO 1998-99 Estimates](#)) resulted in a slow erosion of the capabilities for research and monitoring of toxic phytoplankton and phycotoxins. This concern was addressed at a DFO workshop in 1996 that included representatives from east coast federal and provincial agencies, the aquaculture industry, and the private sector (Bates and Keizer 1996); the workshop also summarized the existing phytoplankton and phycotoxin monitoring programs in the DFO Maritimes Region. Below is a summary of phytoplankton monitoring programs that have existed in eastern Canada.

9.2.2.1 Nova Scotia

Beginning in November 1988, the Maritimes Region of DFO initiated a survey of potentially toxic phytoplankton, along with physical and chemical properties of the water column, at five coastal sites in NS (Whitehaven Harbour, Ship Harbour, St. Margarets Bay, Woods Harbour and Annapolis Basin) over a 3-year period (Keizer et al. 1996). Starting in 1991, a phytoplankton monitoring program was operated jointly by DFO, the Nova Scotia Department of Fisheries & Aquaculture, and the Aquaculture Association of Nova Scotia (AANS) to protect consumers and the aquaculture industry in NS (Watson-Wright et al. 1993a). However, this program is no longer funded and has ceased operation (I. Tremblay, AANS, Halifax, NS, 2018 pers. commun).

On the eastern shore of NS, monitoring of potentially toxic phytoplankton species was carried out by the NRC, in Ship Harbour, starting during June to December 2004 (Garnett et al. 2005), and ending in 2006. Continuous physical and optical measurements were obtained from a moored instrument for much of the time. Shifts in species composition were observed, with *Pseudo-nitzschia delicatissima*, *Alexandrium ostenfeldii* and *A. catenella* (as *tamarense*) occurring primarily in June and July, followed by *P. seriata* in late July, and then a peak of *Dinophysis acuminata* and *D. norvegica* in August.

9.2.2.2 Prince Edward Island

Because the above phytoplankton monitoring programs were not designed to provide an operational early-warning system, DFO's Inspection Branch integrated phytoplankton monitoring into its Phycotoxin Monitoring Program, in 1990, for the southeastern Gulf of St. Lawrence (Bouchard-Steeves et al. 1993a,b,c,d). In particular, monitoring the presence and concentrations of *Pseudo-nitzschia multiseries* in bays of PE had the advantage of providing a 10–14 day advance warning of possible accumulation of DA by molluscan shellfish (Smith et al. 1990a; Bouchard-Steeves et al. 1993a). The inclusion of *P. multiseries* monitoring had resulted in more efficient and effective management of resources. It allowed redirection of sampling efforts from low risk to high risk areas, and reduced costs by analyzing for DA with HPLC only when *P. multiseries* (and also *P. pseudodelicatissima*) began to multiply. This program ceased in 1996 due to budgetary pressures.

The Prince Edward Island Department of Agriculture and Fisheries operates a [Mussel Monitoring Program](#) as a service to the mussel aquaculture industry (Bernard 1994; Bates and Keizer 1996). It also monitors for the concentration of total *Pseudo-nitzschia* species and for the presence of *Alexandrium* and *Dinophysis* species. Because not all *Pseudo-nitzschia* species produce DA (Bates et al. 2018), such a monitoring program is useful only if all *Pseudo-nitzschia* cells present can be identified to the species level. Therefore, when the total number of *Pseudo-nitzschia* cells increased, it was important to know if any toxigenic species were present. On those occasions, DFO provided funding for samples to be sent to the [Digital Microscopy Facility](#), where *Pseudo-nitzschia* were identified to the species level using SEM. Most of the time, only nontoxic species were encountered, which alleviated fears for the mussel aquaculture industry. However, occasionally, toxigenic *Pseudo-nitzschia* species were present. This SEM service was discontinued when funding was no longer available.

9.2.2.3 Bay of Fundy

A salmonid aquaculture industry was introduced to the southwest NB portion of the Bay of Fundy in 1979, and subsequently experienced a period of rapid growth. With the increased number of aquaculture sites and increased awareness of coastal systems being impacted by human activities throughout the world, DFO's SABS initiated a phytoplankton monitoring program in 1987, in the Western Isles region of the Bay of Fundy (Chang et al. 2007b). It was through this program that the organism associated with the production of DA (*Pseudo-nitzschia pseudodelicatissima*) was detected in 1988 (Martin et al. 1990a; Haya et al. 1991). The goal was to study the whole phytoplankton community, including HABs, with the specific purposes to: 1) establish baseline data because little detailed work had been published since earlier studies by Gran and Braarud (1935); 2) document species composition and abundance; 3) determine temporal and spatial variations of phytoplankton species; 4) determine patterns and trends; 5) act as an early

warning of HABs to both the salmon aquaculture industry and the CFIA; 6) provide data for hindcasting events; 7) determine whether changes in species occur; and 8) determine linkages with the physical and chemical oceanography. In addition to phytoplankton identification and abundance, the following parameters were also measured weekly during bloom periods and monthly during late October through late March: Secchi depth (a proxy for chlorophyll abundance), and depth profiles for fluorescence, temperature, salinity, and nutrients (ammonium, nitrite, nitrate, phosphate, silicate) (Martin et al. 1995, 1999, 2001b, 2008b, 2009, 2014b; Martin and LeGresley 2014). This monitoring project was a collaborative effort involving scientists from DFO and salmon farmers in southwest NB, with funding provided by the DFO ACRDP, DFO Science, and the participating companies (Chang et al. 2007b).

The number of sampling sites has changed since the program started in 1987. Initially, 12 sites were sampled, with 10 located in the Letang area, where the majority of the aquaculture sites were located at that time, and the remaining two were in Harbour de Lute (Campobello Island), in close proximity to other aquaculture sites. The following year (1988), the number of sampling sites expanded to 18, with additional sites in Passamaquoddy Bay (near Deer Island), Deadmans Harbour, and an offshore site at The Wolves Islands.

In 1992, sampling was reduced, due to financial constraints, to the four stations that continue to be monitored at present ([Fig. 12](#)) (Martin et al. 1995, 1999, 2001b, 2006, 2014b; Martin and LeGresley 2014), with subsidies from other programs. These sites include: Brandy Cove (station #17 – a brackish water site influenced by the Saint Croix River estuary); Lime Kiln Bay (station #3 – Letang estuary, where a number of aquaculture sites are located); Deadmans Harbour (station #15 – an open bay with offshore influence); and The Wolves (station #16 – an offshore indicator site). A fifth sampling site was added in mid-Passamaquoddy Bay in 1999, following observations of brick-red patches of water. Results from the monitoring program were published in the following reports: Wildish et al. (1988, 1990), Martin et al. (1995, 1999, 2001b, 2006a, 2014b,c), and Martin and LeGresley (2014).

Phytoplankton monitoring has shown that there is often a time lag between an increase in *Alexandrium catenella* cell numbers and a rise in shellfish STX group toxicity. This provides an early warning (days and, in some cases, weeks) to industry and to the CFIA so they could increase sampling effort in the affected areas (Martin et al. 1995). It has also proven to be advantageous for knowing when to increase shellfish sampling to ensure that harvesting areas are promptly closed upon contamination and then re-opened after the danger has passed. Phytoplankton monitoring also provided an early warning to the salmon aquaculture industry so that mitigation measures could be implemented (Chang et al. 2005, 2006, 2007a,b; Harrison et al. 2007).

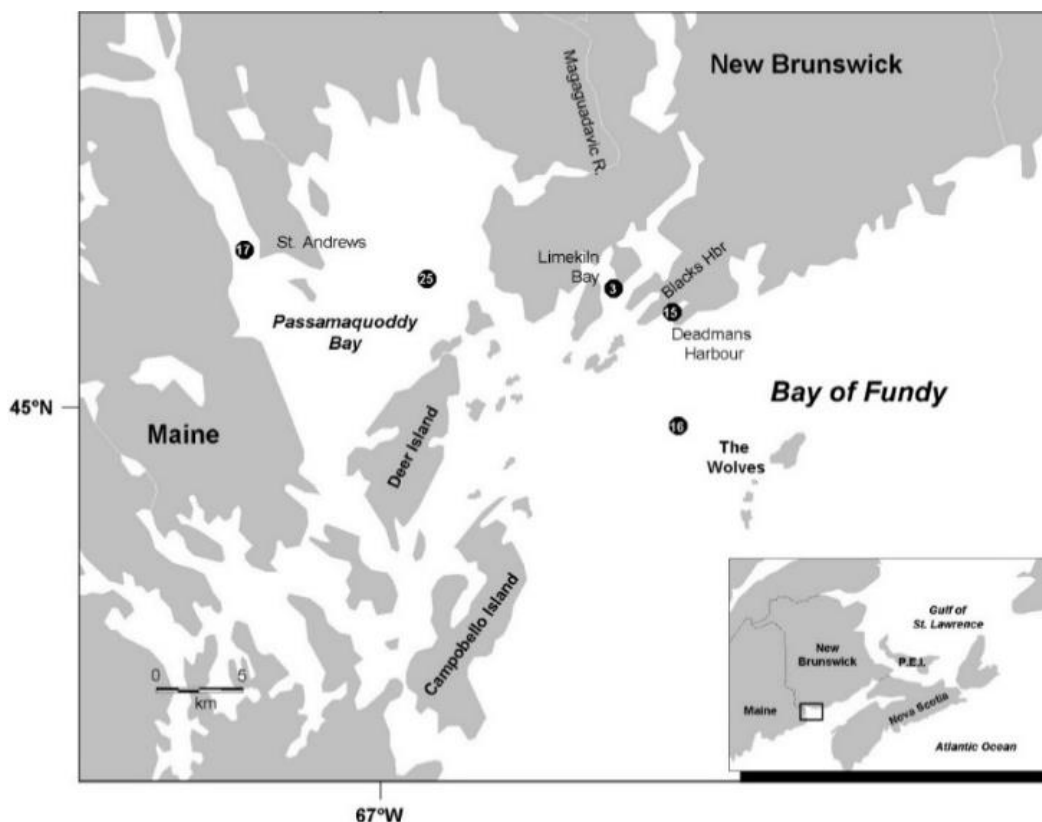


Figure 12. Location of sampling stations in the Bay of Fundy, eastern Canada.

Martin and LeGresley (2008) further showed the importance of a phytoplankton monitoring program when they reported 27 new species to the southwest NB portion of the Bay of Fundy (eight dinoflagellates, 14 diatoms, and five flagellates and smaller zooplankton). Through records in long-term time series, it is possible to determine the presence of indigenous and non-indigenous species, as well as document new species. This allowed the determination that, among other taxa, *Trieres chinensis* (as *Odontella sinensis*), *P. subpacifica*, and *A. pseudogonyaulax* were likely introduced via ballast water (Section 8.1.3). The phytoplankton monitoring program ended in December 2013, prior to the 2014 retirement of the two scientists responsible, although *A. catenella* concentrations were monitored during 2014.

9.2.2.4 Quebec Region of the Gulf of St. Lawrence

In 1949, monitoring for phycotoxins by Canadian federal agencies expanded from the Bay of Fundy to include the St. Lawrence region, as a result of an outbreak in this area in the previous year, when three people succumbed to PSP (van de Riet et al. 2006). The Toxic Algae Monitoring Program (TAMP) was started in the Quebec Region of the Gulf of St. Lawrence in 1989, by DFO's MLI (Bonneau et al. 2002). The program aims to: 1) identify and track short- and long-term toxic or HABs in the Estuary and Gulf of St. Lawrence; and 2) determine the factors responsible for their appearance and development. Its objective is to provide complementary information for the CFIA's shellfish biotoxin monitoring program.

As for the Bay of Fundy, the number of sampling stations has changed since the initiation of the program. From 1989 to 2009, the program consisted of a network of 11 coastal stations ([Fig. 13](#)) sampled on a weekly basis between May and October. Following the *Alexandrium* bloom of 2008, which caused the deaths of 10 beluga whales (Section [2.4.3.1](#)), one additional coastal station was initiated in the habitat of beluga whales (Rivière-du-Loup). In 2010, the number of stations was unfortunately reduced, due to financial constraints, to six stations that continue to be monitored at present on a regular basis (usually three to four times per month). These include: Tadoussac, Sept-Îles, Rivière-du-Loup, Gascons, Penouille, and Havre-Aux-Maisons. On a more irregular basis, the historical stations of Mont-Louis and Carleton are nevertheless occasionally re-visited.

TAMP identifies and counts all potential toxic/harmful algae (including *Alexandrium* spp., *Dinophysis* spp., and *Pseudo-nitzschia* spp.), and collects environmental parameters (temperature, salinity, Secchi depth, and nutrients). This allows a better understanding of the conditions that favour the growth of toxic species and the development of their blooms. The program has proven to be mutually beneficial for monitoring and scientific purposes. It also provides information to DFO's [AZMP](#) in the Quebec Region. The reduction of the program in 2010 resulted in some gaps in knowledge about HABs in some regions previously monitored, notably on the north shore, including the site at Baie-Comeau; this is a region recognized as a “hot spot” for the initiation of *Alexandrium* blooms in the region (Section [2.4.3.2.1](#)).

A detailed analysis of data collected since 1989 by TAMP is currently underway at the MLI, funded by DFO's Aquatic Climate Change Adaptation Services Program ([ACCASP](#)). The objective of this retrospective analysis is to develop empirical models that will be directly coupled to DFO's ocean model of the Gulf of St. Lawrence, Scotian Shelf, and Gulf of Maine. This will allow the simulation and study of the occurrence of HAB events on the Atlantic coast under current and future atmospheric and oceanographic forcings. The objective is to produce risk maps of the occurrence of *Alexandrium* and *Pseudo-nitzschia* blooms for each decade for the next century (climate change scenarios). Such a project cannot be performed without the long-term monitoring of HABs provided by TAMP.

Results from the monitoring program were published in the following technical reports: Blasco et al. (1998), Bonneau et al. (2002), and Lessard et al. (2020). Data from 1994 to 2008 are also available on the St. Lawrence Global Observatory ([SLGO](#)) website.

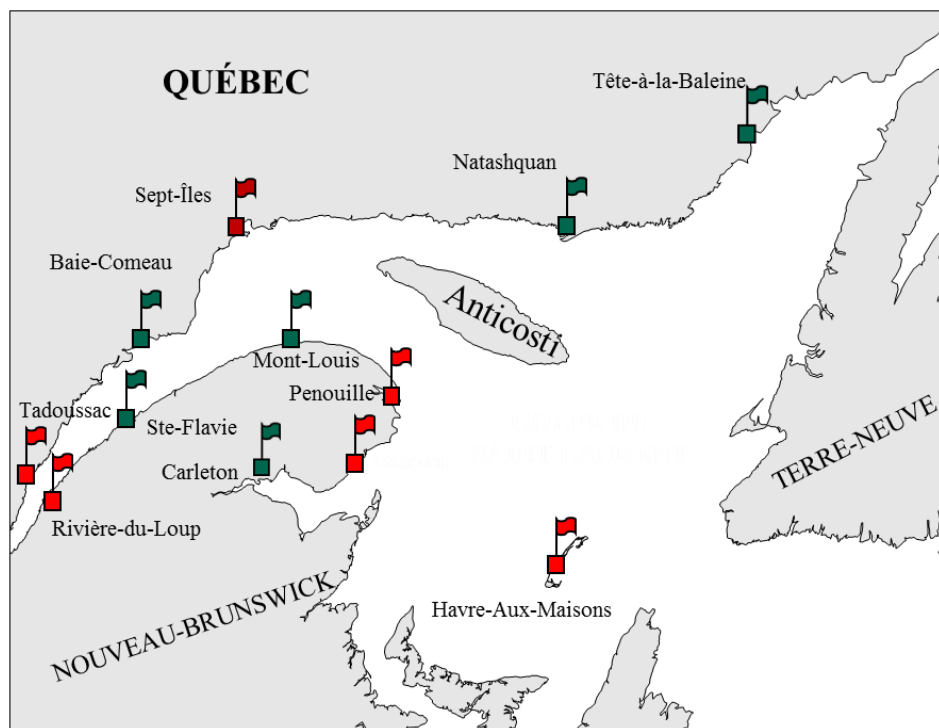


Figure 13. Location of sampling stations (currently active and previously active) of the Toxic Algae Monitoring Program (TAMP) of DFO's Québec Region. Red symbols = Currently active; Green symbols = Previously active/occasionally active

9.2.2.5 Newfoundland and Labrador

In NL, the earliest investigations into biotoxins and the phytoplankton that produced them were collaborations between [Memorial University of Newfoundland](#) and DFO, as there was no official phytoplankton monitoring program (McKenzie 1996). Prior to 1994, detection of the effects of harmful algae on shellfish aquaculture was limited to monitoring toxins in shellfish tissue by the Inspection Branch of DFO. Biotoxin monitoring in molluscan shellfish is now carried out by the CFIA. A phytoplankton monitoring program, with the goal of providing an early warning to the aquaculture industry, was briefly carried out at five mussel aquaculture sites (McKenzie et al. 2003, 2005). Funding was provided by the DFO ACRDP to the [Newfoundland Aquaculture Industry Association](#), the Aquaculture Research Section of DFO, the [Newfoundland and Labrador Department of Fisheries and Aquaculture](#), and the CFIA. Sampling was continued monthly from April to November 2002–2004. NL would currently benefit from a phytoplankton monitoring program.

10.0 Knowledge Gaps and Recommendations for Further Research

Several knowledge gaps remain that require further research. These are outlined, below, with specific recommendations.

10.1 Global warming, ocean acidification and expansion of HAB species

Global warming is of increasing concern with respect to the oceans and cryosphere, including consequences of a decrease in multiyear Arctic ice (IPCC 2019a,b; Thackeraym and Hall. 2019). Given that phytoplankton growth is strongly determined by temperature, light and the availability of nutrients, it is not surprising that changes in these parameters as a result of global warming will also influence the growth and distribution of HAB species (Moore et al. 2008; Trainer et al. 2019) and the appearance of novel phycotoxins (Botana 2016). The IPCC (2009b) report identified “increased frequency in coastal areas since the 1980s of harmful algal blooms, attributed to both climatic and non-climatic drivers, with *high confidence*”. It should be noted, however, that some HAB scientists believe this statement from the IPCC may seem premature (Kudela 2019), given that linking climate change and HABs has previously been identified as a “formidable predictive challenge” (Hallegraeff 2010), and that there is uncertainty about how changes in temperature, ocean acidification, precipitation, nutrient stress or availability, and the physical structure of the water column integrate to shape future HABs (Wells et al. 2020).

Nevertheless, climate change and global eutrophication are contributing to the expanding global footprint of HABs (Glibert 2020). HAB species are expected to move to more northern latitudes as a result of climate change, when ocean temperatures rise, water circulation patterns change, and stratification increases (Hallegraeff 2010, 2016; Poulin et al. 2011; Glibert et al. 2014; Barton et al. 2016; Gobler et al. 2017; Townhill et al. 2018). Statistical models of HABs in a changing climate can be used for near-term HAB forecasting and resource management, but they are not well suited for longer-term projections because of diverges from past observations (Ralston and Moore 2020). An increase in the abundance of diatoms, including toxigenic *Pseudo-nitzschia*, relative to dinoflagellates in the northeast Atlantic may be attributed to climate change (Hinder et al. 2012). However, this diatom increase has not occurred in all parts of the Atlantic, or example, in the Bay of Fundy.

The presence of tetrodotoxin (TTX) in mussels and oysters on the south coast of England (Section [10.3.2](#)), as well as elsewhere in Europe, may be linked to warming surface waters (Turner et al. 2015a; Rodríguez et al. 2017; Clark et al. 2019a). Indeed, these events occurred mostly during summer, when waters were warmer and salinity lower. Furthermore, the growth and TTX production by bacteria of the genus *Vibrio* are strongly dependent on high temperature and low salinity, conditions associated with climate change. This case study illustrates the threat of a new, unexpected, emerging natural marine toxin, traditionally associated with warm tropical waters and tropical fish species, to consumers of shellfish in temperate regions.

Warming ocean temperatures and decreased seasonal ice cover will further expand the window of growth for endemic HAB species already present in arctic waters (Section [10.2](#); [Appendix 1](#)). This range expansion poses a threat to human and ecosystem health in the Arctic (Anderson et al. 2018). The impact will be especially significant in a region where traditional monitoring for phycotoxins is a major challenge because the coastlines are remote and expansive.

Climate change may also lead to extreme events that affect precipitation, temperatures and wind patterns for given regions, resulting in changes in water quality that could favour HABs (Michalak 2018). For example, the unusually warm water of the west coast of North America triggered the wide-spread bloom of toxic *Pseudo-nitzschia* that extended into BC waters (Section 3.3). Increased local precipitation related to climate change could provide terrestrial-based fertilizer that would exacerbate the severity of HABs in regions previously unaffected.

A recent Canadian report on climate change (Bush and Lemmen 2019) has come to the following conclusions that are relevant to this review:

- Canada's climate has warmed and will warm further in the future, driven by human influence.
- Both past and future warming in Canada is, on average, about double the magnitude of global warming. The Canadian Arctic is warming even more quickly, nearly three times the global rate.
- Canadian areas of the Arctic and Atlantic oceans have experienced longer and more widespread sea-ice-free conditions.
- Oceans surrounding Canada have warmed, become more acidic, and are less oxygenated, consistent with observed global ocean changes over the past century.
- The effects of widespread warming are evident in many parts of Canada and are projected to intensify in the future.

Given the above, and as also reflected in Gao et al. (2019a), there is no question that HAB research will have climate as a major focal area for the foreseeable future (Anderson 2014). Hennon and Dyhrman (2020) provide an overview of how certain HAB species may respond to climate change. A recent meta-analysis revealed enhanced growth of marine HAB species from temperate regions characterized by warming and elevated CO₂ levels, although phycotoxin production did not show a consistent response towards these climate change variables (Brandenburg et al. 2019). For example, increased ocean acidification due to higher levels of CO₂ entering ocean waters increases DA production by some diatom species of *Pseudo-nitzschia* (Sun et al. 2011; Fu et al. 2012; Tatters et al. 2012; Wingert 2017). However, there is also contrary evidence (Lundholm et al. 2004; Trimbom et al. 2008). Nevertheless, a long-term mesocosm study (~100 d) in the Gullmar Fjord, Sweden, showed increased levels of DA production by *Pseudo-nitzschia* spp. in the CO₂-enriched mesocosm towards the end of the study (Wohlrab et al. 2020). The dinoflagellate *Alexandrium* shows similar contradictions. For example, *A. catenella* (as *A. fundyense*) from Northport Bay, New York (Hattenrath-Lehmann et al. 2015) and *A. catenella* from Southern California waters (Tatters et al. 2012) produced significantly greater levels of STX group toxins at a higher atmospheric partial pressure of CO₂ (pCO₂). However, *A. catenella* (as *A. fundyense*) from the Bay of Fundy (Hattenrath-Lehmann et al. 2015) showed no changes in toxicity and *A. catenella* (as *A. tamarensis*) from the Scottish North Sea (Van de Waal et al. 2014) contained lower cellular levels of toxins under this condition. Elevated pCO₂ increased STX group toxin production of *A. ostentifidii* from the Baltic Sea, and its growth rate increased at higher temperatures, which may be indicative of increased toxic bloom events in the future (Kremp et al. 2012). Indeed, the ecological impacts of ocean and coastal acidification, including on HABs, are of increasing concern (Goldsmith et al. 2019; Saba et al. 2019). Furthermore, HABs, in combination with acidification and temperature increase, are now considered as co-stressors to coastal ecosystems, including aquaculture operations (Griffith and Gobler 2020).

Based on evidence presented by Kim et al. (2013), Haigh et al. (2015) anticipated that the fish-killing alga *Heterosigma akashiwo* in BC (Section 7.3.1) may gain a competitive advantage under conditions of ocean acidification, making blooms more frequent, which would seriously threaten the salmon aquaculture industry there. Indeed, elevated CO₂, with or without elevated temperature, stimulated *H. akashiwo* growth (Fu et al. 2008). Furthermore, genes involved in cell motility of *H. akashiwo* were significantly changed by both elevated CO₂ and growth rate, suggesting that future ocean conditions could modify swimming behaviour in this species (Hennon et al. 2019). Significant increases in potential growth rates and bloom season length for North American ribotypes of *Margalefidinium polykrikoides* (Section 7.2.4) were observed along the U.S. east coast, attributed in part to global warming (Griffith et al. 2019b). Palynological records show an increase in the abundance of dinoflagellate cysts, including those of *Protoceratium reticulatum* (Section 5.2), in sediments on the east and west coasts of Canada (Mudie et al. 2002). This abundance was associated with the summer warming of sea surface temperatures of up to 5 °C during the early Holocene. The authors implicated the importance of global warming for the historical increase in the frequency of HABs.

Some models include *Dinophysis* as a potential winner in global warming scenarios (Gobler et al. 2017). How other HAB species grow and produce phycotoxins in warmer, more acidic waters requires more research (Fu et al. 2012; Brandenburg et al. 2019; Raven et al. 2020). Interestingly, Brandenburg et al. (2019) found no significant change in growth rate for non-HAB species under conditions of elevated *p*CO₂, perhaps because they possess different carbon concentrating mechanisms. A conclusion of their meta-analysis is therefore that increases in growth rate with more CO₂ may present an additional competitive advantage for HAB species.

Case studies presented by Trainer et al. (2019), including the toxic *Pseudo-nitzschia* bloom on the west coast of North America during the warm water anomaly in 2015 (Section 3.3), illustrate how such extreme weather events can mimic future climate conditions and provide a “dress rehearsal” for understanding the extent, frequency and intensity of future HABs under conditions of climate change.

Many major questions remain about HABs and climate change, as detailed in Wells et al. (2015). For example, they indicate that “there is a critical absence of tenable hypotheses for how climate pressures mechanistically affect HAB species”. An international symposium on HABs and climate change considered new research directions to better define the linkages between the two (Wells et al. 2020). There is a need to conduct community-wide scientific studies, using agreed upon laboratory protocols, in order to generate reliable data for climate change models (Boyd et al. 2013). These studies should focus on keystone species and address intra-strain variability in response to multifactorial conditions that might be expected under climate change. This includes changes in temperature, irradiance, pH/*p*CO₂, stratification, and nutrients. The response of HAB species to climate change becomes even more complicated when interactions between ocean acidification and eutrophication (Flynn et al. 2015), or global warming and UV radiation (Gao et al. 2018, 2019a), are considered. The limited and often conflicting experimental data make it difficult to establish whether there is a link between HABs and climate change, let alone how dramatic a change in HABs might be expected in the future (Wells et al. 2015, 2020; Hallegraeff 2016; Wells and Trainer 2016; Wells and Karlson 2018; Trainer et al. 2019).

10.2 HAB species in the Canadian Arctic

The importance of understanding the distribution, bloom frequency and food web linkages of HAB species in Canadian Arctic waters, including Hudson Bay, is becoming more pressing. This is especially relevant because of increasing access and shipping to arctic waters due to reduction in sea ice (Chan et al. 2018). Current climate trends are resulting in dramatic declines in sea ice, increases in water temperatures and available light for photosynthesis, as well as changes in salinity stratification regimes and local hydrography (Joli et al. 2018; IPCC 2019a,b; Thackeray and Hall 2019). This is likely to expand the northern geographic range and duration of favourable conditions for HABs, making algal toxins a growing concern in Arctic marine food webs (Poulin et al. 2011; Walsh et al. 2011; Lefebvre et al. 2016; Dhifallah 2019).

A fundamental knowledge gap that is starting to be addressed is an understanding of which known or potential HAB species are present in Canadian Arctic waters ([Appendix 1](#)), where almost 70% of species globally considered toxic are already present across various habitats of the Canadian Arctic and adjacent waters (Poulin et al. 2011; Pućko et al. 2019). Species recorded in the Canadian Arctic include: *Alexandrium catenella*, *A. ostenfeldii* (Section [2.4.5](#)), *Pseudo-nitzschia* spp. (Section [3.4](#)), *Dinophysis* spp., *Prorocentrum lima*, *Phalacroma rotundatum* (Section [4.6](#)), *Protoceratium reticulatum*, *Lingulodinium polyedrum*, *Gonyaulax spinifera* (Section [5.2](#)), *Protoperidinium crassipes* (Section [5.3](#)), *Chaetoceros* spp. (Section [7.1.1](#)), *Skeletonema costatum* (Section [7.1.2](#)), *Leptocylindrus minimus* (Section [7.1.4](#)), *Ditylum brightwellii* (Section [7.1.5](#)), *Alexandrium monilatum* (Section [7.2.1](#)), *Gyrodinium aureolum* (Section [7.2.2](#)), *Karenia mikimotoi* (Section [7.2.2](#)), *Coolia monotis* (Section [7.2.5](#)), *Heterosigma akashiwo* (Section [7.3.1](#)), *Chrysochromulina* spp. (Section [7.3.2](#)), *Octactis speculum* (Section [7.3.3.1](#)), *Prorocentrum minimum* (Section [10.3.2](#)), *Phaeodactylum tricornutum*, *Proboscia inermis*, *Conticribra weissflogii*, *Halamphora coffeiformis*, *Heterocapsa triquetra*, and *Aphanizomenon flos-aquae* (Section [10.3.3](#)). Investigation of the presence of additional HAB species in the Arctic requires more attention.

There is great opportunity to continue addressing these issues with the announcement that DFO is creating a new Arctic Region over the next several years. Its purpose is “to improve program and service delivery in the North, to strengthen the development and implementation of the Arctic and Northern Policy Framework, and to strengthen delivery on other key priorities such as reconciliation with Indigenous Peoples and Canada’s Oceans Protection Plan”.

10.3 Emerging potential marine toxins of concern

10.3.1 *Microcystis* and microcystins in marine coastal waters

Certain species of cyanobacteria (blue-green algae), including *Microcystis* spp. (Harke et al. 2016), produce powerful toxins, one of which is the hepatotoxic peptide microcystin-LR (MC-LR) (Carmichael 1994; Vareli et al. 2013; Huang and Zimba 2019). Microcystin toxins are usually associated with freshwaters ([Appendix 2](#)), but there are increasing incidences of their occurrence in estuarine and coastal waters worldwide (O’Neil et al. 2012; Harke et al. 2016; Preece et al. 2017; Peacock et al. 2018). As mentioned above (Section [4.3](#)), MC-LR is an inhibitor of protein phosphatase enzymes; it could therefore hasten tumour growth by contributing to the unchecked division of cells (Carmichael 1994). It should be pointed out that OA group toxins are also inhibitors of protein phosphatases (Luu et al. 1993). As such, they are also tumour promoters, at least when large doses

are applied to mammalian cell cultures. The effects due to chronic low-level exposure are not known. Reviews are available on the chemistry, toxicology and genetics of cyanobacterial toxins (Pearson et al. 2010), the environmental implications and human health risks of microcystins (Fessard 2014; Preece et al. 2017), the global distribution of cyanotoxins and cyanobacterial poisonings (Svirčev et al. 2019), and on solutions for managing cyanobacterial blooms (Burford et al. 2019).

Although there are reports of harmful marine cyanobacterial blooms in Puget Sound (WA), there have been no accounts of toxic marine cyanobacteria in Canadian waters (Preece et al. 2017). Nevertheless, Canadian studies in the early 1990s have led to some pertinent discoveries about the presence of microcystins and OA group toxins (Boland et al. 1993). Chen et al. (1993) thus detected MC-LR, a toxin most often associated with the cyanobacterium *Microcystis*, in mussels collected in 1991 from New London Bay (northern PE) and from Vancouver Island (BC). These same PE mussels also contained the OA group toxin DTX1 (Section 4.1), according to the phosphatase inhibition assay (Holmes 1991). Additional research is required to compare this assay with other analytical techniques in order to remove any remaining uncertainties about the identity of the toxins reported by the assay.

MC-LR has been implicated as the cause of netpen liver disease (Andersen et al. 1993), which has been a major problem for farmed salmon in Pacific waters since at least the mid-1980s (Kent 1990). The source of the MC-LR in the fish was traced to their food, i.e. crab larvae and copepods. The ultimate source of the toxin has not yet been determined, although the study by Miller et al. (2010) (see below) suggests that it could be another example of freshwater to marine transfer of microcystins. Netpen liver disease looks very similar to hepatic megalocytosis, whose cause is unknown but may be linked to MC-LR (Stephen et al. 1993). It should be pointed out that the presence of microcystins in intoxicated salmon has not yet been definitively confirmed using modern analytical methods (e.g. LC-MS/MS), and other questions remain about the link between MC-LR and netpen liver disease (Smith 2015).

Although not a Canadian example, the following demonstrates a potential risk that has been largely overlooked in Canadian waters. Microcystins were implicated as the cause of the mortality of wild southern sea otters (*Enhydra lutris nereis*) in Monterey Bay, CA, in 2007 (Miller et al. 2010). Necropsied otters had hepatic lesions suggestive of acute liver failure and biochemical testing (using LC-MS/MS) confirmed that microcystins were present in the livers. Carcasses of otters, which had died because of microcystin intoxication, were mainly found near river mouths, coastal ponds, embayments and harbours. Various forms of microcystins were detected. It was demonstrated by environmental sampling using SPATT collectors at the freshwater-marine interfaces of rivers draining into Monterey Bay that there was transfer of microcystins from freshwater cyanobacteria growing in lakes to marine waters during the rainy season. Experiments showed that the freshwater microcystins were taken up and bioconcentrated by marine invertebrates (clams, mussels, oysters, and snails, but not crabs), which were then eaten by the otters. The microcystins could persist for at least 21 days in seawater. This study confirmed the existence of a novel class of marine “HAB” in the Pacific coastal environment, i.e. hepatotoxic shellfish poisoning (HSP). Miller et al. (2010) suggested that animals and humans are at risk from microcystin poisoning if shellfish harvested at the land-sea interface are consumed.

Cyanotoxins are analyzed by LC-MS/MS (Hollingdale et al. 2015; Yilmaz et al. 2019), although commercial microcystin test kits are available and have been evaluated (Watson et al. 2017). Increasingly, genetic tools that allow for the rapid and sensitive screening of the genes responsible

for microcystin biosynthesis in the environment are being used in research and routine monitoring (Crawford et al. 2017). The Biotoxin Metrology group of the NRC, in Halifax (NS), produces certified calibration solutions for the determination of cyanobacterial toxins ([NRC Certified Reference Materials](#); [Wikipedia](#); Quilliam 2006; Hollingdale et al. 2015).

10.3.2 Tetrodotoxin

Tetrodotoxin (TTX) is a hydrophilic, heat-stable lethal neurotoxin that selectively binds and blocks voltage-gated sodium channels (Cestèle and Catterall 2000; Bane et al. 2014; Jal and Khora 2015; Lago et al. 2015; Biessy et al. 2019), and is of medical interest as an anesthetic and analgesic drug (Melnikova et al. 2018). It was named after the pufferfish family Tetraodontidae and is the causative agent in pufferfish (*fugu*) poisoning. TTX is also found in newts, horseshoe and xanthid crabs, frogs, edible marine snails, sea slugs, star fishes, octopuses, and ribbon worms (Chau et al. 2011; Jal and Khora 2015; Zhang et al. 2015; Mifsud et al. 2019; Whitelaw et al. 2019), suggesting a common exogenous (i.e. via ingestion) biological origin. For example in pufferfish, it is believed to be symbiotic bacteria of the genus *Vibrio*, although many other genera, including *Pseudomonas*, are reported to be TTX producers (reviewed in Chau et al. 2011; Bane et al. 2014; Jal and Khora 2015). In some species, TTX may serve as a defensive function (Whitelaw et al. 2019).

Despite having a toxicity similar to that of STX (Finch et al. 2018), no maximum level is defined for TTX, nor its 30 naturally occurring analogues (Biessy et al. 2019), in seafood in the EU (Bane et al. 2014; Knutsen et al. 2017) or Canada. Now, however, the presence of TTX in shellfish is regulated in one country, the Netherlands, with a regulatory level of 44 mg kg⁻¹, as suggested by the the EFSA (Gerssen et al. 2018). Establishing a regulatory level is important because TTX has been reported in 21 species of bivalves and edible gastropods from 10 countries since the 1980s (Biessy et al. 2019), e.g. scallops (*Patinopecten yessoensis*) in Japan (Kodama et al. 1996), pipi clams (*Paphies australis*) in New Zealand (Biessy et al. 2018), Manila clams (*Ruditapes philippinarum*) in China (Zhang et al. 2015), mussels (*M. edulis*) and oysters (*Crassostrea gigas*) on the south coast of England (Turner et al. 2015a), mussels (*Mytilus galloprovincialis*) in Greece (Vlamiš et al. 2015), and mussels and oysters in the Netherlands (Knutsen et al. 2017; Gerssen et al. 2018).

The biological source of TTX in bivalve molluscs remains uncertain (Biessy et al. 2019) and there is contradictory evidence regarding whether the source is exogenous or endogenous (Jal and Khora 2015; Biessy et al. 2018). As mentioned above, bacteria are believed to be the biological source of TTX. However, two reports suggest an association between the presence of TTX in bivalves and the occurrence of dinoflagellates. First, Kodama et al. (1996) reported a significant amount of TTX in cultured cells of *Alexandrium catenella* (as *A. tamarense*) from Japan, a species that also produces STX group toxins (Section 2.2). This finding supports their previous observation that *A. catenella* was the origin of TTX detected in highly toxic scallops (*Patinopecten yessoensis*) during a bloom of *A. catenella*. Thus far, however, no other studies have confirmed TTX production by this dinoflagellate. Second, Vlamiš et al. (2015) detected TTX in mussels (*Mytilus galloprovincialis*) in Greece, coinciding with blooms of the dinoflagellate *Prorocentrum minimum*, which gave a potential link to this algal species. Then, Rodríguez et al. (2017) reported that two out of three strains of *P. minimum* (those from Ecuador and Florida) produced compounds that showed high activity as inhibitors of sodium channels, in the same way as does TTX. Furthermore, characterization of those compounds by LC–MS/MS showed for the first time that they were “TTX-like” compounds with a similar ion pattern and C9-base to TTX analogues. *Prorocentrum*

minimum was previously associated with Venerupin Shellfish Poisoning (VSP) in Japan and similar poisonings elsewhere (references in Vlamis et al. 2015). Earlier, Grzebyk et al. (1997) reported that extracts of four *P. minimum* strains from the French Mediterranean Sea produced a water-soluble neurotoxic component that rapidly killed mice, although the symptoms were different than that produced by VSP. Similar results were reported by Denardou-Queneherve et al. (1999). Other strains of *P. minimum* were toxic to a number of bivalves, although a toxin was not directly quantified or identified (references in Heil et al. 2005; Saba et al. 2011).

Thus far, *P. minimum* has been found at high enough concentrations to cause water discolouration in Bocabec Bay (Bay of Fundy, southwest NB) in the 1990s (J.L. Martin unpubl. data), and was reported by Martin et al. (2001b). It is also present in the Gulf of St. Lawrence (Lessard et al. 2020), coastal NL (C.H. McKenzie unpubl. data), as well as in the eastern and western Arctic, the Canadian Archipelago and the Hudson Bay system (Simard et al. 1996; Harvey et al. 1997; Lovejoy et al. 2002; McLaughlin et al. 2009; Poulin et al. 2011; Simo-Matchim et al. 2017; Dhifallah 2019; M. Poulin unpubl. data) ([Appendix 1](#)). It has also been reported as being widely distributed along the U.S. central and northeast coast, including Maine (Hargraves and Maranda 2002; Heil et al. 2005). Because of the potential presence of TTX in Canadian bivalves, the NRC and the CFIA are collaborating to establish analytical methods for its analysis. To this end, in order to address current and future method needs in Canada and abroad, the NRC is also developing calibration solutions and pilot-scale matrix reference materials for analysis of TTX in seafood (McCarron et al. 2018). Furthermore, monitoring programs should be vigilant for the presence of potentially toxigenic *P. minimum*.

10.3.3 β -N-methylamino-L-alanine (BMAA)

In the last 30 years, several environmental studies have pointed to the potential role of dietary exposure to the neurotoxic non-protein amino acid, β -N-methylamino-L-alanine (BMAA), as a possible risk factor for the progressive neurodegenerative diseases Amyotrophic Lateral Sclerosis (ALS; also known as Lou Gehrig's disease), Alzheimer's dementia and Parkinson's disease (reviewed in Regueiro et al. 2017). The association was first hypothesized as one of several possible causes for the widespread observation of neurodegenerative disease in the Chamorro people of Guam, who consumed seeds of the cycad plant (Vega and Bell 1967).

BMAA suppresses cell cycle progression in mouse fibroblast cells involved in collagen synthesis of connective tissue (Dunlop et al. 2013; Okamoto et al. 2018), inhibits the activity of certain enzymes, and interferes with protein folding (van Onselen and Downing 2018). Similar to DA, BMAA acts as an agonist of glutamate receptors (Manzoni et al. 1991), although it is not very potent as an excitatory neurotoxin. This mismatch between the acute toxicity of BMAA and the severity of symptoms observed in Guam was one of several reasons the hypothesis did not gain widespread acceptance. Decades later, the topic was revisited with a series of publications proposing that: 1) BMAA is produced by a cyanobacterium (*Nostoc*) living symbiotically in the roots of the cycad, rather than by the plant itself; 2) BMAA could be biomagnified through food webs, leading to much higher concentrations in bats consumed as a food source than originally measured in the cycad; 3) this biomagnification was due to misincorporation of BMAA into protein in the place of L-serine; 4) BMAA could be detected in the brains of deceased ALS patients from Canada; and 5) BMAA production by cyanobacteria was widespread throughout the phylum (Cox et al. 2003, 2005; Murch et al. 2004). Because cyanobacteria are widespread throughout the

aquatic environment, this work suggested that BMAA could potentially be a significant global public health risk, including in food supplements based on cyanobacteria (Manolidi et al. 2019).

There has been much controversy in the literature surrounding the “BMAA hypothesis” as described above (Nunn 2017). Of particular note is the fact that measurements of BMAA supporting this work (Cox et al. 2003, 2005; Murch et al. 2004) were done using poorly selective LC–FLD methods and have never been replicated with modern LC–MS/MS methods (Faassen et al. 2012; Krüger et al. 2012; Faassen 2014; Rutkowska et al. 2019). Subsequent attempts to detect BMAA in brain samples from patients who died of neurodegenerative disease using modern LC–MS/MS methods have been unsuccessful (Combes et al. 2014; Meneely et al. 2016) and the elevated levels (high mg kg⁻¹) of BMAA originally reported in cyanobacteria (Cox et al. 2005) have never been detected since. Similarly, a recent attempt to confirm high levels of BMAA in the same museum specimens of bats from Guam originally analyzed did not detect BMAA at all (Foss et al. 2018). Research has also pointed out that cycad plants devoid of cyanobacteria still produce high levels of BMAA (Marler et al. 2010), but the biosynthetic origin of BMAA in other organisms remains unknown. Furthermore, in mice and rats administered BMAA, this toxin was distributed in all tissues examined, and there was no greater affinity in the brain than in any other organs or tissues (Waidyanatha et al. 2018).

Since the challenge of accurately measuring BMAA in complex samples is central to much of this controversy, a great deal of work has been done to improve analytical methods. The two most common approaches for sensitive and selective analysis of BMAA both use LC–MS/MS and involve either analysis of underivatized BMAA using hydrophilic interaction liquid chromatography (HILIC) (Rosén and Hellenäs 2008; Li et al. 2012; Réveillon et al. 2014) or analysis of derivatized BMAA with reverse phase LC (Jiang et al. 2012, 2013; Glover et al. 2015). Capillary electrophoresis (CE) coupled with MS/MS was also developed as an alternative method to quantify BMAA and applied to a small subset of Canadian seafood samples where BMAA was detected (Kerrin et al. 2017). The literature reveals a wide range of BMAA concentrations in the same organism, when different studies are considered, particularly with respect to cyanobacterial samples. This could possibly be explained by: 1) the different analytical technique used (i.e. HILIC versus HPLC); 2) differences in sample analyzed or conditions of cyanobacterial culture; and 3) the complex speciation of BMAA in biological samples between various free and bound forms. Faassen et al. (2016) evaluated different LC–MS/MS methods for BMAA analysis, and defined “bound” versus “free” forms of BMAA operationally, based on their presence in the solvent or precipitate fraction. The chemical speciation of these fractions in positive samples also remains unknown, as demonstrated by Beach et al. (2018a), who showed that release of bound BMAA from shellfish samples was not correlated to the hydrolytic release of protein amino acids. This suggests that BMAA detected in shellfish samples may not be incorporated into protein and raises further questions about its origin and significance. Glover et al. (2012) hypothesized that the reactivity of BMAA with metal ions in the sample matrix and the formation of metal adducts in electrospray ionization MS analysis confound the results. Beach et al. (2015, 2018a) examined the analytical procedures used, in order to explain the inconsistent results, and to determine how to separate the BMAA isomers. One outcome of their study was to analyze a variety of samples previously reported to contain BMAA and isomers, using the multidimensional hydrophilic-interaction liquid chromatography-differential mobility spectrometry-tandem mass spectrometry (HILIC-DMS–MS/MS) method. Interestingly, the NRC mussel tissue reference material for DA (CRM-ASP-Mus) contained BMAA and three isomers. Thus, the commercially available CRM-

ASP-Mus can now be used for comparison of BMAA detection methods, as demonstrated by Foss et al. (2018). BMAA was also detected in archived mussel samples from Cardigan Bay (PE) used to prepare the CRM-ASP-Mus. The question of the origin of the BMAA in these mussel samples still remains.

While controversy still surrounds the links between cyanobacteria and BMAA production, and BMAA and neurodegenerative disease, research using modern MS/MS-based methods has now consistently detected BMAA in seafood worldwide, regardless of the analytical technique used, particularly in bivalve shellfish. BMAA has been reported to exist in the free form, or bound with the protein of marine organisms consumed by humans (Brand et al. 2010; Jiang et al. 2014a). For example, BMAA has been found in various shellfish, including blue mussels (*M. edulis*) and oysters (*Ostrea edulis*) from the Kattegat Sea and the Swedish coast, mussels (*M. galloprovincialis*) and oysters (*Crassostrea gigas*) from southern France, mussels (*Utterbackia imbecillis*), oysters (*C. virginica*, *Pinctada margaritifera*) and blue crab (*Callinectes sapidus*) from Florida and the Chesapeake Bay, cockles (*Cerastoderma edule*) from Portugal, spiny lobster (*Panulirus* sp.) from Florida (Brand et al. 2010; Jonasson et al. 2010; Christensen et al. 2012; Field et al. 2013; Masseret et al. 2013; Banack et al. 2014; Jiang et al. 2014a; Lage et al. 2014; Réveillon et al. 2014, 2015, 2016a; Rosén et al. 2016), as well as in 48 mollusc species from China (Li et al. 2016b, 2018b). Several fish consumed by humans, including smelt (*Osmerus eperlanus*), turbot (*Scophthalmus maximus*), and common whitefish (*Coregonus lavaretus*) (but not Atlantic salmon; *Salmo salar*) from the Baltic Sea also contained BMAA (Jonasson et al. 2010). Zooplankton from these waters also contained BMAA, suggesting that they were the vectors, and illustrating the bioaccumulation of this toxin (Jonasson et al. 2010). Interestingly, Berntzon et al. (2015) detected BMAA in the cerebrospinal fluid of humans potentially exposed to Baltic Sea BMAA-“contaminated” food items (fish, shellfish). More recently, a detailed food web study in the Baltic Sea concluded that there was insufficient evidence to support BMAA bioaccumulation in the food webs studied (Zguna et al. 2019). Elsewhere, an epidemiological study found a significant ALS cluster surrounding the Thau lagoon (southern France), where mussels (*Mytilus galloprovincialis*) and oysters (*Crassostrea gigas*) that contained BMAA were consumed (Masseret et al. 2013).

BMAA has also been detected in the fins of five species of shark collected from South Florida waters (Mondo et al. 2012), suggesting that consumption of shark fins may increase the risk for human exposure to this cyanobacterial neurotoxin. Finally, high levels of BMAA were recently found in the brains of 13 of 14 stranded dolphins from Florida and Massachusetts (Davis et al. 2019). Those brains showed neuropathological changes characteristic of ALS, the human neurodegenerative disease. The high levels of BMAA reported in these studies (Mondo et al. 2012; Davis et al. 2019) should, however, be regarded as suspect since the authors chose to use poorly selective LC-FLD methods for quantitation and only used modern LC-MS/MS methods for qualitative confirmation of BMAA identity.

The number of types of phytoplankton found to contain this neurotoxin has also expanded to include marine diatoms (*Achnanthes* sp., *Chaetoceros calcitrans*, *Navicula pelliculosa*, *Phaeodactylum tricornutum*, *Proboscia inermis*, *Skeletonema marinoi*, *Thalassiosira pseudonana*, *Conticribra weissflogii* (formerly *Thalassiosira weissflogii*), *Halamphora coffeiformis*) (Jiang et al. 2014b; Lage et al. 2016, 2019; Réveillon et al. 2016b), as well as dinoflagellates (*Heterocapsa triquetra*, *Gymnodinium catenatum*) (Jiang and Ilag 2014; Lage et al. 2014). No BMAA was detected in *Pseudo-nitzschia delicatissima* (Réveillon et al. 2016b). BMAA was detected in natural

populations of the cyanobacteria *Aphanizomenon* spp. and *Nodularia* spp. from the Baltic Sea (Jonasson et al. 2010). Because the cyanobacteria *Spirulina* and *Aphanizomenon flos-aquae* are frequently consumed via dietary supplements, the presence of BMAA in these products may have public health implications (Roy-Lachapelle et al. 2017), including in Canada (Glover et al. 2015). This led the NRC laboratory in Halifax (NS) to test commercial products for the presence of BMAA, using two complementary LC–MS/MS methods; none were above the limit of detection in the small number of samples tested (McCarron et al. 2014b). However, BMAA was detected in two samples of *A. flos-aquae*. (Roy-Lachapelle et al. 2017), as well as in Canadian natural health products containing *Spirulina* (Glover et al. 2015).

Phaeodactylum tricornutum has been reported in the sea ice from the Canada Basin (Melnikov et al. 2002). *Proboscia inermis* has been reported in the phytoplankton from the eastern Arctic and the Hudson Bay system (Grøntved and Seidenfaden 1938; Seidenfaden 1947; Hsiao 1983). *Conticribra weissflogii* has been reported in the phytoplankton from the eastern Arctic and the Canadian Archipelago (Sekerak et al. 1979; Hsiao 1983). *Halamphora coffeiformis* has been reported in the sea ice and the phytoplankton from the eastern Arctic (Grainger and Hsiao 1982). *Heterocapsa triquetra* has been reported in the phytoplankton from the eastern and western Arctic, the Canadian Archipelago and the Hudson Bay system (Grøntved and Seidenfaden 1938; Bursa 1961a; MacLaren Atlantic Limited 1978; Anderson et al. 1981; Hsiao 1983; Lovejoy et al. 2002; M. Poulin unpubl. data). *Aphanizomenon flos-aquae* has been reported in the phytoplankton from the Hudson Bay system (Bursa 1961a) ([Appendix 1](#)).

A comprehensive review of the toxicological and medical basis for BMAA as the causative agent in neurodegenerative disease (Chernoff et al. 2017), as well as expert appraisal on the acute and chronic toxicity of BMAA (Arnich et al. 2017), conclude that the assumption that BMAA causes these diseases is not supported by scientific evidence. However, given the apparent widespread occurrence of BMAA in marine ecosystems and the possible pathways to expose humans and marine organisms (Li et al. 2016b, 2018b), further basic research is required to determine its significance in Canadian shellfish, including determining its source. However, as Jiang et al. (2014a) concluded, “caution and vigilance must be exercised without causing alarm.”

10.4 Whale mortalities

The deaths of some of the 18 North Atlantic right whales (*Eubalaena glacialis*) in 2017, mostly in the Gulf of St. Lawrence (Cabana 2018), were attributed to ship strikes and entanglement with fishing gear (Daoust et al. 2017; Stokstad 2017). This endangered species ([North Atlantic Right Whale DFO website](#)) is believed to have a population of <450 individuals (Stokstad 2017; Cabana 2018). Mortality of some right whales has previously been attributed to STX group toxins (Section [2.4.1.1](#)). As well, DA was found in Bay of Fundy right whale fecal samples and their food, the copepod *Calanus finmarchicus*, suggesting that this toxin may contribute to the failed recovery of the *E. glacialis* population (Leandro et al. 2010a,b). DA was also found in fecal samples of the Southern right whale (*Eubalaena australis*), in Argentina (D’Agostino et al. 2017). However, no biotoxins (DA, STX group toxins, OA group toxins) were reported in samples of whales from the 2017 incident (W.A. Rourke unpubl. data). Nor were STX group toxins found in copepod samples collected concurrently with the whale mortalities (Scarratt et al. 2017). Nevertheless, this possibility should always be considered when necropsies are carried out on dead whales. The failure to detect

toxins in partially decomposed whale tissues may be of limited value in ruling out toxins as a cause of death, as there is little knowledge of the persistence of toxins in whale tissues after death.

10.5 Recommendations

Below are recommendations for further research on Canadian marine HABs and phycotoxins, for consideration by DFO and other scientists. Some of these, plus other recommendations, are found in CSAS reports (DFO 2020; McKenzie et al. 2020a).

Strengthen capability for monitoring and predicting HABs:

- Provide support to build and maintain phytoplankton taxonomic expertise, so that researchers can couple traditional morphological knowledge with advanced molecular tools, in order to improve our understanding of the diversity of HAB species.
- Establish cultures of toxic and harmful phytoplankton for taxonomic and physiological research, as a source of phycotoxin extraction for NRC Certified Reference Materials program, and for use as reference material in phytoplankton taxonomy, in order to provide information on factors triggering phycotoxin production and for predicting HABs.
- Develop phytoplankton monitoring programs for toxigenic species identification and enumeration in coastal regions of Canada where they have been discontinued (Bay of Fundy, NS, BC – except for the salmon aquaculture industry), and strengthen existing phytoplankton monitoring efforts in other coastal provinces; it should be noted that NL has never had a phytoplankton monitoring program.
- Canadian phytoplankton monitoring programs should be alert for the presence of the dinoflagellates *Karenia selliformis*, *Prorocentrum minimum*, and the possibility of GYM production by *Alexandrium ostenfeldii*.
- Supplement traditional light and electron microscopy approaches with high-throughput sequencing (HTS) technologies to allow molecular approaches, including microarray, real-time quantitative PCR (qPCR), and metagenetic (metabarcoding) tools, for routine monitoring of HAB species in Canada (cf. Kudela et al. 2010; Danovaro et al. 2016).
- Investigate implementation of automated *in situ* monitoring techniques for harmful algae monitoring, e.g. moored Environmental Sample Processors (Bowers et al. 2016, 2017, 2018), the Imaging FlowCytobot ([McClane Research Laboratories, Inc.](#)), portable imaging flow cytometers (Göröcs et al. 2018), or the MolluScan eye for signalling behavioural anomalies in molluscan bivalve sentinels (Andrade et al. 2016).
- Optimize field-deployable phycotoxin monitoring systems/kits.
- Further develop models to predict HABs as an early warning and to mitigate their impacts, using knowledge of phytoplankton physiology, triggers of toxin production, oceanographic parameters, and satellite imagery.
- Further explore and refine sea-surface colour information obtained from drones, aircraft, and satellites, as a method to detect HABs and to optimize *in situ* monitoring (e.g. Sathyendranath et al. 1997, 2004; Harrison et al. 2007; McGillicuddy et al. 2014; Devred et al. 2018; Torres Palenzuela et al. 2019).
- Strengthen lines of communication regarding HABs between researchers within Canada and with those sharing waterways in common with the U.S. on the east and west coasts (e.g. Barth et al. 2019).
- Expand citizen science programs to sample for HAB species.

- Establish a procedure that could allow easy cross-referencing and synchronization of DFO Prohibition Orders with CFIA biotoxin data as part of the CSSP, specifically by specifying the phycotoxin responsible for the enacted closure. This would allow more complete and accurate information to be entered into the HAEDAT system, which would increase our understanding of HABs in Canada. Currently, this can only be done manually, which can lead to gaps and errors in the HAEDAT database.

Fill knowledge gaps:

- Research is required on the physiology of toxigenic phytoplankton, as well as on the presence and cycling of phycotoxins in Canadian marine food webs.
- Research is required on the kinetics of phycotoxin uptake and depuration by different age groups of bivalve molluscs.
- Research is required on the symptoms of intoxication by PnTXs. It is not known if the source dinoflagellate, *Vulcanodinium rugosum*, is present in Canadian waters, nor whether other species are capable of producing PnTXs.
- The biological source of AZAs in Canadian waters must be identified. Research is needed to fully understand the variability of AZA production and the presence of novel AZA varieties in different dinoflagellate species, as well as in shellfish. Nothing is known about the biosynthesis of AZAs.
- Strains of *Alexandrium ostenfeldii* from the Gulf of St. Lawrence and western Canada should be tested for the ability to produce STX group toxins.
- Given the presence of OA group toxins in BC waters, especially during 2012 to 2016, the source organisms of these toxins should be verified.
- Given the current uncertainty surrounding the chemical speciation of BMAA detected in shellfish, more basic research is required to determine its significance in Canadian shellfish, including determining its source organism(s).
- Identify the origin, nature, and fate of microcystins in the Canadian marine environment.
- Given the increased prevalence of cyanobacterial blooms in Canadian freshwaters and the potential for their discharge into marine environments, research is needed to investigate the occurrence or future risk of cyanotoxins in commercially harvested shellfish (Miller et al. 2010).
- Determine if the mortality of farmed salmon in the presence of toxic *Alexandrium catenella* cells is due to soluble STX group toxins released during the bloom, or to the synergistic interaction between reactive oxygen species (superoxide anion) and certain polyunsaturated fatty acids produced by *A. catenella* (Mardones et al. 2015).
- Conditions that promote blooms of *Mesodinium rubrum* in the Bay of Fundy remain elusive and this requires research.
- The significance of the dinoflagellate parasite, *Hematodinium* sp., to the snow crab fishery in NL requires research.
- Given that a toxic compound was isolated from waters of a 1988 Scandinavian bloom of *Chrysochromulina polylepis*, and that this species was associated with the mortality of farmed salmon in BC, it is important to characterize toxins produced in culture.
- The role of oomycete and chytrid parasites that infect HAB species requires more study in order to better understand bloom dynamics and toxin production.
- Given the recent decline in ground fisheries, intensification in research on the possible impacts of phycotoxins on fish survival and recruitment is warranted.

- Investigate the impact of sublethal, cumulative effects of phycotoxins on marine organisms.

Arctic research and global warming:

- Given the concern about introductions of exotic species into the Canadian Arctic as global warming opens up these waterways, the presence of potentially toxigenic algal species should be monitored, and studies carried out to better understand the potential for the dispersal and survival of HAB species in this region.
- Given that toxic algal species are found throughout the Canadian Arctic, that conditions favourable for bloom occurrence are anticipated to be more frequent (Walsh et al. 2011; Bush and Lemmen 2019), and that phycotoxins accumulate at lower and higher trophic levels (Lefebvre et al. 2016), potentially even more so at colder temperatures due to low depuration rates, it is essential to study and monitor the occurrence and potential toxicity of HABs in the Arctic, and the movement of phycotoxins within that food web.
- Given that potentially toxic *Dinophysis* spp. are present in Canadian Arctic waters (Poulin et al. 2011), they should be tested for their ability to produce OA group toxins.
- Given the substantial climate-driven changes in coastal oceanographic and ecological systems, accelerated efforts across disciplines should be directed at how climate change will influence the future of HABs (Wells et al. 2020).
- Additional HAB species should be studied to fill knowledge gaps regarding the effects of warming and $p\text{CO}_2$ (both separately and together) on phycotoxin production, and about the metabolic pathways underlying these changes (Brandenburg et al. 2019).

Ballast water:

- Given that ballast water inoculates Canadian waters with non-indigenous toxic/harmful algae, it is essential to: document the phytoplankton already present in Canadian waters; identify harmful algae that have the potential to be introduced via ballast water; and study the conditions that would favour the development of the harmful blooms.

11.0 Summary and Conclusions

From the list of marine phycotoxins and toxic/harmful algae shown in Tables 1–4, it is evident that Canada, like many other countries around the world, is vulnerable to a multitude of harmful algal events. This list has grown considerably over the last decades. Until 1982, the only major marine algal genus of concern to Canada was *Alexandrium*, some species of which produced STX group toxins in the Bay of Fundy, the northern Gulf of St. Lawrence and BC. Today, this dinoflagellate is also found in waters off NL, northeastern NB, and east-central NS, where these toxins have become problematic. Now, all coastal regions of Canada have been impacted by recent discoveries of novel toxic algae and phycotoxins. NL, which until 1982 had been free of reported phycotoxins, is now faced with STX group toxins, OA group toxins, DA, as well as the more recently discovered SPXs, PnTXs, GYM, PTXs, YTXs, and AZAs. These phycotoxins have now also been detected in the Gulf of St. Lawrence, NB, PE, and NS. The southeastern Gulf of St. Lawrence has also experienced the potentially problematic algal species *Chrysochromulina polylepis* (synonym = *Prymnesium polylepis*) and *Gyrodinium aureolum*. The finding of high concentrations of DA on Georges and Browns banks temporarily shut down the development of a roe-on scallop industry. Reports of harmful *Chaetoceros* species in the Bay of Fundy and in NL are of concern to the salmon aquaculture industry. BC, where

historically only STX group toxins (produced by *Alexandrium catenella*) were a problem, now experiences threats by DA, OA group toxins, MC-LR, SPXs, PnTXs, GYMs, PTXs, YTXs (but not yet AZAs), and the algal species *Heterosigma akashiwo*, *Chaetoceros* spp., *Chrysochromulina polylepis*, *Gyrodinium aureolum*, *Gonyaulax spinifera*, and *Noctiluca scintillans*.

So far, the newly discovered phycotoxins (SPXs, PnTXs, GYMs, PTXs, YTXs, AZAs) have been reported only at low concentrations, and little is known of their potential harm. Some of the phytoplankton species that produce them have likewise been found only at low numbers and have not posed an immediate danger. The identity of other potentially harmful species (e.g. *Chrysochromulina polylepis*) must be confirmed. However, conditions can change rapidly, as has already occurred several times in the past, permitting these organisms to bloom to dangerous levels. Recent events, such as the extensive DA-producing blooms associated with the anomalously warm water “blob” off BC and the Pacific coast of the U.S. in 2015, clearly demonstrated this potential. The situation is exacerbated by the dispersal of toxic/harmful algae due to ships’ ballast water and global warming. Due to the scale of the problem, there is therefore a requirement for sustained research and monitoring programs to be incorporated into a national plan. Such a national program on marine phycotoxins and harmful algae has been developed in the U.S. (Anderson et al. 1993; Anderson 1995). Lessons learned from the DA events in PE in 1987, and along the North American west coast in 2015 (Ekstrom et al. 2020; Moore et al. 2020), will guide responses for future novel HABs in Canada.

In Canada, the Phycotoxins Working Group was formerly a national advisory and management body reporting to DFO. Its role was to develop advice on the planning, coordination and setting of priorities for DFO research on phycotoxins and toxic or harmful algae. Currently, a recently established National DFO Working Group on Harmful Algal Blooms is evaluating the status of HABs and the state of HAB knowledge in Canadian marine waters. The Ecosystem Stressors Program, under DFO’s Ecosystems and Oceans Science Sector, has requested science advice on the national scope of HAB incidences and impacts in Canadian marine waters, including the identification of knowledge gaps and areas or issues of particular concern to Canadian marine ecosystems. This is being accomplished in part using funding from DFO’s Strategic Program for Ecosystem-Based Research and Advice ([SPERA](#)), and an advisory document was produced (McKenzie et al. 2020a).

Research is only beginning to understand the complexities of HAB dynamics and factors that regulate phycotoxin production; this information will eventually lead to a predictive capability that will help to protect human health and the aquaculture industry in Canada. For example, we are only starting to grasp the importance of hydrographic and meteorological factors in initiating and controlling blooms of toxic dinoflagellates in the St. Lawrence Estuary and the Bay of Fundy. However, there is still much to learn about other factors (biotic and abiotic) that could cause these and other phytoplankton to suddenly increase to concentrations that cause harm (Pennell 1988). An example is the DA-producing *Pseudo-nitzschia multiseries* that suddenly bloomed to dangerous levels in Cardigan Bay (PE) in the autumn of 1987, resulting in human illnesses and deaths, molluscan shellfish harvesting closures, and economic losses. Likewise, there was an unexpected toxic bloom of *P. seriata*, which started prior to March 2002, that closed shellfish harvesting along the southern Gulf of St. Lawrence until early May 2002. Only with long-term (20–50 years) monitoring programs, such as those that used to operate in the Bay of Fundy and in BC for STX group toxins and *Alexandrium*, can we start to discern trends. Phytoplankton monitoring will enable us to discriminate between natural and anthropogenic causes of the harmful and toxic algal blooms, and to determine which phytoplankton are indigenous or may have been introduced by ballast water discharge. On a

shorter time scale (5–10 years), it is still possible to begin to identify inter-annual variations in bloom intensity and to develop an understanding of the dynamics of toxic blooms. This requires monitoring to be in place prior to the beginning of a bloom, in order to obtain oceanographic measurements and phytoplankton samples during its development, as well as during its peak and decline. Use of molecular tools (Kudela et al. 2010; Danovaro et al. 2016), moored buoys (e.g. CytoBuoy, Environmental Sample Processor) (Doucette et al. 2018), and remote sensing technology (Gower 1994; Harrison et al. 2007; Andrade et al. 2016; Devred et al. 2018; Doucette et al. 2018) will improve our understanding of toxic blooms. At the shortest time scale (weeks), monitoring for the presence of toxic or harmful phytoplankton using conventional approaches has already proven to be an effective way to provide an advanced warning in order for government agencies and aquaculturists to carry out mitigating measures (e.g. Bouchard-Steeves et al. 1993a). Because some toxic blooms are not an annual event, there is a tendency for complacency. This has resulted in reduced funding for long-term to mid-term research and monitoring programs, which has jeopardized the progress made so far. The loss of government participation in “phytoplankton watch” monitoring programs has, in some cases, been taken up by private industry, as is already the case in several countries around the world. However, this incentive would benefit from coordination with a national research program. Although the CFIA biotoxin monitoring program provides excellent data to protect shellfish consumers, it is not designed to fully evaluate HAB trends and broader-scale ecological impacts.

A major concern is that the source of several phycotoxins remains unknown in some locations. This is true for DA in razor clams and Dungeness crabs on the Pacific coast, in scallops in the Gulf of Maine, Georges/Browns Bank and southwestern NS, and in other molluscan shellfish in coastal NL. The source of OA group toxins on the south shore of NS, in northeastern NB, and in NL is still not known. In BC, the source of MC-LR is unknown. When an obvious source organism, such as *Dinophysis* spp. for OA group toxins is present, bulk samples from the field have, for unknown reasons, been nontoxic. It is possible that conditions for toxin synthesis were simply not appropriate at the time. Alternatively, the so-called “phycotoxins” may not be produced by phytoplankton, but rather by bacteria, either autonomously or in association with the algae (Doucette 1995). Such organisms would not be detected in samples collected with the phytoplankton nets or filtration systems conventionally used by phytoplankton monitoring programs.

Several reasons may explain why new harmful species and phycotoxins have been increasingly reported. Heightened awareness, precipitated by the 1987 “mussel crisis” (Section 3.1), undoubtedly resulted in the discovery of phycotoxins and algae that may have been present for decades. Some toxic/harmful algae have been introduced via ships’ ballast water, or by the transfer of molluscan shellfish or their spat used in aquaculture (Scarratt et al. 1993). Other algal species, which form the “hidden flora” (Smayda 1990) in that they are normally found at low concentrations, may have recently flourished as a result of coastal eutrophication or unusual meteorological conditions. Our coastal inlets are being used increasingly for aquaculture. This has led to an elevated awareness of species present in our waters because of monitoring and research programs. In particular, the CFIA’s biotoxin monitoring program detects all known phycotoxins in Canadian waters, which protects consumers of molluscan shellfish. However, there is still a need to be vigilant for emerging phycotoxins.

The increased occurrence of phycotoxin-producing species in the Arctic is a significant concern. Climate change will likely continue to expand the range of problematic species, and conditions will become more favourable for HABs in the Arctic (Walsh et al. 2011). This will lead to the

accumulation of phycotoxins at higher trophic levels (Lefebvre et al. 2016). Information is required to learn how phycotoxins impact the health of Canadian Arctic marine mammals and of coastal communities that rely on local harvest for their diets. It is therefore essential to develop and maintain research and monitoring activities on Canadian Arctic harmful algae and phycotoxins, in order to inform resource management in that region. These concerns could be noted by the new DFO Arctic Region, which will be created over the next few years.

Since the mid-2000s, DFO's priorities have shifted from HAB research to aquatic invasive (or marine non-indigenous) species (e.g. McKenzie et al. 2010; Adams et al. 2012; Casas-Monroy et al. 2014; Lehtiniemi et al. 2015; Chan et al. 2018)). For example, since the late 1990s, the major problem for mussel aquaculture on PE is no longer blooms of toxic *Pseudo-nitzschia*, but rather three invasive ascidian tunicates (*Botrylloides violaceus*, *Ciona intestinalis*, and *Styela clava*), which have since also become problematic in NS and NL (Comeau et al. 2015). It is tempting to speculate that a rise in filter-feeding invasive tunicates at these locations may coincide with the decline in *Pseudo-nitzschia* blooms. Other phytoplankton species, which are also a food source for tunicates, may have likewise declined in abundance. However, they may have gone unnoticed because they are nontoxic and therefore not being monitored.

HAB species are still included in ballast water studies. Consumers of molluscan shellfish are protected by the CFIA's biotoxin monitoring program. However, the danger is that it is very likely that new phycotoxins, and new toxigenic phytoplankton that are not included in current monitoring programs, will be discovered, or even remain undiscovered.

Other reasons for the decline in HAB research and monitoring by DFO are discussed in the Cohen Commission Report (Cohen 2012a). A DFO official testified that the thinking behind the decision not to fund or prioritize HAB research after 2004–2005 was because any human health issues related to HABs are caused by the consumption of molluscan shellfish, which is under the mandate of the CFIA. Furthermore, because losses of fish in netpens are not a human health concern, this issue should be dealt with primarily by the industry. Impacts of HABs on wild fish were not generally considered to be an important issue. However, as shown by numerous examples provided throughout this review, HABs do not simply impact human health; they have also been demonstrated to affect the dynamics and health of marine ecosystems. As a consequence, research on HAB issues is important not only for understanding marine ecosystem health, but also for improving the management of marine resources.

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13.0 List of Acronyms and Abbreviations

AANS	Aquaculture Association of Nova Scotia
ACRDP	Aquaculture Collaborative Research and Development Program (DFO)
ADAM	Anthryldiazomethane reagent for OA and DTX1 detection
AOAC	Association of Official Analytical Chemists
AZMP	Atlantic Zone Monitoring Program
ASP	Amnesic Shellfish Poisoning
AZA	Azaspiracid
AZP	Azaspiracid poisoning
BIO	Bedford Institute of Oceanography (Dartmouth, NS)
BMAA	β -N-methylamino-L-alanine
CAISN	Canadian Aquatic Invasive Species Network
CFIA	Canadian Food Inspection Agency
CFP	Ciguatera Fish Poisoning
CRM	Certified Reference Material (National Research Council, Halifax, NS)
CSAS	Canadian Science Advisory Secretariat
CSSP	Canadian Shellfish Sanitation Program
CWHA	Canadian Workshop on Harmful Algae
CWHMA	Canadian Workshop on Harmful Marine Algae
DA	Domoic acid
DFO	Department of Fisheries and Oceans (Fisheries and Oceans Canada)
DSP	Diarrhetic Shellfish Poisoning
DTX	Dinophysistoxin
EC	European Commission
ECCC	Environment and Climate Change Canada
EEZ	Exclusive Economic Zone
EU	European Union
ECOHAB	Ecology and Oceanography of Harmful Algal Blooms
EFSA	European Food Safety Authority
FAO	Food and Agriculture Organization of the United Nations
FDA	Food and Drug Administration of the United States
GEOHAB	Global Ecology and Oceanography of Harmful Algal Bloom Program
GTX	Gonyautoxin
GYM	Gymnodimine
HAB	Harmful algal bloom

HAE	Harmful algal event
HAEDAT	ICES-IOC Harmful Algal Event Database
HAMP	Harmful Algae Monitoring Program (in BC)
HPLC	High performance liquid chromatography
HPLC-PCOX	HPLC post-column oxidation
HRMS	High-resolution mass spectrometry
ICES	International Council for the Exploration of the Sea
IMO	International Maritime Organization
IOC	International Oceanographic Commission of UNESCO
IOS	Institute of Ocean Sciences (Sidney, BC)
IPCC	Intergovernmental Panel on Climate Change
IPHAB	International Panel on Harmful Algal Blooms (of IOC)
LC-MS	Liquid chromatography with mass spectrometry detection
LC-MS/MS	Liquid chromatography-tandem mass spectrometry
LFI	Lateral flow immunoassay
LSTs	Lipophilic shellfish toxins
MC-LR	Microcystin-LR
MLI	Maurice Lamontagne Institute (Mont-Joli, QC)
NEO	Neosaxitoxin
NHW	Department of National Health and Welfare
NOAA	National Oceanic and Atmospheric Administration
NMR	Nuclear magnetic resonance
NRC	National Research Council of Canada
NSP	Neurotoxic shellfish poisoning
OA	Okadaic acid
OA _{eq}	Okadaic acid equivalents
ORHAB	Olympic Region Harmful Algal Bloom program (WA, U.S.)
PICES	North Pacific Marine Science Organization
PSP	Paralytic Shellfish Poisoning
PTX	Pectenotoxin
PnTX	Pinnatoxin
PPIA	Protein phosphatase inhibition assay (for OA group toxins)
PWG	Phycotoxins Working Group
qPCR	Quantitative polymerase chain reaction assay
SABS	St. Andrews Biological Station (St. Andrews, NB)
SCOR	Scientific Committee on Oceanic Research
SPATT	Solid Phase Adsorption Toxin Tracking
SPERA	Strategic Program for Ecosystem-Based Research and Advice
SPX	Spirolide toxin
STX	Saxitoxin
STX _{eq}	Saxitoxin equivalents
TAMP	Toxic Algae Monitoring Program (Quebec Region of the Gulf of St. Lawrence)
TTX	Tetrodotoxin
UPLC-MS/MS	Ultra performance liquid chromatography-tandem mass spectrometry
WG HABD	Working Group on Harmful Algal Bloom Dynamics (of ICES-IOC)
WHO	World Health Organization
WHOI	Woods Hole Oceanographic Institution (Woods Hole, MA, U.S.)

ww	Wet weight
YTX	Yessotoxin
YTXeq	Yessotoxin equivalents

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Appendices

Appendix 1. Occurrence of potentially harmful and/or toxic algal species reported in sea ice and plankton from five Canadian Arctic regions: Canada Basin, eastern Arctic (EA), western Arctic (WA), Canadian Archipelago (CA), and Hudson Bay system (HB). References are listed in the main [reference list](#). See also Pućko et al. (2019).

Taxon	Habitat	Canadian Arctic sites	Reference
Centric diatoms			
<i>Chaetoceros concavicornis</i>	Sea ice	Canada Basin	Melnikov et al. 2002; Mather et al. 2010
	Plankton	Canada Basin	Melnikov et al. 2002
		EA, Baffin Bay	Sekerak et al. 1979; Lovejoy et al. 2002
		EA, Davis Strait	MacLaren Atlantic Limited 1977, 1978
		EA, Lincoln Sea	Seidenfaden 1947
		EA, Baffin Bay – Davis Strait	Grøntved and Seidenfaden 1938; Hsiao 1983; Mather et al. 2010
		EA, Lancaster Sound	Seidenfaden 1947; Sekerak et al. 1976a, 1979
		EA, Lancaster Sound+Jones Sound	Grøntved and Seidenfaden 1938; Seidenfaden 1947; Sekerak et al. 1976a, 1979
		CA, Amundsen Gulf	McLaughlin et al. 2009
		CA, Barrow Strait	Sekerak et al. 1976b
		Canadian Archipelago	Hsiao 1983
		WA, Beaufort Sea	Hsiao 1983; Mather et al. 2010; Crawford et al. 2018
		HB, Hudson Bay+Strait	Bursa 1961a; Simard et al. 1996; Harvey et al. 1997
		HB, Hudson Bay	Hsiao 1983
		HB, Foxe Basin	Bursa 1961b
<i>C. convolutus</i>	Sea ice	Canada Basin	Melnikov et al. 2002
	Plankton	Canada Basin	Melnikov et al. 2002
		EA, Frobisher Bay	Bursa 1971; Hsiao and Pinkewycz 1984; Hsiao 1985

Taxon	Habitat	Canadian Arctic sites	Reference
		EA, Baffin Bay	Sekerak et al. 1979; Lovejoy et al. 2002
		EA, Davis Strait	MacLaren Atlantic Limited 1977, 1978
		EA, Lincoln Sea	Seidenfaden 1947
		EA, Baffin Bay – Davis Strait	Grøntved and Seidenfaden 1938; Hsiao 1983; Mather et al. 2010
		EA, Lancaster Sound	Seidenfaden 1947; Sekerak et al. 1976a, 1979; Crawford et al. 2018
		EA, Lancaster Sound+Jones Sound	Grøntved and Seidenfaden 1938; Seidenfaden 1947; Sekerak et al. 1976a, 1979; Crawford et al. 2018
		CA, Amundsen Gulf	McLaughlin et al. 2009; M. Poulin unpubl. data
		CA, Barrow Strait	Sekerak et al. 1976b
		Canadian Archipelago	Hsiao 1983
		WA, Beaufort Sea	McLaughlin et al. 2009; Crawford et al. 2018; M. Poulin unpubl. data
		HB, Hudson Strait	Seidenfaden 1947; Percy et al. 1992
		HB, Hudson Strait, Ungava Bay	Polunin 1934; Seidenfaden 1947; Percy et al. 1992
		HB, Hudson Bay+Strait	Bursa 1961a
		HB, Hudson Bay	Davidson 1931; Anderson et al. 1981; Hsiao 1983
<i>C. danicus</i>	Plankton	EA, Frobisher Bay	Bursa 1971
		EA, Baffin Bay – Davis Strait	Hsiao 1983
		HB, Hudson Strait	Percy et al. 1992
		HB, Hudson Bay	Anderson et al. 1981
<i>C. debilis</i>	Plankton	EA, Frobisher Bay	Bursa 1971; Hsiao and Pinkewycz 1984, 1985b; Hsiao 1985
		EA, Baffin Bay	Sekerak et al. 1979; Lovejoy et al. 2002
		EA, Davis Strait	MacLaren Atlantic Limited 1977; Hsiao and Trucco 1980

Taxon	Habitat	Canadian Arctic sites	Reference
<i>C. gelidus</i> (formerly <i>C. socialis</i>)		EA, Eclipse Sound	Bursa 1971
		EA, Lincoln Sea	Seidenfaden 1947
		EA, Baffin Bay – Davis Strait	Grøntved and Seidenfaden 1938; Hsiao 1983; Mather et al. 2010; Crawford et al. 2018
		EA, Labrador Sea fjords	Simo-Matchim et al. 2017
		EA, Lancaster Sound	Seidenfaden 1947; Sekerak et al. 1976a, 1979
		EA, Lancaster Sound+Jones Sound	Grøntved and Seidenfaden 1938; Seidenfaden 1947; Sekerak et al. 1976a, 1979
		CA, Amundsen Gulf	McLaughlin et al. 2009; M. Poulin unpubl. data
		CA, Barrow Strait	Thomson et al. 1975
		CA, Resolute Bay	von Quillfeldt 2000
		Canadian Archipelago	Hsiao 1983
		WA, Beaufort Sea	Mather et al. 2010; Crawford et al. 2018; M. Poulin unpubl. data
		HB, Hudson Strait	Seidenfaden 1947; Percy et al. 1992
		HB, Hudson Strait, Ungava Bay	Polunin 1934; Seidenfaden 1947; Pinkewycz et al. 1987; Percy et al. 1992
		HB, Hudson Bay+Strait	Bursa 1961a; Simard et al. 1996; Harvey et al. 1997
		HB, Hudson Bay	Anderson et al. 1981; Hsiao 1983
		HB, Foxe Basin	Bursa 1961b
	Sea ice	Canada Basin	Melnikov et al. 2002
	Plankton	EA, Baffin Bay	Simard 2003
		Canada Basin	Melnikov et al. 2002; Joo et al. 2012; Crawford et al. 2018
		EA, Frobisher Bay	Bursa 1971; Grainger and Hsiao 1982; Hsiao and Pinkewycz 1983, 1984; Hsiao 1985

Taxon	Habitat	Canadian Arctic sites	Reference
<i>Conticribra weissflogii</i> (formerly		EA, Baffin Bay	Sekerak et al. 1979; Lovejoy et al. 2002; Joli et al. 2018
		EA, Davis Strait	Hsiao and Trucco 1980
		EA, Labrador Sea	MacLaren Marex 1979a,b
		EA, Lincoln Sea	Seidenfaden 1947
		EA, Baffin Bay – Davis Strait	Grøntved and Seidenfaden 1938; Hsiao 1983; Mather et al. 2010; Crawford et al. 2018
		EA, Labrador Sea fjords	Simo-Matchim et al. 2017
		EA, Lancaster Sound	Seidenfaden 1947; Sekerak et al. 1976a, 1979; Crawford et al. 2018
		EA, Lancaster Sound+Jones Sound	Grøntved and Seidenfaden 1938; Seidenfaden 1947; Sekerak et al. 1976a, 1979; Crawford et al. 2018
		CA, Amundsen Gulf	McLaughlin et al. 2009; M. Poulin unpubl. data
		CA, Darnley Bay	Mundy et al. 2011
		CA, Barrow Strait	Sekerak et al. 1976b; Bain et al. 1977; Riedel et al. 2003
		CA, Resolute Bay	von Quillfeldt 2000
		Canadian Archipelago	Hsiao 1983; Mather et al. 2010
		WA, Beaufort Sea	Foy and Hsiao 1976; Hsiao 1976, 1983; Hsiao et al. 1977; McLaughlin et al. 2009; Mather et al. 2010; Ardyna et al. 2017; Crawford et al. 2018; M. Poulin unpubl. data
		HB, Hudson Strait	Seidenfaden 1947; Percy et al. 1992
		HB, Hudson Strait, Ungava Bay	Polunin 1934; Seidenfaden 1947; Percy et al. 1992
		HB, Hudson Bay+Strait	Bursa 1961a; Simard et al. 1996; Harvey et al. 1997
		HB, Hudson Bay	Hsiao 1983
		HB, Foxe Basin	Bursa 1961b
	Plankton	EA, Baffin Bay	Sekerak et al. 1979

Taxon	Habitat	Canadian Arctic sites	Reference
<i>Thalassiosira weissflogii</i>		EA, Baffin Bay – Davis Strait	Hsiao 1983
		Canadian Archipelago	Hsiao 1983
<i>Ditylum brightwellii</i>	Plankton	EA, Frobisher Bay	Hsiao and Pinkewycz 1985a
		EA, Baffin Bay	Lovejoy et al. 2002
		EA, Baffin Bay – Davis Strait	Mather et al. 2010
<i>Leptocylindrus minimus</i>	Sea ice	Canada Basin	Melnikov et al. 2002
	Plankton	Canada Basin	Melnikov et al. 2002
		EA, Davis Strait	MacLaren Atlantic Limited 1978
		EA, Baffin Bay – Davis Strait	Hsiao 1983
		EA, Labrador Sea fjords	Simo-Matchim et al. 2017
		CA, Amundsen Gulf	M. Poulin unpubl. data
		CA, Barrow Strait	Sekerak et al. 1976b
		WA, Beaufort Sea	M. Poulin unpubl. data
		HB, Hudson Strait	Seidenfaden 1947
		HB, Hudson Strait, Ungava Bay	Polunin 1934; Seidenfaden 1947
		HB, Hudson Bay+Strait	Bursa 1961a; Simard et al. 1996; Harvey et al. 1997
		HB, Hudson Bay	Anderson et al. 1981; Hsiao 1983
<i>Proboscia inermis</i>	Plankton	EA, Lincoln Sea	Seidenfaden 1947
		EA, Baffin Bay – Davis Strait	Grøntved and Seidenfaden 1938; Hsiao 1983
		HB, Hudson Strait	Seidenfaden 1947
		HB, Hudson Bay	Hsiao 1983
<i>Skeletonema costatum</i>	Sea ice	HB, Hudson Bay	Poulin et al. 1983
	Plankton	EA, Baffin Bay	Sekerak et al. 1979; Lovejoy et al. 2002

Taxon	Habitat	Canadian Arctic sites	Reference
<i>Thalassiosira aestivalis</i>	Sea ice	EA, Davis Strait	MacLaren Atlantic Limited 1977, 1978
		EA, Baffin Bay – Davis Strait	Hsiao 1983; Mather et al. 2010
		EA, Labrador Sea fjords	Simo-Matchim et al. 2017
		EA, Lancaster Sound	Sekerak et al. 1979
		CA, Amundsen Gulf	M. Poulin unpubl. data
		WA, Beaufort Sea	Foy and Hsiao 1976; Hsiao 1976, 1983; Hsiao et al. 1977; M. Poulin unpubl. data
		HB, Hudson Strait	Percy et al. 1992
		HB, Hudson Bay+Strait	Bursa 1961a; Simard et al. 1996; Harvey et al. 1997
		HB, Hudson Bay	Anderson et al. 1981; Hsiao 1983
		EA, Frobisher Bay	Grainger and Hsiao 1982
		Plankton EA, Frobisher Bay	Grainger and Hsiao 1982
		EA, Davis Strait	MacLaren Atlantic Limited 1978
		HB, Hudson Bay	Hsiao 1983
		HB, Foxe Basin	Bursa 1961b
<i>T. rotula</i>	Plankton	EA, Frobisher Bay	Bursa 1971
		EA, Baffin Bay	Sekerak et al. 1979
		EA, Davis Strait	MacLaren Atlantic Limited 1977, 1978
		EA, Baffin Bay – Davis Strait	Hsiao 1983
		EA, Lancaster Sound	Sekerak et al. 1979
		Canadian Archipelago	Hsiao 1983
		WA, Beaufort Sea	Mather et al. 2010
		HB, Hudson Strait	Seidenfaden 1947
		HB, Hudson Strait, Ungava Bay	Polunin 1934; Seidenfaden 1947
		HB, Hudson Bay+Strait	Bursa 1961a

Taxon	Habitat	Canadian Arctic sites	Reference
		HB, Hudson Bay	Hsiao 1983; Hsiao et al. 1984
		HB, Foxe Basin	Bursa 1961b
Pennate diatoms			
<i>Halamphora coffeiformis</i>	Sea ice	EA, Frobisher Bay	Grainger and Hsiao 1982
	Plankton	EA, Frobisher Bay	Grainger and Hsiao 1982
<i>Phaeodactylum tricornutum</i>	Sea ice	Canada Basin	Melnikov et al. 2002
<i>Pseudo-nitzschia arctica</i>	Plankton	CA, Barrow Strait	Percopo et al. 2016
		WA, Beaufort Sea	Percopo et al. 2016
<i>P. delicatissima</i>	Sea ice	Canada Basin	Melnikov et al. 2002
		EA, Baffin Bay – Davis Strait	Hsiao 1980
		EA, Labrador Sea	MacLaren Marex 1979a,b
		EA, Davis Strait	Booth 1984
		EA, Frobisher Bay	Grainger and Hsiao 1982; Hsiao 1983
		EA, Eclipse Sound+ Pond Inlet	Hsiao 1979a, 1983; Cross 1980, 1982, 1987
		EA, Pond Inlet	Cross 1980, 1982
		CA, Franklin Bay	Róžańska et al. 2009
		CA, Barrow Strait	Thomson et al. 1975; Riedel et al. 2003
		WA, Beaufort Sea	Pett et al. 1983; Woods and Smiley 1987
		HB, Hudson Bay	Poulin et al. 1983; Hsiao et al. 1984
	Plankton	EA, Frobisher Bay	Bursa 1971; Hsiao and Pinkewycz 1985a,b
		EA, Lancaster Sound	Sekerak et al. 1976a, 1979
		EA, Baffin Bay	Sekerak et al. 1979; Hsiao and Pinkewycz 1987; Lovejoy et al. 2002
		EA, Davis Strait	MacLaren Limited Atlantic 1978
		EA, Labrador Sea	MacLaren Marex 1979a,b
		EA, Lincoln Sea	Seidenfaden 1947

Taxon	Habitat	Canadian Arctic sites	Reference
		EA, Baffin Bay – Davis Strait	Hsiao 1983
		EA, Labrador Sea fjords	Simo-Matchim et al. 2017
		EA, Lancaster Sound+Jones Sound	Sekerak et al. 1976a, 1979; Hsiao and Pinkewycz 1987
		CA, Amundsen Gulf	Adams 1975; M. Poulin unpubl. data
		CA, Barrow Strait	Thomson et al. 1975; Bain et al. 1977; Riedel et al. 2003
		CA, Resolute Bay	von Quillfeldt 2000
		Canadian Archipelago	Hsiao 1983
		WA, Beaufort Sea	Pett et al. 1983; Pinkewycz and Hsiao 1987; M. Poulin unpubl. data
		HB, Hudson Strait	Percy et al. 1992
		HB, Hudson Strait, Ungava Bay	Pinkewycz et al. 1987; Percy et al. 1992
		HB, Hudson Bay+Strait	Bursa 1961a; Simard et al. 1996; Harvey et al. 1997
		HB, Hudson Bay	Davidson 1931; Anderson et al. 1981; Hsiao et al. 1984
<i>P. granii</i>	Plankton	EA, Baffin Bay	Lovejoy et al. 2002
<i>P. obtusa</i>	Plankton	EA, Davis Strait	Crawford et al. 2018
		EA, Labrador Sea fjords	Simo-Matchim et al. 2017
		EA, Lancaster Sound	Crawford et al. 2018
		CA, Amundsen Gulf	M. Poulin unpubl. data
		CA, Resolute Bay	von Quillfeldt 2000
<i>P. pseudodelicatissima</i>	Sea ice	WA, Beaufort Sea	M. Poulin unpubl. data
		Canada Basin	Mather et al. 2010
		EA, Baffin Bay	Simard 2003
	Plankton	CA, Franklin Bay	Róžańska et al. 2009
		WA, Beaufort Sea	Róžańska et al. 2008
		EA, Labrador Sea fjords	Simo-Matchim et al. 2017

Taxon	Habitat	Canadian Arctic sites	Reference
<i>P. pungen</i>	Sea ice	CA, Amundsen Gulf	Róžańska et al. 2008; McLaughlin et al. 2009; M. Poulin unpubl. data
		CA, Darnley Bay	Mundy et al. 2011
		CA, Resolute Bay	von Quillfeldt 2000
		WA, Beaufort Sea	Róžańska et al. 2008; M. Poulin unpubl. data
		HB, Hudson Strait	Percy et al. 1992
		CA, Franklin Bay	Róžańska et al. 2009
		Plankton EA, Eclipse Sound	Bursa 1971
		HB, Hudson Strait	Percy et al. 1992
		HB, Hudson Bay+Strait	Bursa 1961a
		HB, Hudson Bay	Hsiao 1983
<i>P. seriata</i>	Sea ice	HB, Foxe Basin	Bursa 1961b
		Canada Basin	Melnikov et al. 2002
		EA, Davis Strait	Hsiao 1979c
		EA, Eclipse Sound+Frobisher Bay	Hsiao 1979a, 1980, 1983; Cross 1987
		EA, Frobisher Bay	Grainger and Hsiao 1982; Hsiao 1983
		EA, Pond Inlet	Cross 1980, 1982
		CA, Franklin Bay	Róžańska et al. 2009
		CA, Barrow Strait	Thomson et al. 1975
		CA, Resolute Passage	Kikuchi-Kawanobe and Kudoh 1995
		WA, Eskimo Lakes	Hsiao 1979b, 1980, 1983
		WA, Beaufort Sea	Woods and Smiley 1987; Hopky et al. 1994a,b
		HB, Hudson Bay	Hsiao et al. 1984
	Plankton	EA, Frobisher Bay	Bursa 1971; Grainger and Hsiao 1982; Hsiao and Pinkewycz 1983, 1984, 1985a,b
		EA, Davis Strait	MacLaren Atlantic Limited 1977, 1978; Hsiao and Trucco 1980
		EA, Baffin Bay	Sekerak et al. 1979; Hsiao and Pinkewycz 1987; Lovejoy et al. 2002

Taxon	Habitat	Canadian Arctic sites	Reference
<i>P. turgidula</i>		EA, Eclipse Sound	Bursa 1971
		EA, Labrador Sea	MacLaren Marex 1979b
		EA, Labrador Sea fjords	Simo-Matchim et al. 2017
		EA, Lincoln Sea	Seidenfaden 1947
		EA, Baffin Bay – Davis Strait	Grøntved and Seidenfaden 1938; Hsiao 1983
		EA, Lancaster Sound	Seidenfaden 1947; Sekerak et al. 1976a, 1979
		EA, Lancaster Sound+Jones Sound	Grøntved and Seidenfaden 1938; Seidenfaden 1947; Sekerak et al. 1976a, 1979; Hsiao and Pinkewycz 1987
		CA, Amundsen Gulf	Adams 1975; McLaughlin et al. 2009
		CA, Dolphin and Union Strait	Mann 1925
		CA, Barrow Strait	Thomson et al. 1975; Sekerak et al. 1976b; Bain et al. 1977
		CA, Resolute Bay	von Quillfeldt 2000; Brown et al. 2014
		Canadian Archipelago	Hsiao 1983
		WA, Beaufort Sea	Foy and Hsiao 1976; Hsiao 1976, 1983; Pett et al. 1983; Pinkewycz and Hsiao 1987; Mather et al. 2010
		HB, Hudson Strait	Seidenfaden 1947; Percy et al. 1992
		HB, Hudson Strait, Ungava Bay	Polunin 1934; Seidenfaden 1947; Pinkewycz et al. 1987; Percy et al. 1992
		HB, Hudson Bay+Strait	Bursa 1961a; Simard et al. 1996; Harvey et al. 1997
		HB, Hudson Bay	Davidson 1931; Anderson et al. 1981; Hsiao et al. 1984
		HB, Foxe Basin	Bursa 1961b
	Sea ice	CA, Franklin Bay	Róžańska et al. 2009
	Plankton	CA, Amundsen Gulf	McLaughlin et al. 2009; M. Poulin unpubl. data

Taxon	Habitat	Canadian Arctic sites	Reference
		WA, Beaufort Sea	M. Poulin unpubl. data
Dinoflagellates			
<i>Alexandrium monilatum</i>	Sea ice	HB, Hudson Bay	Hsiao et al. 1984
	Plankton	EA, Frobisher Bay	Hsiao and Pinkewycz 1983, 1984, 1985a,b
		HB, Hudson Strait, Ungava Bay	Pinkewycz et al. 1987
		HB, Hudson Bay	Hsiao et al. 1984
<i>A. ostenfeldii</i>	Plankton	CA, Amundsen Gulf	M. Poulin unpubl. data
		WA, Beaufort Sea	M. Poulin unpubl. data
		HB, Hudson Bay+Strait	Simard et al. 1996; Harvey et al. 1997
		HB, Hudson Bay	Hsiao 1983
		HB, Foxe Basin	Bursa 1961b
<i>A. tamarense</i>	Sea ice	CA, Barrow Strait	Riedel et al. 2003
		WA, Beaufort Sea	Niemi et al. 2011
	Plankton	EA, Frobisher Bay	Hsiao and Pinkewycz 1985b
		CA, Amundsen Gulf	M. Poulin unpubl. data
		WA, Beaufort Sea	Niemi et al. 2011; M. Poulin unpubl. data
		HB, Hudson Strait	Percy et al. 1992
		HB, Hudson Bay+Strait	Bursa 1961a
		HB, Hudson Bay	Hsiao 1983; Hsiao et al. 1984
		HB, Foxe Basin	Bursa 1961b
<i>Coolia monotis</i>	Sea ice	HB, Hudson Bay	Hsiao et al. 1984
	Plankton	HB, Hudson Bay	Hsiao et al. 1984
<i>Dinophysis acuminata</i>	Sea ice	CA, Franklin Bay	Róžańska et al. 2009
		WA, Beaufort Sea	Woods and Smiley 1987; Róžańska et al. 2008
		HB, Hudson Bay	Hsiao et al. 1984
	Plankton	Canada Basin	Melnikov et al. 2002

Taxon	Habitat	Canadian Arctic sites	Reference
		EA, Frobisher Bay	Hsiao and Pinkewycz 1983, 1985a,b
		EA, Baffin Bay	Lovejoy et al. 2002
		EA, Lincoln Sea	Seidenfaden 1947
		EA, Baffin Bay – Davis Strait	Grøntved and Seidenfaden 1938; Hsiao 1983
		EA, Labrador Sea fjords	Simo-Matchim et al. 2017
		CA, Amundsen Gulf	Róžańska et al. 2008; McLaughlin et al. 2009; M. Poulin unpubl. data
		WA, Beaufort Sea	Foy and Hsiao 1976; Hsiao 1976, 1983; Hsiao et al. 1977; Róžańska et al. 2008; McLaughlin et al. 2009; M. Poulin unpubl. data
		HB, Hudson Strait	Percy et al. 1992
		HB, Hudson Bay+Strait	Bursa 1961a; Simard et al. 1996; Harvey et al. 1997
		HB, Hudson Bay	Anderson et al. 1981; Hsiao et al. 1984
		Churchill, Deception Bay, Milne Inlet Wager Bay	Dhifallah 2019
<i>D. acuta</i>	Plankton	Canada Basin	Melnikov et al. 2002
		EA, Labrador Sea	MacLaren Marex 1979b
		EA, Lincoln Sea	Seidenfaden 1947
		EA, Baffin Bay – Davis Strait	Grøntved and Seidenfaden 1938; Hsiao 1983
		EA, Labrador Sea fjords	Simo-Matchim et al. 2017
		HB, Hudson Strait	Percy et al. 1992
		HB, Hudson Bay+Strait	Bursa 1961a; Simard et al. 1996
		HB, Hudson Bay	Anderson et al. 1981
		HB, Churchill, Deception Bay	Dhifallah 2019
<i>D. caudata</i>	Sea ice	HB, Foxe Basin	Daugbjerg and Vørs 1994
	Plankton	HB, Foxe Basin	Daugbjerg and Vørs 1994

Taxon	Habitat	Canadian Arctic sites	Reference
<i>D. fortii</i>	Plankton	HB, Hudson Bay	Anderson et al. 1981
<i>D. norvegica</i>	Plankton	Canada Basin	Melnikov et al. 2002
		EA, Labrador Sea	MacLaren Marex 1979b
		EA, Lincoln Sea	Seidenfaden 1947
		EA, Baffin Bay – Davis Strait	Grøntved and Seidenfaden 1938; Hsiao 1983
		EA, Labrador Sea fjords	Simo-Matchim et al. 2017
		EA, Lancaster Sound+Jones Sound	Grøntved and Seidenfaden 1938; Seidenfaden 1947
		EA, Lancaster Sound	Seidenfaden 1947
		Canadian Archipelago	Hsiao 1983
		HB, Hudson Strait	Percy et al. 1992
		HB, Hudson Bay+Strait	Bursa 1961a; Simard et al. 1996
		HB, Hudson Bay	Anderson et al. 1981; Hsiao 1983
		HB, Foxe Basin	Bursa 1961b
		HB, Churchill, Deception Bay	Dhifallah 2019
<i>Gonyaulax spinifera</i>	Plankton	EA, Baffin Bay	Lovejoy et al. 2002
		CA, Amundsen Gulf	McLaughlin et al. 2009
		HB, Hudson Bay	Bursa 1961a; Simard et al. 1996; Harvey et al. 1997
		HB, Churchill, Deception Bay	Dhifallah 2019
<i>Gyrodinium cf. aureolum</i>	Plankton	EA, Baffin Bay	Lovejoy et al. 2002
		EA, Davis Strait	MacLaren Atlantic Limited 1978
		EA, Baffin Bay – Davis Strait	Hsiao 1983
<i>Heterocapsa triquetra</i>	Plankton	EA, Baffin Bay	Lovejoy et al. 2002
		EA, Davis Strait	MacLaren Atlantic Limited 1978
		EA, Baffin Bay – Davis Strait	Grøntved and Seidenfaden 1938; Hsiao 1983

Taxon	Habitat	Canadian Arctic sites	Reference
<i>Karenia mikimotoi</i>	Plankton	CA, Amundsen Gulf	M. Poulin unpubl. data
		WA, Beaufort Sea	M. Poulin unpubl. data
		HB, Hudson Bay+Strait	Bursa 1961a
		HB, Hudson Bay	Anderson et al. 1981
		WA, Beaufort Sea	McLaughlin et al. 2009; Poulin et al. 2011
<i>Lingulodinium polyedrum</i>	Plankton	EA, Baffin Bay	Lovejoy et al. 2002
<i>Phalacroma rotundatum</i>	Plankton	CA, Amundsen Gulf	McLaughlin et al. 2009
		HB, Hudson Bay	Roff and Legendre 1986
		EA, Lincoln Sea	Seidenfaden 1947
		EA, Baffin Bay – Davis Strait	Grøntved and Seidenfaden 1938; Hsiao 1983
		EA, Labrador Sea fjords	Simo-Matchim et al. 2017
		EA, Lancaster Sound+Jones Sound	Grøntved and Seidenfaden 1938
		EA, Pond Inlet, Milne Inlet	Dhifallah 2019
		CA, Amundsen Gulf	McLaughlin et al. 2009
		HB, Hudson Strait	Percy et al. 1992
		HB, Hudson Bay+Strait	Bursa 1961a; Simard et al. 1996
<i>Prorocentrum lima</i>	Sea ice	HB, Hudson Bay	Anderson et al. 1981
		HB, Churchill, Deception Bay, Milne Inlet	Dhifallah 2019
	Plankton	WA, Cape Bathurst Polynya	Niemi et al. 2011
		EA, Davis Strait	Hsiao and Trucco 1980
		EA, Baffin Bay – Davis Strait	Hsiao 1983
		WA, Cape Bathurst Polynya	Niemi et al. 2011

Taxon	Habitat	Canadian Arctic sites	Reference
<i>P. minimum</i> (= <i>P. cordatum</i>)	Plankton	HB, Hudson Strait	Percy et al. 1992
		HB, Hudson Bay	Roff and Legendre 1986
		EA, Baffin Bay	Lovejoy et al. 2002
		EA, Labrador Sea fjords	Simo-Matchim et al. 2017
		CA, Amundsen Gulf	McLaughlin et al. 2009; M. Poulin unpubl. data
		WA, Beaufort Sea	M. Poulin unpubl. data
		HB, Hudson Bay+Strait	Simard et al. 1996; Harvey et al. 1997
<i>Protoceratium reticulatum</i>	Plankton	HB, Foxe Basin, Wager Bay	Dhifallah 2019
		Canada Basin	Melnikov et al. 2002
		EA, Baffin Bay	Lovejoy et al. 2002
		EA, Baffin Bay – Davis Strait	Grøntved and Seidenfaden 1938; Hsiao 1983
		EA, Lancaster Sound+Jones Sound	Grøntved and Seidenfaden 1938
		EA, Frobisher Bay/Iqaluit	Dhifallah 2019
		EA, Pond Inlet, Milne Inlet	Dhifallah 2019
		CA, Amundsen Gulf	Seidenfaden 1947
		Canadian Archipelago	M. Poulin unpubl. data
		WA, Beaufort Sea	Hsiao 1983
			M. Poulin unpubl. data
		HB, Hudson Strait	Percy et al. 1992
		HB, Hudson Bay+Strait	Bursa 1961a; Simard et al. 1996
		HB, Hudson Bay	Hsiao 1983; Hsiao et al. 1984
		HB, Foxe Basin	Bursa 1961b
		HB, Churchill, Deception Bay,	Dhifallah 2019

Taxon	Habitat	Canadian Arctic sites	Reference
<i>Protoperidinium crassipes</i>	Plankton	Iqaluit, Milne Inlet, Wager Bay	
		HB, Foxe Basin, Wager Bay	Dhifallah 2019
		Canada Basin	Melnikov et al. 2002
		EA, Baffin Bay	Sekerak et al. 1979
		EA, Baffin Bay – Davis Strait	Hsiao 1983
		EA, Lancaster Sound	Sekerak et al. 1976a
		Canadian Archipelago	Hsiao 1983
		HB, Hudson Bay+Strait	Bursa 1961a; Simard et al. 1996
Prymnesiophyte <i>Chrysochromulina birgeri</i>	Sea ice	HB, Foxe Basin	Daugbjerg and Vørs 1994
	Plankton	CA, Amundsen Gulf	M. Poulin unpubl. data
		WA, Beaufort Sea	M. Poulin unpubl. data
		HB, Foxe Basin	Daugbjerg and Vørs 1994
	Plankton	CA, Amundsen Gulf	M. Poulin unpubl. data
		WA, Beaufort Sea	M. Poulin unpubl. data
Raphidophyte <i>Heterosigma akashiwo</i>	Plankton	EA, Labrador Sea fjords	Simo-Matchim et al. 2017
		CA, Amundsen Gulf	McLaughlin et al. 2009; M. Poulin unpubl. data
		CA, Barrow Strait	Riedel et al. 2003
		WA, Beaufort Sea	Ardyna et al. 2017; M. Poulin unpubl. data
Silicoflagellate <i>Octactis speculum</i>	Sea ice	EA, Labrador Sea	MacLaren Marex 1979a,b
		EA, Davis Strait	Booth 1984
		EA, Frobisher Bay	Grainger and Hsiao 1982

Taxon	Habitat	Canadian Arctic sites	Reference
	Plankton	CA, Franklin Bay	Róžańska et al. 2009
		WA, Cape Bathurst Polynya	Niemi et al. 2011
		WA, Beaufort Sea	Róžańska et al. 2008
		Canada Basin	Gosselin et al. 1997; Joo et al. 2012; Crawford et al. 2018
		EA, Frobisher Bay	Hsiao and Pinkewycz 1985b
		EA, Baffin Bay	Sekerak et al. 1979; Lovejoy et al. 2002
		EA, Davis Strait	MacLaren Atlantic Limited 1978
		EA, Labrador Sea	MacLaren Marex 1979b
		EA, Baffin Bay – Davis Strait	Hsiao 1983; Crawford et al. 2018
		EA, Labrador Sea fjords	Simo-Matchim et al. 2017
		EA, Lancaster Sound	Sekerak et al. 1976a, 1979; Crawford et al. 2018
		CA, Amundsen Gulf	Adams 1975; Róžańska et al. 2008; M. Poulin unpubl. data
		CA, Barrow Strait	Thomson et al. 1975; Sekerak et al. 1976b; Riedel et al. 2003
		CA, Resolute Bay	von Quillfeldt 2000
		Canadian Archipelago	Hsiao 1983
		WA, Cape Bathurst Polynya	Niemi et al. 2011
		WA, Beaufort Sea	Foy and Hsiao 1976; Hsiao 1976, 1983; Hsiao et al. 1977; Róžańska et al. 2008; Crawford et al. 2018; M. Poulin unpubl. data
		HB, Hudson Strait	Percy et al. 1992
		HB, Hudson Bay+Strait	Bursa 1961a; Simard et al. 1996; Harvey et al. 1997
		HB, Hudson Bay	Hsiao et al. 1984
Cyanobacteria			
<i>Aphanizomenon flos-aquae</i>	Plankton	HB, Hudson Bay+Strait	Bursa 1961a

Taxon	Habitat	Canadian Arctic sites	Reference
Protozoan ciliate			
<i>Mesodinium rubrum</i>	Plankton	EA, Smith Sound, Central Northern Baffin Bay	Lovejoy et al. 2002; Dhifallah 2019

Appendix 2. Selected information on freshwater cyanobacteria and cyanotoxins in Canada.

Freshwater cyanobacteria and cyanotoxins have caused problems in British Columbia (HealthLinkBC 2014), Alberta (e.g. Kotak et al. 1993), Saskatchewan (Hammer 1968; Government of Saskatchewan 2014), Manitoba (e.g. Schindler et al. 2012), Ontario (e.g. Hotto et al. 2007; Winter et al. 2011), Quebec (e.g. Fortin et al. 2010), Nunavut (Vézina and Vincent 2009), New Brunswick (Department of Environment 2014; [CBC article](#); [CBC interview](#); [NB Government](#)), Nova Scotia (Nova Scotia Environment 2011), Prince Edward Island (Department of Environment, Labour and Justice 2014), and Newfoundland and Labrador (Department of Environment and Conservation 2014). Across Canada, there is evidence that toxic cyanobacterial blooms have continued to increase in lakes (Winter 2011; Pick 2016). Uncontrolled, cyanobacterial blooms on Lake Erie could cost Canada \$5.3 billion over 30 years (Smith et al. 2019). Huang and Zimba (2019) reviewed cyanobacterial bioactive metabolites.

Most analytical surveys focus on a few well known structures such as MC-LR. However, many new metabolites have been discovered and need to be added to the list of target analytes (Bouaïcha et al. 2019). A recent example is the characterization of [D-Leu¹] microcystin-LY (previously detected in field samples) from a culture isolated from a Saskatchewan lake (LeBlanc et al. 2020).

The Ontario Ministry of the Environment held workshops in 2017 and 2018, which discussed freshwater cyanobacterial blooms ([Interdisciplinary Freshwater Harmful Algal Blooms Workshop](#)); this will be a continuing workshop series. The rise in harmful cyanobacterial blooms, the fate of microcystins in the environment and their monitoring have recently been reviewed (O’Neil et al. 2012; Vareli et al. 2013; Schmidt et al. 2014).

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Appendix 3. Canadian Food Inspection Agency (CFIA) laboratories, the phycotoxins analyzed for, and the CFIA official methods used.

Laboratory	Phycotoxin	Method
Dartmouth, NS	Saxitoxin group toxins	HPLC–PCOX ¹
	Domoic acid	HPLC–UV ²
	Okadaic acid group toxins	UPLC–MS/MS ³
	Yessotoxins	UPLC–MS/MS
	Pectenotoxins	UPLC–MS/MS
	Spirolides	UPLC–MS/MS
	Azaspiracid	UPLC–MS/MS
	Gymnodimine	UPLC–MS/MS
	Pinnatoxins	UPLC–MS/MS
Longueuil, QC	Saxitoxin group toxins	HPLC–PCOX ¹
	Domoic acid	HPLC–UV ²
Burnaby, BC	Saxitoxin group toxins	HPLC–PCOX ¹
	Domoic acid	HPLC–UV ²

¹ High performance liquid chromatography post-column oxidation (van de Reit et al. 2011; [Website](#))

² High performance liquid chromatography with ultraviolet detection (“Analysis of domoic acid in molluscs and crustaceans by liquid chromatography”)

³ Ultra performance liquid chromatography-tandem mass spectrometry (“Analysis of lipophilic shellfish toxins by liquid chromatography/mass spectrometry”)

Appendix 4. Occurrence of potentially bloom-forming, harmful and/or toxigenic algal species found in ballast water or sediments of vessels arriving in Canadian ports on the Atlantic or Pacific coast, or that transited Atlantic Canada en route to the Laurentian Great Lakes and upper St. Lawrence River. Occurrences are organized by ballast management regime (ballast water exchange was not mandatory during 1992–2002 but was mandatory in 2006–2009; Section [8.1.2](#)). P = Present. Blank cells indicate the taxon was not detected. Sources of data: Subba Rao et al. 1994; Carver and Mallet 2002, 2003; Casas-Monroy et al. 2011; Gosselin et al. 1995; Harvey et al. 1999; Kaczmarska and Ehrman 2015; Klein et al. 2010; Lang and Kaczmarska 2012; Roy et al. 2012, 2014; Villac et al. 2013; Waters et al. 2001.

Ballast water regime	Exchange not mandatory:	Exchange mandatory:
Taxon	1992–2002	2006–2009
Cyanophytes		
<i>Anabaena</i> sp.	P	
<i>Chroococcus</i> sp.	P	
<i>Merismopedia punctata</i>	P	
<i>Oscillatoria</i> sp.	P	
Diatoms		
<i>Cerataulina pelagica</i>	P	
<i>Chaetoceros concavicornis</i>	P	
<i>C. convolutus</i>	P	P
<i>C. danicus</i>		P
<i>C. debilis</i>	P	P
<i>C. cf. hispidus</i>		P
<i>Corethron pennatum</i> (as <i>C. criophilum</i>)	P	
<i>Coscinodiscus wailesii</i>	P	
<i>Cylindrotheca closterium</i>		P
<i>Eucampia zodiacus</i>		P
<i>Fragilaria crotonensis</i>	P	P
<i>F. cf. crotonensis</i>	P	
<i>Fragilariopsis oceanica</i>	P	
<i>Guinardia delicatula</i>		P
<i>Leptocylindrus danicus</i>	P	P
<i>L. minimus</i>	P	P
<i>Pseudo-nitzschia australis</i> (?)	P	

Ballast water regime	Exchange not mandatory:	Exchange mandatory:
Taxon	1992–2002	2006–2009
<i>P. australis</i>		P
<i>P. cuspidata</i>		P
<i>P. delicatissima</i>	P	P
<i>P. delicatissima</i> complex		P
<i>P. fraudulenta</i>	P	P
<i>P. multiseriata</i>	P	P
<i>P. pseudodelicatissima</i>	P	
<i>P. pseudodelicatissima</i> complex	P	P
<i>P. pungens</i>	P	P
<i>P. seriata</i>	P	P
<i>P. turgidula</i>	P	P
<i>Rhizosolenia alata</i> f. <i>gracillima</i> (= <i>Proboscia alata</i>)	P	
<i>R. alata</i> f. <i>alata</i> (= <i>Proboscia alata</i>)	P	
<i>R. delicatula</i> (= <i>Guinardia delicatula</i>)	P	
<i>R. hebetata</i>	P	
<i>R. hebetata</i> f. <i>semispina</i>	P	
<i>R. setigera</i>	P	
<i>R. shrubsolei</i> (= <i>Rhizosolenia imbricata</i>)	P	
<i>R. stolterfothii</i> (= <i>Guinardia striata</i>)	P	
<i>R. styliformis</i>	P	
<i>Skeletonema costatum</i>	P	P
<i>Thalassiosira decipiens</i>	P	P
<i>T. gravida</i>	P	P
<i>T. nordenskioldii</i>	P	P
<i>T. rotula</i> (= <i>Thalassiosira gravida</i>)	P	P
<i>Thalassiosira</i> sp.	P	
Silicoflagellates (dictyochophytes)	P	
<i>Dictyocha fibula</i>	P	

Ballast water regime	Exchange not mandatory: 1992–2002	Exchange mandatory: 2006–2009
Taxon		
<i>Octactis octonaria</i> (as <i>Dictyocha octonaria</i>)	P	
<i>Octactis speculum</i> (as <i>Dictyocha speculum</i>)	P	
Chrysophytes		
<i>Ebria tripartita</i>	P	
Dinoflagellates		
<i>Alexandrium</i> cf. <i>pseudogonyaulax</i>		P
<i>A. tamarense</i> (= <i>A. catenella</i>)	P	P
<i>Alexandrium</i> sp.	P	
<i>Ceratium arcticum</i> (now <i>Tripos arcticus</i>)	P	
<i>C. furca</i> (now <i>Tripos furca</i>)	P	P
<i>C. fusus</i> (now <i>Tripos fusus</i>)	P	
<i>Cochlodinium polykrikoides</i> (= <i>Margalefidinium polykrikoides</i>)		P
<i>Dinophysis acuminata</i>	P	P
<i>D. acuta</i>	P	P
<i>D. caudata</i>	P	P
<i>D. dens</i> (= <i>D. acuta</i>)	P	P
<i>D. fortii</i>	P	P
<i>D. mitra</i> (= <i>Phalacroma mitra</i>)	P	
<i>D. norvegica</i>	P	P
<i>D. ovum</i>	P	
<i>D. pulchella</i>	P	
<i>D. punctata</i>	P	
<i>D. tripos</i>	P	P
<i>Gonyaulax birostris</i>	P	
<i>G. polygramma</i>		P
<i>G. turbynei</i>	P	
<i>G.</i> cf. <i>spinifera</i>		P
<i>Gymnodinium catenatum</i>		P

Ballast water regime	Exchange not mandatory:	Exchange mandatory:
Taxon	1992–2002	2006–2009
<i>Gymnodinium</i> sp.	P	
<i>Lingulodinium machaerophorum</i> (= <i>Lingulodinium polyedra</i>)		P
<i>Lingulodinium polyedra</i>		P
<i>Phalacroma mitra</i> (= <i>Dinophysis mitra</i>)	P	P
<i>P. rotundatua</i> (formerly <i>Dinophysis rotundata</i>)	P	P
<i>Polykrikos kofoidii</i>	P	
<i>Polysphaeridium zoharyi</i>		P
<i>Prorocentrum compressum</i> (now <i>P. bidens</i>)	P	
<i>P. gracile</i>	P	
<i>P. lima</i>	P	
<i>P. cf. lima</i>		P
<i>P. micans</i>	P	
<i>P. minimum</i> (= <i>P. cordatum</i>)	P	
<i>P. reticulatum</i>	P	P
<i>P. cf. reticulatum</i>		P
<i>P. scutellum</i>	P	
<i>Protoperidinium crassipes</i>	P	
<i>P. curtipes</i>	P	
<i>Scrippsiella trochoidea</i>	P	P

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