DINOFLAGELLÉS DANS LE PLANCTON ET LES SÉDIMENTS DU BRAS DE MER MILNE, NUNAVUT, CANADA

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PAR

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RÉSUMÉ

Entre 2013 et 2023, la distance parcourue par les vraquiers en Arctique a augmenté de plus de 300%, et ce à la suite de l'ouverture de la mine Mary River en 2014. La mine Mary River est une mine à ciel ouvert sur l'île de Baffin qui exploite un gisement de minerai de fer de haute qualité. Les vraquiers en attente de chargement jettent l'ancre dans le bras de mer Milne où ils vidangent leurs eaux de ballast. Bien qu'il existe des normes internationales et canadiennes afin de limiter l'introduction d'espèces non indigènes (ENI), le rejet d'eau de ballast demeure un vecteur important d'introduction d'ENI dans les écosystèmes aquatiques. Dans ce contexte, ce projet de recherche se penche sur les assemblages de dinoflagellés et de kystes de dinoflagellés du bras de mer Milne (Nunavut, Canada). En s'établissant dans un nouveau milieu, une ENI transforme les relations interspécifiques de la communauté d'accueil nuisant ainsi à la santé de l'écosystème. Ainsi, il est essentiel d'acquérir des informations de référence sur la diversité, la structure et la répartition des communautés indigènes de dinoflagellés de l'Arctique canadien afin de détecter d'éventuelles introductions d'ENI et d'évaluer les risques associés aux invasions biologiques. Un total de 32 taxa appartenant à 16 familles ont été identifiés, dont cinq sont des producteurs potentiels de toxines : *Alexandrium* cf. *catenella*, *Dinophysis acuminata*, *Gonyaulax* cf. *spinifera*, *Phalacroma rotundatum* et *Protoceratium reticulatum*. Parmi les 32 taxa identifiés, deux ont été observés dans l'Arctique canadien pour la première fois : *Gymnodinium* cf. *irregulare* et *Phalacroma pulchellum*. Les communautés de dinoflagellés de 2017 et de 2019 étaient significativement différentes. Un total de 12 taxa de kystes de dinoflagellés, tous indigènes à l'Arctique canadien, ont été identifiés. Bien qu'aucune preuve ne suggère irréfutablement que des ENI aient été introduites dans le bras de mer, les risques liés au rejet d'eau de ballast sont bien présents, et les changements climatiques et l'augmentation du trafic maritime dans l'Arctique ne font qu'aggraver ce problème.

Mots clés : Arctique canadien, Milne Inlet, Mary River Mine, Phytoplancton, Dinoflagellés, Kystes de dinoflagellés, Eau de ballast, Espèces non indigènes, Espèces toxiques

ABSTRACT

Between 2013 and 2023, the distance sailed by bulk carriers in the Arctic increased by more than 300% following the opening of the Mary River Mine. The Mary River Mine, an open-pit iron ore mine on Baffin Island, has been exploiting a high-grade iron ore deposit since 2014. Bulk carriers awaiting loading anchor in Milne Inlet, where they release their ballast water. Although international and Canadian standards exist to limit the introduction of non-indigenous species (NIS), the introduction of NIS into aquatic ecosystems through ballast water discharge remains a significant threat to ecosystem health. In this context, this research project focuses on the dinoflagellate and dinoflagellate cyst (dinocyst) assemblages in Milne Inlet (Nunavut, Canada). NIS can threaten native species as direct predators or competitors, potentially altering interspecies interactions. The introduction of NIS may, therefore, have significant ecological and economic impacts on affected regions. It is essential to acquire baseline information on the diversity, structure, and distribution of native dinoflagellate communities in the Canadian Arctic to detect potential NIS and assess the risks associated with biological invasions. A total of 32 dinoflagellate taxa from 16 families were identified, including five known to produce toxins: *Alexandrium* cf. *catenella*, *Dinophysis acuminata*, *Gonyaulax* cf. *spinifera*, *Phalacroma rotundatum*, and *Protoceratium reticulatum*. Among the 32 taxa identified, two were recorded in the Canadian Arctic for the first time: *Gymnodinium* cf. *irregulare* and *Phalacroma pulchellum*. The 2017 and 2019 dinoflagellate communities were significantly different. A total of 12 dinocyst taxa, all native to the Canadian Arctic, were identified. Although no irrefutable evidence suggests that NIS have been introduced into Milne Inlet, the risks associated with ballast water discharge are real, and climate change and the increase in maritime traffic in the Arctic only exacerbate this issue.

Keywords: Canadian Arctic, Milne Inlet, Mary River Mine, Phytoplankton, Dinoflagellates, Dinoflagellate cysts, Ballast water, Non-indigenous species, Toxic species

TABLE DES MATIÈRES

REMERCIEMENTS vii

RÉSUMÉ ix

ABSTRACT xi

TABLE DES MATIÈRES xiii

LISTE DES TABLEAUX xvii

LISTE DES FIGURES xix

LISTE DES ABRÉVIATIONS, DES SIGLES ET DES ACRONYMES xxi

INTRODUCTION GÉNÉRALE 1

1. TRAFIC MARITIME 1

2. ESPÈCES NON INDIGÈNES 2

3. EAUX DE BALLAST 4

4. DINOFLAGELLÉS 6

5. PHYCOTOXINES 8

6. MINE MARY RIVER 15

7. OBJECTIF DU MÉMOIRE 18

CHAPITRE 1 DINOFLAGELLATE AND DINOFLAGELLATE CYST ASSEMBLAGES IN MILNE INLET, NUNAVUT, CANADA 19

1.1 INTRODUCTION 19

1.2 MATERIALS AND METHODS 22

1.2.1 Study area 22
1.2.2 Sampling 23
1.2.3 Environmental data 24
1.2.4 Sample processing and analysis 25
1.2.5 Statistical analyses 27

1.3 RESULTS 29

1.3.1 Dinoflagellate assemblage composition 29
1.3.2 Comparison of the 2017 and 2019 dinoflagellate communities 38
1.3.3 Dinocyst assemblage composition 44
1.3.4 Comparison of the cyst-producing dinoflagellate and dinocyst communities 46

1.4 DISCUSSION 48

1.4.1 Newly recorded dinoflagellate taxa in the Canadian Arctic and insights into the dinoflagellate community 48
1.4.2 Observed differences between the 2017 and 2019 communities 51
1.4.3 Insights into the dinocyst community 52
1.4.4 Dinoflagellate and dinocyst taxa contributing to the differences in community composition 55
1.4.5 Ballast water and NIS 57

1.5 CONCLUSION 59

CONCLUSION GÉNÉRALE 61

ANNEXE I SAMPLING INFORMATION 63

ANNEXE II ENVIRONMENTAL DATA 65

ANNEXE III SEM MICROGRAPHS OF *PROTOPERIDINIUM PELLUCIDUM* 67

ANNEXE IV SEM MICROGRAPHS OF *TRIPOS ARCTICUS* 69

ANNEXE V SEM MICROGRAPHS OF FOUR *PROTOPERIDINIUM* TAXA 71

ANNEXE VI SEM MICROGRAPHS OF TWO PROBLEMATIC TAXA 73

ANNEXE VII MICROGRAPHS OF TWO *ALEXANDRIUM* TAXA 75

ANNEXE VIII MICROGRAPHS OF TWO *DINOPHYSIS* TAXA AND TWO *PHALACROMA* TAXA 77

ANNEXE IX MICROGRAPHS OF THE THREE MOST ABUNDANT DINOFLAGELLATE TAXA IDENTIFIED IN MILNE INLET AND TWO PROBLEMATIC TAXA 79

ANNEXE X PROBLEMATIC TAXA AND INSIGHTS ON SPECIFIC SPECIES 81

ANNEXE XI RESULTS OF THE VARIATION PARTITIONING ANALYSIS 85

ANNEXE XII LIST OF THE DINOFLAGELLATE TAXA IDENTIFIED AND THEIR CURRENT DISTRIBUTION IN THE ARCTIC 87

ANNEXE XIII SUPPLEMENTARY DATA ON THE BULK CARRIERS INVOLVED IN IRON ORE EXPORT 105

RÉFÉRENCES BIBLIOGRAPHIQUES.....109

LISTE DES TABLEAUX

Tableau 1. Phycotoxines produites dans les eaux canadiennes et espèces de dinoflagellés et de diatomées potentiellement toxiques 12

Tableau 2. Données sur l'exportation de minerai de fer et les vraquiers impliqués 17

Table 3. Results of the SIMPER analysis for the 2017 and 2019 dinoflagellate communities 40

Table 4. Results of the SIMPER analysis for the cyst-producing dinoflagellate and dinocyst communities 47

Table 5. Data on iron ore export and the bulk carriers involved 58

LISTE DES FIGURES

Figure 1. Map of the study area. The purple and pink dots represent the stations sampled.....24

Figure 2. Concentration (cells L^{-1}) of the 32 identified dinoflagellate taxa across all phytoplankton samples. Taxa marked with an asterisk (*) have a concentration of less than 1 cell L^{-1}. Stations are listed in the order in which they were sampled. Taxa sharing the same number belong to the same family: 1) Actiniscaceae, 2) Ceratiaceae, 3) Cladopyxidaceae, 4) Dinophysaceae, 5) Ensiculiferaceae, 6) Gonyaulacaceae, 7) Gymnodiniaceae, 8) Lessardiaceae, 9) Lingulodiniaceae, 10) Oxyphysaceae, 11) Peridiniales familia incertae sedis, 12) Protoceratiaceae, 13) Protodiniferaceae, 14) Protoperidiniaceae, 15) Pyrophacaceae, 16) Thoracosphaeraceae 30

Figure 3. Relative abundance (%) of the 32 identified dinoflagellate taxa across all phytoplankton samples. Taxa marked with an asterisk (*) have a relative abundance of less than 1%. Stations are listed in the order in which they were sampled. Taxa sharing the same number belong to the same family: 1) Actiniscaceae, 2) Ceratiaceae, 3) Cladopyxidaceae, 4) Dinophysaceae, 5) Ensiculiferaceae, 6) Gonyaulacaceae, 7) Gymnodiniaceae, 8) Lessardiaceae, 9) Lingulodiniaceae, 10) Oxyphysaceae, 11) Peridiniales familia incertae sedis, 12) Protoceratiaceae, 13) Protodiniferaceae, 14) Protoperidiniaceae, 15) Pyrophacaceae, 16) Thoracosphaeraceae.....31

Figure 4. SEM micrographs of potentially harmful dinoflagellate taxa identified in Milne Inlet. A. *Alexandrium* cf. *catenella*, ventral view. B. *Dinophysis acuminata*, right lateral view. C. *Gonyaulax* cf. *spinifera*, ventral view. D. *Phalacroma rotundatum*, right lateral view. E-F. *Protoceratium reticulatum*. E. Ventral view. F. Right lateral view 32

Figure 5. Relative abundance (%) of the 16 dinoflagellate families across all phytoplankton samples.....33

Figure 6. Relative abundance (%) of the 17 dinoflagellate genera across all phytoplankton samples.....34

Figure 7. Relative abundance (%) of the main dinoflagellate taxa and hierarchical clustering of the phytoplankton samples. Taxa marked with an asterisk (*) have a relative abundance of less than 1% 36

Figure 8. Map of the spatial distribution of the dinoflagellate and dinocyst assemblage zones near Ragged Island. A. Dinoflagellate assemblage zones; Zone Ib is not shown as it includes only station 3C. B. Dinocyst assemblage zones 37

Figure 9. Mean (± standard error) dinoflagellate concentrations (cells L^{-1}) and Shannon diversity indices 39

Figure 10. NMDS biplot 39

Figure 11. RDA biplot showing the relationships between the sampling stations and the environmental variables 42

Figure 12. RDA biplot showing the relationships between the dinoflagellate taxa identified in 2017 and 2019 and the environmental variables 43

Figure 13. Relative abundance (%) of the observed dinocyst taxa and hierarchical clustering of the surface sediment samples. Taxa marked with an asterisk (*) have a relative abundance of less than 1% 45

Figure 14. Micrographs of two newly recorded taxa in the Baffin Bay–Davis Strait ecoregion. A. *Gymnodinium* cf. *irregulare*. B. *Phalacroma pulchellum*. Scale bars are 20 µm 49

Figure 15. *Tripos* sp. 1, dorsal view 82

LISTE DES ABRÉVIATIONS, DES SIGLES ET DES ACRONYMES

BIM	Baffinland Iron Mines Corporation
BWE	Ballast water exchange
BWMS	Ballast water management system
CNER	Commission du Nunavut chargée de l'examen des répercussions
Convention	Convention internationale de 2004 pour le contrôle et la gestion des eaux de ballast et sédiments des navires – International Convention for the Control and Management of Ships' Ballast Water and Sediments
Dinocyst	Dinoflagellate cyst
DSIMS	Number of days between sea ice melt and sampling
EAN	Efflorescence algale nuisible
ENI	Espèce non indigène
IMO	International Maritime Organization
NIS	Non-indigenous species
NMDS	Non-metric multidimensional scaling
OMI	Organisation Maritime Internationale
PERMANOVA	Permutational multivariate analysis of variance
RDA	Redundancy analysis

$\mathbf{S_{bottom}}$	Bottom salinity
SEM	Scanning electron microscope
SGEB	Système de gestion des eaux de ballast
SIMPER	Similarity percentage
$\mathbf{S_{mean}}$	Mean salinity
$\mathbf{S_{surface}}$	Surface salinity
$\mathbf{T_{bottom}}$	Bottom temperature
$\mathbf{T_{mean}}$	Mean temperature
$\mathbf{T_{surface}}$	Surface temperature
VIF	Variance inflation factor
Z	Sampling depth
$\mathbf{Z_{Secchi}}$	Secchi depth

INTRODUCTION GÉNÉRALE

1. TRAFIC MARITIME

Plus de 90% des marchandises vendues mondialement sont transportées par navire (Organisation for Economic Co-operation and Development, 2020). Traditionnellement, ces navires empruntent les canaux de Panama et de Suez afin de passer d'un océan à l'autre et de sauver de nombreux jours de transit. Les changements climatiques et la diminution du couvert de glace qui s'ensuit pourraient toutefois accroître la navigabilité transarctique. On observe, en effet, une diminution spectaculaire du couvert de glace dans l'Arctique depuis plus de 50 ans (p. ex. Serreze et al., 2007; Stroeve et al., 2012; Walsh et al., 2017). Depuis les années 1960, dans l'Arctique canadien, la superficie du couvert de glace en été a diminué de 5 à 20% par décennie selon le secteur (Derksen et al., 2019; Tivy et al., 2011).

Les projections climatiques indiquent que ce phénomène n'est pas près de ralentir et que les routes transarctiques (c.-à-d. le passage du Nord-Ouest, le pont arctique, la route maritime du Nord et la route maritime transpolaire) se prêteront mieux à la navigation d'ici quelques années. Les navires de classe polaire 6 (c.-à-d. les navires conçus à des fins de navigation estivale et automnale dans la glace moyenne de première année incluant des sections de vieille glace) pourront naviguer ces routes 12 mois par an d'ici une quarantaine d'années (Min et al., 2022; Transport Canada, 2010). Les navires conçus pour la navigation en eau libre bénéficieront, quant à eux, d'une saison de navigation deux fois plus longue d'ici 2050 (Melia et al., 2016; Smith & Stephenson, 2013). Ces routes transarctiques permettront de sauver plusieurs jours de transit et de diminuer les coûts de transport (Liu & Kronbak, 2010; Somanathan et al., 2009). Deux de ces routes, le passage du Nord-Ouest et le pont arctique, empruntent les eaux canadiennes, ce qui fait de l'Arctique canadien une région clé quant au transport maritime transarctique (Mudryk et al., 2021).

Des saisons de navigation plus longues et de meilleures conditions de navigation s'accompagnent d'un développement économique en plein essor (Arctic Council, 2009). On observe, d'ailleurs, une augmentation notable du trafic maritime dans l'Arctique depuis une dizaine d'années. De 2013 à 2019, le nombre de navires empruntant le passage du Nord-Ouest a augmenté de 44% (de 112 à 160 navires) alors que la distance parcourue par ces navires a augmenté de 107% (de 2,98 à 6,17 millions de milles nautiques; Protection of the Arctic Marine Environment [PAME], 2021). De 2013 à 2023, le nombre de navires circulant dans l'Arctique a augmenté de 37%, passant de 1 298 à 1 782 navires (PAME, 2024a). La distance parcourue par ces navires a, quant à elle, plus que doublé, passant de 6,1 à 12,9 millions de milles nautiques. Outre les navires de pêche, la plupart des navires qui circulent dans l'Arctique sont destinés au transport des ressources naturelles (p. ex. vraquiers, chimiquiers, pétroliers), au tourisme (p. ex. bateaux de croisière, yachts, voiliers) ou au réapprovisionnement des communautés (Arctic Council, 2009; PAME, 2024b). De 2013 à 2023, la distance parcourue par les vraquiers a augmenté de plus de 300% à la suite de l'ouverture de la mine Mary River en 2014 et de la construction du port de Milne (PAME, 2024a). L'augmentation du trafic maritime dans l'Arctique a pour conséquence d'accroître les risques de contamination, incluant l'introduction d'espèces aquatiques non indigènes (p. ex. Miller & Ruiz, 2014; Ruiz & Hewitt, 2009; Sardain et al., 2019). Les eaux et les sédiments de ballast et l'encrassement biologique de la coque des navires sont d'importants vecteurs d'introduction (p. ex. Bailey et al., 2020; Gollasch, 2002).

2. ESPÈCES NON INDIGÈNES

Une espèce non indigène (ENI) est une espèce ayant établi des populations en dehors de son aire de répartition naturelle (Casas-Monroy et al., 2014; Chan et al., 2012). L'introduction d'ENI peut avoir d'importantes répercussions écologiques et économiques sur les régions touchées. Les écosystèmes de l'Arctique sont d'autant plus vulnérables à ces invasions biologiques puisqu'ils sont déjà déstabilisés par les changements climatiques

(Holbech & Pedersen, 2018). Lorsqu'elles s'établissent dans un nouveau milieu, les ENI transforment les relations interspécifiques (c.-à-d. compétition et prédation) de la communauté d'accueil (Millennium Ecosystem Assessment [MEA], 2005). L'introduction d'ENI perturbe l'équilibre du réseau trophique et y modifie le rôle de certaines espèces indigènes (Mack et al., 2000; Ricciardi et al., 2013; Simberloff et al., 2013). Une invasion biologique peut ultimement entraîner une perte de biodiversité dans l'écosystème touché (Mack et al., 2000; Reaser et al., 2007). De fait, l'introduction d'ENI est l'une des principales causes d'extinction dans les écosystèmes d'eau douce (MEA, 2005). À l'échelle du globe, on estime que les invasions biologiques causent plus de 1,4 billion de dollars US de dommages par an (Pimentel et al., 2001). Au Canada, les coûts associés aux invasions biologiques s'élèvent à plusieurs milliards de dollars par année (Colautti et al., 2006).

De 1965 à 2015, 2 209 espèces aquatiques non indigènes ont été identifiées et signalées à l'échelle mondiale (c-à-d. 1 442 espèces uniques appartenant à 17 phyla; Bailey et al., 2020). Cela correspond à environ 43 signalements par année ou à un signalement tous les neuf jours. Parmi les 2 209 espèces aquatiques non indigènes recensées, au moins 1 639 ont une ou plusieurs populations toujours actives à ce jour. Comme mentionné précédemment, les rejets d'eau et de sédiments de ballast et l'encrassement biologique de la coque des navires sont les principaux vecteurs d'introduction (p. ex. Bailey et al., 2020; Gollasch, 2002). L'aquaculture, le transport d'organismes vivants, le rejet d'organismes domestiques et la construction de canaux ou de voies navigables artificielles sont aussi d'importants vecteurs d'introduction (Bailey et al., 2020).

Les eaux de ballast sont utilisées afin de stabiliser les navires et d'ajuster leur tirant d'eau (Casas-Monroy et al., 2014). Elles sont donc indispensables puisqu'elles permettent aux navires de naviguer en toute sécurité (Transport Canada, 2019). De nombreux organismes pélagiques et benthiques sont involontairement pompés avec les eaux et les sédiments de ballast et sont transportés du port d'origine sur des centaines, voire des milliers, de kilomètres. Bien que les conditions dans les réservoirs de ballast soient peu propices à la vie et que la diversité spécifique et l'abondance diminuent drastiquement avec la longueur

du voyage, bon nombre d'organismes subsistent (p. ex. Gollasch et al., 2000; Simard et al., 2011). Les organismes qui subsistent sont vidangés dans le port d'accueil à la fin du voyage où ils pourraient s'établir et fragiliser l'écosystème. Afin de s'y établir, un organisme doit d'abord survivre au nouvel environnement et se reproduire (Casas-Monroy et al., 2014; Hallegraeff, 1998). Une espèce est considérée comme établi lorsque sa population se reproduit dans ce nouveau milieu.

3. EAUX DE BALLAST

Il existe toutefois des normes internationales et canadiennes afin de limiter l'introduction d'espèces aquatiques non indigènes lors de la vidange des eaux de ballast dans un port d'accueil. En 2004, les États membres de l'Organisation Maritime Internationale (OMI) ont adopté la Convention internationale de 2004 pour le contrôle et la gestion des eaux de ballast et sédiments des navires (Convention; International Maritime Organization [IMO], 2004). La Convention exige que chaque navire mette en place un plan de gestion des eaux de ballast certifié par l'OMI (Government of Canada, 2021). Ce plan de gestion doit respecter les normes D-1 et D-2 et un registre des eaux de ballast doit être tenu à bord. Conformément à la norme D-2, un navire est contraint de rejeter :

- moins de 10 organismes viables par mètre cube dont la taille minimale est supérieure ou égale à 50 µm;
- moins de 10 organismes viables par millilitre dont la taille est comprise entre 10 et 50 µm.

Le plan de gestion de chaque navire doit comprendre un système de gestion des eaux de ballast (SGEB) afin de neutraliser les organismes et de limiter leur rejet. De nombreux SGEB ont été développés et ceux-ci incluent de deux à trois étapes et un ou plusieurs procédés mécaniques, physiques ou chimiques afin d'assainir les eaux de ballast (Bailey et

al., 2022). La première étape chez la majorité des SGEB consiste à filtrer les eaux de ballast afin d'éliminer les organismes dont la taille excède 50 µm (Casas-Monroy & Bailey, 2021; IMO, 2015b). Le traitement principal implique typiquement un procédé physique ou chimique et constitue la deuxième étape du SGEB. Bien que de nombreuses technologies existent, l'électrochloration et les traitements aux ultraviolets sont les procédés les plus couramment utilisés (IMO, 2015b). Ces deux procédés d'assainissement sont inclus dans plus de 65% des SGEB en 2019 (DESMI Ocean Guard, 2020). Les SGEB utilisant un procédé d'assainissement chimique produisent des résidus et ceux-ci doivent être neutralisés avant que le navire ne puisse vidanger ses eaux de ballast dans le port d'accueil (Casas-Monroy & Bailey, 2021). La neutralisation des résidus constitue la troisième étape du SGEB et doit respecter la norme établie par l'OMI (IMO, 2015a).

Jusqu'à l'entrée en vigueur de la Convention en 2017, l'OMI exigeait que tous les navires échangent leurs eaux de ballast en pleine mer (Government of Canada, 2021). Depuis 2017, seuls les navires non équipés d'un SGEB doivent effectuer cette opération. Conformément à la norme D-1, un navire sans SGEB doit obligatoirement échanger 95% de ses eaux de ballast à au moins 200 milles nautiques de la côte et dans des eaux atteignant minimalement 200 m de profondeur. Les navires construits après le 8 septembre 2017 doivent être équipés d'un SGEB fonctionnel et certifié dès leur mise à l'eau. Les navires préexistants avaient jusqu'au 8 septembre 2024 pour se munir d'un SGEB. Au Canada, l'échange des eaux de ballast doit s'effectuer à au moins 200 milles nautiques des côtes canadiennes et dans des eaux atteignant une profondeur minimale de 2 000 m (Government of Canada, 2021). Les navires ayant comme destination un port canadien en eau douce doivent combiner l'échange des eaux de ballast au large à un SGEB afin d'être conformes aux normes canadiennes.

L'efficacité d'un SGEB dépend notamment du ou des procédés d'assainissement utilisés, du groupe taxonomique, du stade de vie des organismes, du pH, de la turbidité, de la température et de la salinité. L'électrochloration est plus efficace dans des eaux acides et salées alors qu'une forte turbidité diminue l'efficacité des traitements aux ultraviolets

(Batista et al., 2017; Sayinli et al., 2022). L'efficacité des traitements aux ultraviolets dépend également de la taille et de la morphologie des organismes (Gregg et al., 2009). Ce procédé neutralise plus efficacement les bactéries et les virus que les microalgues et est inefficace contre les kystes de dinoflagellés (Gregg et al., 2009; Montani et al., 1995; Oemcke, 1999).

Plusieurs études ont évalué l'efficacité des SGEB sur des navires opérationnels (p. ex. Bailey et al., 2022; 2023; Casas-Monroy & Bailey, 2021; Feng et al., 2023; Outinen et al., 2024). Ces études ont démontré que les eaux de ballast ne respectent pas toujours la norme D-2 de l'OMI. L'étude de Bailey et al. (2023) a cependant démontré que les SGEB peuvent neutraliser plus de 98% des organismes viables. L'échange des eaux de ballast en pleine mer pourrait, toutefois, augmenter l'efficacité des SGEB et permettre aux navires de systématiquement rejeter des eaux de ballast respectant la norme D-2 (p. ex. Bradie et al., 2021; Briski et al., 2013; 2015; Drake et al., 2020; Paolucci et al., 2015). Les eaux océaniques sont typiquement moins turbides et plus salées et se traitent ainsi plus efficacement que les eaux portuaires.

Quelques études se sont penchées sur l'efficacité de l'échange des eaux de ballast en pleine mer exigé par l'OMI entre 2004 et 2017 (p. ex. Dickman & Zhang, 1999; Simard et al., 2011). Ces études ont démontré que l'efficacité de l'échange varie selon le navire, les conditions océaniques, la technique utilisée et le groupe taxonomique. L'étude menée par Roy et al. (2012) a, d'ailleurs, démontré que l'échange des eaux de ballast n'influe pas significativement sur les concentrations de dinoflagellés toxiques et non indigènes.

4. DINOFLAGELLÉS

Les dinoflagellés forment un groupe de protistes biflagellés à la base de la chaîne alimentaire aquatique. Il existe environ 2 500 espèces de dinoflagellés et quelque 2 000 espèces fossiles (Hoppenrath et al., 2009; Taylor, 1987). Les dinoflagellés évoluent aussi bien en eau douce (10% des espèces) que dans les environnements marins (90% des espèces).

Parmi les espèces marines, environ 10% sont des espèces benthiques (Hoppenrath et al., 2009). Ces organismes occupent tous les environnements marins de la planète, des mers tropicales aux mers polaires. De nombreuses espèces sont, d'ailleurs, cosmopolites (Steidinger & Tangen, 1997).

Environ 50% des espèces de dinoflagellés sont autotrophes, auxotrophes (c.-à-d. des organismes photosynthétiques qui requièrent un ou plusieurs composés organiques externes — p. ex. vitamines) ou mixotrophes (c.-à-d. des organismes qui combinent autotrophie et hétérotrophie) et 50% sont hétérotrophes ou parasitiques (Gaines & Elbrächter, 1987; Hoppenrath et al., 2009; Taylor, 1987). On observe trois modes de nutrition chez les dinoflagellés hétérotrophes : la phagotrophie, la myzocytose et la nutrition par déploiement d'un pallium (p. ex. Gaines & Elbrächter, 1987; Schnepf & Elbrächter, 1992). La phagotrophie est l'absorption d'une cellule par phagocytose. La digestion de la cellule proie s'effectue à l'intérieur du dinoflagellé. Certaines espèces se nourrissent à l'aide d'un pédoncule, un appendice tubulaire qu'elles utilisent pour percer la cellule proie et en aspirer le contenu (c.-à-d. la myzocytose; Schnepf & Deichgräber, 1984). D'autres espèces interceptent les cellules proies à l'aide d'un pallium, une extension cytoplasmique qui engloutit la ou les cellules et entraîne la formation d'une vacuole (Gaines & Taylor, 1984). La digestion s'effectue dans la vacuole à l'extérieur du dinoflagellé. La myzocytose et la nutrition par déploiement d'un pallium permettent aux dinoflagellés d'absorber des cellules proies excédant leur propre taille. La nutrition par déploiement d'un pallium permet également aux dinoflagellés d'engloutir des chaînes de diatomées. De nombreuses espèces de dinoflagellés se nourrissent, d'ailleurs, exclusivement de diatomées. C'est le cas, notamment, des espèces du genre *Protoperidinium* (p. ex. Buskey, 1997; Jacobson & Anderson, 1986).

Les dinoflagellés ont un cycle de vie complexe qui comprend une phase haploïde ou végétative et une phase diploïde (Evitt, 1985). Durant la phase haploïde, les cellules se divisent par mitose (c.-à-d. reproduction asexuée). Une cellule se divise en deux cellules filles, puis deux en quatre, quatre en huit et ainsi de suite. Lorsque les conditions (p. ex.

température, lumière, concentrations en nutriments) sont favorables, la reproduction asexuée peut conduire à la formation d'efflorescences algales. Sous certaines conditions, ces cellules végétatives se transforment en gamètes. Deux à deux, ces gamètes fusionnent et produisent un zygote mobile diploïde, dit planozygote (c.-à-d. reproduction sexuée). Chez certaines espèces, le planozygote possède deux flagelles longitudinaux. Chez 13 à 16% des espèces, un kyste se forme à l'intérieur du planozygote (Head, 1996). La cellule se disloque et perd sa mobilité (Evitt, 1985). Le kyste, ou hypnozygote, sédimente et se dépose à la surface du sédiment. Le kyste agit alors comme une particule sédimentaire jusqu'à l'exkystement de la cellule. La paroi des kystes de dinoflagellés contient un biopolymère résistant appelé dinosporine (Versteegh et al., 2012). La dinosporine permet aux dinoflagellés de survivre dans les sédiments lorsque les conditions dans la colonne d'eau ne sont pas favorables à leur croissance (Bravo & Figueroa, 2014). Les kystes demeurent viables de quelques jours à plusieurs décennies (Lundholm et al., 2011), mais sont conservés non viables dans les archives sédimentaires pendant des millions d'années. Les kystes de dinoflagellés sont utilisés comme bioindicateurs dans de nombreuses études paléo-océanographiques afin de retracer les conditions océaniques passées (p. ex. de Vernal et al., 2000; Kunz-Pirrung et al., 2001). Lorsque les conditions environnementales sont favorables à la croissance des cellules, la cellule exkyste via l'archéopyle et rejoint la colonne d'eau (Evitt, 1985). La cellule nouvellement formée se divise par méiose et forme quatre cellules haploïdes qui se divisent à leur tour par mitose. La phase diploïde de la plupart des espèces ne comprend pas de phase d'enkystement. Chez ces espèces, le planozygote se divise en quatre cellules haploïdes par méiose.

5. PHYCOTOXINES

Parmi les quelque 5 000 espèces connues de phytoplancton marin, environ 2% produisent des phycotoxines (Smayda, 1997; Sournia et al., 1991). Chez l'humain, une intoxication survient lorsqu'un individu consomme un bivalve (p. ex. palourdes, huîtres,

moules) ou tout autre organisme (p. ex. poissons) contaminé. Un bivalve est un organisme suspensivore qui filtre son milieu et qui se nourrit des particules organiques et des organismes en suspension. Lorsqu'elles sont présentes dans le milieu, ces phycotoxines s'accumulent dans les estomacs et la chair des bivalves (Bates et al., 2020). Elles peuvent également s'accumuler dans le réseau trophique passant du phytoplancton au zooplancton, du zooplancton aux poissons et du poisson aux oiseaux et aux mammifères marins. Ces phycotoxines peuvent, dans certains cas, entraîner la mort de l'individu ou de l'animal ayant consommé un organisme contaminé (p. ex. Farabegoli et al., 2018; Visciano et al., 2016). Au Canada, on observe trois types d'intoxications : l'intoxication paralysante, l'intoxication amnésique et l'intoxication diarrhéique (voir **Tableau 1**). Ces trois types d'intoxications sont causés par trois phycotoxines distinctes : la saxitoxine (et ses dérivés), l'acide domoïque et l'acide okadaïque (et les dinophysistoxines). Une fois ingérées, ces phycotoxines perturbent la fonction normale de certaines cellules provoquant l'apparition de symptômes.

La saxitoxine est une neurotoxine produite par certaines espèces de dinoflagellés. Au Canada, trois espèces du genre *Alexandrium* sont d'intérêt : *Alexandrium acatenella*, *Alexandrium catenella* (espèce ici utilisée pour regrouper *Alexandrium catenella*, *Alexandrium fundyense* et *Alexandrium tamarense*) et *Alexandrium ostenfeldii* (Bates et al., 2020). Les symptômes apparaissent quelques minutes (et jusqu'à 10 heures) après consommation d'un organisme contaminé. Ces symptômes incluent une sensation de picotement et d'engourdissement de la bouche, des lèvres et des doigts, des maux de tête, des vertiges, une perte d'équilibre et de coordination et une faiblesse musculaire (p. ex. Acres & Gray, 1978). Ces symptômes neurologiques s'accompagnent généralement de symptômes gastro-intestinaux (diarrhée et vomissements). Dans les cas plus graves, une intoxication par phycotoxine paralysante entraîne une paralysie respiratoire et la mort. Au Canada, le premier cas d'intoxication paralysante a été documenté par le capitaine George Vancouver, en 1793 (Acres & Gray, 1978). Les symptômes décrits par le capitaine Vancouver s'apparentent à ceux d'une intoxication par phycotoxine paralysante et sont apparus après consommation de moules. Les premiers cas modernes ont été signalés en 1936, en Nouvelle-Écosse (Gibbard et al., 1939).

L'acide domoïque est produit par plusieurs espèces de diatomées du genre *Pseudo-nitzschia*. Quatre espèces sont d'intérêt au Canada : *Pseudo-nitzschia australis*, *Pseudo-nitzschia multiseries*, *Pseudo-nitzschia pseudodelicatissima* et *Pseudo-nitzschia seriata* (Bates et al., 2020). L'espèce *Halamphora coffeiformis* est une autre espèce potentiellement toxique retrouvée dans les eaux canadiennes. Les premiers symptômes sont de type gastro-intestinal (crampes abdominales, diarrhée et vomissements) et apparaissent quelques heures (et jusqu'à 24 heures) après consommation d'un organisme contaminé (p. ex. Todd, 1993; Trainer et al., 2008). Les symptômes neurologiques incluent désorientation, confusion, hallucinations et perte de mémoire. Dans les cas plus graves, une intoxication par phycotoxine amnésique entraîne ultimement la mort. En 1987, plus de 150 personnes tombent gravement malades et 3 personnes perdent la vie après avoir consommé des moules communes (*Mytilus edulis*) récoltées dans la baie de Cardigan à l'Île-du-Prince-Édouard (Bates et al., 1988; 1989; Subba Rao et al., 1988). Il s'agit des premiers cas répertoriés d'intoxication amnésique sur le territoire canadien.

L'acide okadaïque est une phycotoxine produite par quelques espèces de dinoflagellés des genres *Dinophysis*, *Phalacroma* et *Prorocentrum*. Au Canada, sept espèces sont d'intérêt : *Dinophysis acuminata*, *Dinophysis acuta*, *Dinophysis caudata*, *Dinophysis fortii*, *Dinophysis norvegica*, *Phalacroma rotundatum* et *Prorocentrum lima* (Bates et al., 2020). Les premiers symptômes apparaissent une trentaine de minutes (et jusqu'à quelques heures) après avoir consommé un aliment contaminé et incluent nausées, diarrhée, et vomissements (p. ex. Sosa & Tubaro, 2016; Tubaro et al., 2008). En 1990, 13 personnes ont développé des symptômes gastro-intestinaux après avoir consommé des moules récoltées près de Mahone Bay en Nouvelle-Écosse (Quilliam et al., 1993). Il s'agit des premiers cas d'intoxication diarrhéique documentés au Canada.

Entre 1988 et 2017, au Canada, 593 efflorescences algales nuisibles (EAN) ont été documentées (McKenzie et al., 2021). Parmi ces 593 évènements, 367 sont liés à une phycotoxine paralysante, 70 à une phycotoxine amnésique et 59 à une phycotoxine diarrhéique.

Bien qu'elles ne soient à l'origine d'aucune intoxication documentée, six autres groupes de phycotoxines sont surveillés par Santé Canada : les spirolides, les pinnatoxines, les gymnodimines, les pecténotoxines, les yessotoxines et les azaspiracides (Bates et al., 2020).

Tableau 1. Phycotoxines produites dans les eaux canadiennes et espèces de dinoflagellés et de diatomées potentiellement toxiques

Intoxication	**Phycotoxine**	**Concentration maximale**	**Taxa**
Intoxication paralysante	Saxitoxine (et ses dérivés)	0.8 µg g^{-1} dans les tissus comestibles	*Alexandrium acatenella* *A. catenella* *A. ostenfeldii*
Intoxication diarrhéique	Acide okadaïque (et les dinophysistoxines)	0.2 µg g^{-1} dans les tissus comestibles; 1 µg g^{-1} dans les tissus gastriques	*Dinophysis acuminata*[1] *D. acuta*[1] *D. caudata*[1] *D. fortii*[1] *D. norvegica*[1] *Phalacroma rotundatum*[2] *Prorocentrum lima*
Intoxication amnésique	Acide domoïque	20 µg g^{-1} dans les tissus comestibles	*Halamphora coffeiformis*[3] *Pseudo-nitzschia australis* *P. multiseries* *P. pseudodelicatissima*

			P. seriata
Aucune	Spirolides	Pas encore établie	*Alexandrium ostenfeldii*
Aucune	Pinnatoxines	Pas encore établie	*Vulcanodinium rugosum*[4]
Aucune	Gymnodimines	Pas encore établie	*Alexandrium ostenfeldii*[5]
			Karenia selliformis[4]
Aucune	Pecténotoxines	0.2 µg g^{-1} dans les tissus comestibles; 1 µg g^{-1} dans les tissus gastriques	*Dinophysis acuminata*
			Dinophysis spp.
Aucune	Yessotoxines	Pas encore établie	*Gonyaulax spinifera*[1]
			Lingulodinium polyedrum
			Protoceratium reticulatum
Aucune	Azaspiracides	Pas encore établie	*Amphidoma languida*[4]
			Azadinium dexteroporum[4]
			A. poporum[4]
			A. spinosum[4]

Modifié à partir de Bates et al. (2020). Les concentrations maximales proviennent de Santé Canada (2020). [1] Bien qu'il y soit présent, l'organisme n'est pas un producteur de toxines avéré dans les eaux canadiennes; [2] Des doutes subsistent quant à la nature toxinogène de cet organisme; [3] Produit de faibles quantités d'acide domoïque en culture; [4] N'a jamais été observé dans les eaux canadiennes; [5] Bien qu'il y soit présent, l'organisme n'est pas un producteur de gymnodimines avéré dans les eaux canadiennes.

Ces phycotoxines peuvent avoir d'importantes répercussions économiques lorsqu'elles sont produites au cœur d'une EAN (p. ex. Hoagland & Scatasta, 2006; Ritzman et al., 2018; Wessels et al., 1995). Une EAN se forme lorsque les populations de dinoflagellés, de diatomées et de plusieurs autres groupes de microalgues photosynthétiques se multiplient rapidement (Smayda, 1992). Lorsque les concentrations de phycotoxines sont trop élevées, les gouvernements se voient obligés de fermer les sites d'aquaculture, de suspendre la pêche et d'interdire la récolte de bivalves, ce qui entraîne d'importantes pertes économiques. Ces EAN entraînent également la mort de nombreux organismes sauvages et d'élevage. En 2000, en Nouvelle-Écosse, de nombreux saumons (*Salmo salar*) d'élevage périssent à la suite d'une EAN de l'espèce de dinoflagellé *Alexandrium catenella* (Cembella et al., 2002). Ce phénomène s'est reproduit en 2003 et 2004 dans la baie de Fundy (Bates et al., 2020; Martin et al., 2006). En 2015, une EAN de l'espèce de diatomée *Pseudo-nitzschia australis* sur la côte ouest du Canada et des États-Unis (de la Californie à l'Alaska) entraîne la fermeture des pêcheries, notamment de la pêche commerciale au crabe dormeur (*Metacarcinus magister*; McCabe et al., 2016). Les pertes économiques liées à cette fermeture de la pêche au crabe dormeur s'élèvent à près de 100 millions de dollars US (National Marine Fisheries Service, 2016; Ritzman et al., 2018). La récolte de moules sauvages est interdite depuis 1943 dans la baie de Fundy en raison de l'annualité des EAN dans la région (Medcof et al., 1947; Prakash et al., 1971). La récolte de bivalves est, de plus, périodiquement interdite partout au Canada lorsque les niveaux de phycotoxines s'élèvent au-dessus des concentrations maximales établies par Santé Canada (p. ex. Bates et al., 2020; Hamer et al., 2012). Les EAN peuvent aussi entraîner des répercussions sociales et économiques lorsque les activités récréotouristiques liées à la mer sont touchées (p. ex. Berdalet et al., 2016; Karlson et al., 2021).

Parmi les quelque 100 espèces de phytoplancton susceptibles de produire des phycotoxines, une trentaine d'espèces ont été documentées dans les eaux de l'Arctique canadien (Bates et al., 2020; McKenzie et al., 2021; Pućko et al., 2019; Schiffrine et al., 2024). Diverses phycotoxines ont, d'ailleurs, été détectées dans les tissus de bivalves récoltés dans la mer de Beaufort (Pućko et al., 2023). Ces phycotoxines sont donc activement

produites dans les eaux de l'Arctique canadien et s'accumulent dans le réseau trophique marin.

6. MINE MARY RIVER

La mine Mary River est une mine à ciel ouvert sur l'île de Baffin qui exploite un gisement de minerai de fer de haute qualité (Baffinland Iron Mines Corporation [BIMC], 2024). Ce gisement, portant le nom de Deposit No. 1, se situe à 160 km au sud-sud-ouest de la communauté de Pond Inlet (Mittimatalik) et est exploité par la compagnie Baffinland Iron Mines Corporation (BIM). L'exploitation du Deposit No. 1 devrait durer plus de 20 ans et l'exploitation de gisements supplémentaires est envisagée. Les activités de la mine pourraient donc se maintenir sur plusieurs décennies.

En 2012, le ministre des Affaires du Nord approuve la Phase 1 du projet de développement de la mine Mary River (BIMC, 2024). La Commission du Nunavut chargée de l'examen des répercussions (CNER) remet alors à BIM un certificat de projet leur permettant d'aller de l'avant avec la Phase 1 qui comprend la construction d'un chemin de fer de 150 km et d'un port dans le bras de mer Steensby. Ce certificat de projet permet également à BIM d'exporter 18 millions de tonnes de minerai de fer annuellement via ce même bras de mer. En 2013, les fluctuations du prix du minerai de fer contraignent BIM à modifier le projet de développement de la mine. BIM propose alors la Early Revenue Phase qui est approuvée par le ministre des Affaires du Nord en 2014. BIM est alors autorisé à exporter 4.2 millions de tonnes de minerai de fer annuellement via le bras de mer Milne. BIM lance ses opérations en 2014.

En 2018, BIM soumet à la CNER un document exposant ses plans pour la Phase 2 du projet de développement de la mine Mary River (BIMC, 2024). La Phase 2 inclut la construction d'un chemin de fer de 100 km reliant la mine au port de Milne. La Phase 2 permettrait également à BIM d'augmenter ses exportations à 12 millions de tonnes par an. En 2022, le ministre des Affaires du Nord rejette la Phase 2 sur recommandation de la CNER (Crown-Indigenous Relations and Northern Affairs Canada, 2022). Après un examen

approfondi, la CNER conclut que la Phase 2 a le potentiel de perturber gravement la faune et la flore du Nunavut. Une perturbation importante du milieu naturel pourrait mettre en péril la sécurité alimentaire des Inuits. La CNER estime que les effets potentiels sur l'environnement et les communautés inuites sont trop importants pour approuver la Phase 2 dans sa forme actuelle. BIM est toutefois autorisé, depuis 2018, à exporter 6 millions de tonnes de minerai de fer annuellement via le bras de mer Milne (BIMC, 2024).

En 2023, 6.1 millions de tonnes de minerai de fer ont été exportées vers les marchés internationaux (voir **Tableau 2**; BIMC, 2024). L'exportation de 6.1 millions de tonnes de minerai a nécessité 39 vraquiers totalisant 75 voyages aller-retour entre les ports européens et asiatiques et le Nunavut.

Tableau 2. Données sur l'exportation de minerai de fer et les vraquiers impliqués

Année	Millions de tonnes de minerai de fer exportées	Nombre de voyages aller-retour	Nombre de voyages impliquant l'échange et le traitement des eaux de ballast	Nombre de vraquiers	Nombre de vraquiers disposant d'un SGEB approuvé par l'OMI
2016	2.8	37	0	18	0
2017	4.1	56	0	23	0
2018	5.1	71	0	31	0
2019	5.9	82	23	41	9
2020	5.5	72	42	36	18
2021	5.6	74	56	39	28
2022	4.7	62	56	35	30
2023	6.1	75	74	39	38

Les informations présentées dans ce tableau proviennent de BIMC (2017; 2018; 2019; 2020; 2021; 2022; 2023a; 2024).

7. OBJECTIF DU MÉMOIRE

Dans ce contexte, ce projet de recherche vise à analyser les communautés de dinoflagellés et de kystes de dinoflagellés du bras de mer Milne (Nunavut, Canada). Trois objectifs spécifiques découlent de cette étude :

1. Accroître nos connaissances sur la diversité, la structure et la répartition des communautés indigènes de dinoflagellés de l'Arctique canadien afin de pallier les lacunes qui persistent, et ce notamment au niveau des relations kyste-thèque.

2. Déceler de potentielles espèces non indigènes et toxiques.

3. Comparer les communautés de dinoflagellés et de kystes de dinoflagellés afin d'identifier de potentielles dissemblances qui pourraient signifier que les conditions dans la colonne d'eau ont évolué depuis la formation des kystes ou que de nouvelles espèces ont récemment été introduites.

CHAPITRE 1
DINOFLAGELLATE AND DINOFLAGELLATE CYST ASSEMBLAGES IN MILNE INLET, NUNAVUT, CANADA

1.1 INTRODUCTION

Over 90% of the world's traded goods are transported by sea (Organisation for Economic Co-operation and Development, 2020). Ships traditionally transit through the Panama and Suez canals, but climate change and reductions in sea ice could increase trans-Arctic navigability (e.g., Melia et al., 2016; Mudryk et al., 2021; Smith & Stephenson, 2013). There has been a marked increase in maritime traffic in the Arctic over the past decade. From 2013 to 2023, the number of ships transiting through and operating in the Arctic increased by 37%, while the distance sailed by these ships more than doubled, rising from 6.1 to 12.9 million nautical miles (PAME, 2024a). Moreover, the distance sailed by bulk carriers increased by more than 300% following the opening of the Mary River Mine on northern Baffin Island in 2014. Increased maritime traffic in the Arctic heightens the risk of introduction of non-indigenous species (NIS; e.g., Miller & Ruiz, 2014; Ruiz & Hewitt, 2009; Sardain et al., 2019). Ballast water discharge and biofouling on ships' hulls are major vectors for the introduction of NIS into aquatic environments (e.g., Bailey et al., 2020; Gollasch, 2002).

A NIS is defined as a species that has established populations outside of its native range (Casas-Monroy et al., 2014; Chan et al., 2012). The introduction of NIS can have significant ecological and economic impacts on affected regions. Arctic ecosystems are especially vulnerable to these biological invasions, as they are already affected by climate change (Holbech & Pedersen, 2018). NIS may threaten native species as direct predators or competitors and may alter interspecies interactions (MEA, 2005). The establishment of NIS

may, therefore, disrupt the food web and modify the role of native species (e.g., Ricciardi et al., 2013; Simberloff et al., 2013). Biological invasions can ultimately lead to biodiversity loss in affected ecosystems (e.g., Mack et al., 2000; Reaser et al., 2007).

Ballast water is used to control the trim, stability, and structural stresses of ships (Casas-Monroy et al., 2014). Many pelagic and benthic organisms are unintentionally pumped along with ballast water and sediments and are transported over hundreds or even thousands of kilometres. Although conditions in ballast tanks are harsh and species abundance and diversity decrease sharply over time, many organisms survive (e.g., Gollasch et al., 2000; Simard et al., 2011). These organisms are released into a new environment at the end of the voyage, where they may become established and disrupt the ecosystem.

There are international and Canadian standards in place to minimize the introduction of NIS linked to ballast water discharge. The International Convention for the Control and Management of Ships' Ballast Water and Sediments (Convention) requires each ship to implement a ballast water management plan certified by the International Maritime Organization (IMO; Government of Canada, 2021; IMO, 2004). This management plan must comply with the D-1 and D-2 standards, and all ships must carry a ballast water record book onboard. In accordance with the D-2 standard, ships are required to release fewer than 10 viable organisms (i.e., living and functional eukaryotic or prokaryotic organisms) per cubic metre with a minimum size of 50 µm, and fewer than 10 viable organisms per millilitre measuring between 10 and 50 µm. Each ship's management plan must include a ballast water management system (BWMS). Most BWMS include two to three steps and one or more mechanical, physical, or chemical processes to treat ballast water (Bailey et al., 2022). The first step in most BWMS involves the filtration of the ballast water to remove organisms larger than 50 µm (Casas-Monroy & Bailey, 2021; IMO, 2015b). The main treatment typically involves a physical or chemical process and constitutes the second step of the BWMS. While many technologies exist, electrochlorination and ultraviolet treatments are the most commonly used (IMO, 2015b).

Until the Convention entered into force in 2017, the IMO required all ships to exchange their ballast water mid-ocean (Government of Canada, 2021). Pre-existing ships had until 8 September 2024 to be fitted with a BWMS. Ships bound for a Canadian freshwater port must combine mid-ocean ballast water exchange with a BWMS to meet Canadian standards. In Canada, ballast water exchange must occur at least 200 nautical miles from the nearest land and in waters with a minimum depth of 2 000 metres (Government of Canada, 2021)

Several studies have evaluated the effectiveness of BWMS on operational ships (e.g., Bailey et al., 2022; 2023; Casas-Monroy & Bailey, 2021; Feng et al., 2023; Outinen et al., 2024). These studies have shown that ballast water does not always meet the D-2 standard. However, Bailey et al. (2023) demonstrated that BWMS can neutralize more than 98% of viable organisms. Mid-ocean ballast water exchange could further enhance the effectiveness of BWMS, allowing ships to release ballast water that meets the D-2 standard (e.g., Bradie et al., 2021; Briski et al., 2013; 2015; Drake et al., 2020; Paolucci et al., 2015). Oceanic waters are typically saltier and less turbid than coastal waters, thereby enhancing the effectiveness of BWMS treatments.

Dinoflagellates are a group of biflagellate protists found in both freshwater (10% of species) and marine environments (90% of species; Hoppenrath et al., 2009; Taylor, 1987). These organisms are found in all marine environments, from tropical to polar seas, and many species are cosmopolitan (Steidinger & Tangen, 1997). Approximately 50% of dinoflagellate species are autotrophic, auxotrophic (i.e., photosynthetic organisms requiring one or more external organic compounds), or mixotrophic (i.e., organisms capable of both autotrophy and heterotrophy), and 50% are heterotrophic or parasitic (Gaines & Elbrächter, 1987; Hoppenrath et al., 2009; Taylor, 1987).

Dinoflagellates, along with diatoms and other groups of microalgae, can be introduced through ballast water and sediments discharge (e.g., Galil & Hülsmann, 1997; Hallegraeff & Bolch, 1991; 1992). Most autotrophic dinoflagellates perish within a few days of ballasting, whereas mixotrophic and heterotrophic dinoflagellates may survive by feeding on other organisms (Hallegraeff & Bolch, 1992; Rigby & Hallegraeff, 1994). Moreover, 13–16% of

dinoflagellate species produce a resting cyst as part of their life cycle (Head, 1996). Once the cell loses its motility, the cyst settles through the water column and remains dormant until excystment (Evitt, 1985). The organic wall of dinoflagellate cysts (dinocysts) contains dinosporin, a resistant biopolymer, allowing dinoflagellates to survive when conditions are unfavourable for vegetative growth (Bravo & Figueroa, 2014; Versteegh et al., 2012).

Some dinoflagellate species produce toxins that can accumulate in the stomachs and tissues of molluscan bivalves and fish (Bates et al., 2020). When these organisms are consumed by humans, the phycotoxins can cause paralytic shellfish poisoning, diarrhetic shellfish poisoning, neurotoxic shellfish poisoning, and ciguatera fish poisoning (e.g., Bates et al., 2020; Landsberg, 2002; Todd, 1985). Saxitoxin, okadaic acid, and their derivatives are the most common dinoflagellate-produced phycotoxins in Canada, although seven other groups of phycotoxins are also monitored by Health Canada (Bates et al., 2020). Toxic dinoflagellate species transported via ballast water, therefore, pose a serious threat to both public and ecosystem health (Hallegraeff & Bolch, 1991).

In this context, the main objective of this study was to analyze the dinoflagellate and dinocyst communities in Milne Inlet, as their composition may be influenced by the mining-related maritime activities occurring in the region. The specific objectives were to (1) assess the diversity, structure, and distribution of the native dinoflagellate communities; (2) identify potential non-indigenous and toxic species; and (3) compare the dinoflagellate and dinocyst communities to identify recent changes in community composition.

1.2 MATERIALS AND METHODS

1.2.1 Study area

Milne Inlet (72°15’ N, 80°30’ W; see **Figure 1**) is a fjord connected to Navy Board Inlet and Eclipse Sound on northern Baffin Island, Nunavut, Canada. Located within an Arctic climate region, it experiences complete land-fast ice cover during winter months, and

the ice-free season typically lasts about three months, from July to October (BIMC, 2024; Young et al., 2019). During the ice-free season, iron ore extracted from the Mary River Mine is transported via Milne Inlet to international markets. In 2023, a total of 6.1 million tonnes of iron ore were shipped from Milne Port (BIMC, 2024). This operation required 39 different bulk carriers, which completed 75 round trips between European and Asian ports and Nunavut. Bulk carriers awaiting loading anchor west of Ragged Island (Imiliit; 72°29'50.4" N, 80°02'32.4" W), located roughly 75 km north-northeast of Milne Port at the fjord's mouth. No more than three bulk carriers may anchor west of Ragged Island at any given time, and releasing ballast water at these anchorage locations is strictly prohibited (BIMC, 2023b). Sampling took place in the waters around Ragged Island to encompass these anchorage locations.

1.2.2 Sampling

Sampling was conducted onboard the RV *Nuliajuk* near Ragged Island between 16 and 25 August 2019 (see **Annex I**). Phytoplankton samples (11) were collected using a 20 µm plankton net (30 cm diameter). At every station, a heavy lead weight was fixed to the plankton net to ensure its vertical stability in the water column. The net was then lowered within 5 m of the bottom and hauled back to the surface at a constant speed of 1 m s^{-1}. The outside and inside walls of the net were then rinsed with local filtered seawater to ensure that nothing was stuck to the net and that all plankton were collected in the godend. The godend was unscrewed, and the content was transferred to a 250 mL plastic bottle. Twenty-seven millilitres of 37% w/v formaldehyde was added to the sample (the final solution contained 4% w/v formaldehyde). The volume was completed to 250 mL with local filtered seawater, and the bottle was sealed with electrical tape. Surface sediment samples (6) were collected using a Van Veen grab. The upper centimetre was subsampled using a spoon, and the sample was collected in a plastic bag. All samples were stored at 4°C in the dark to preserve the organisms.

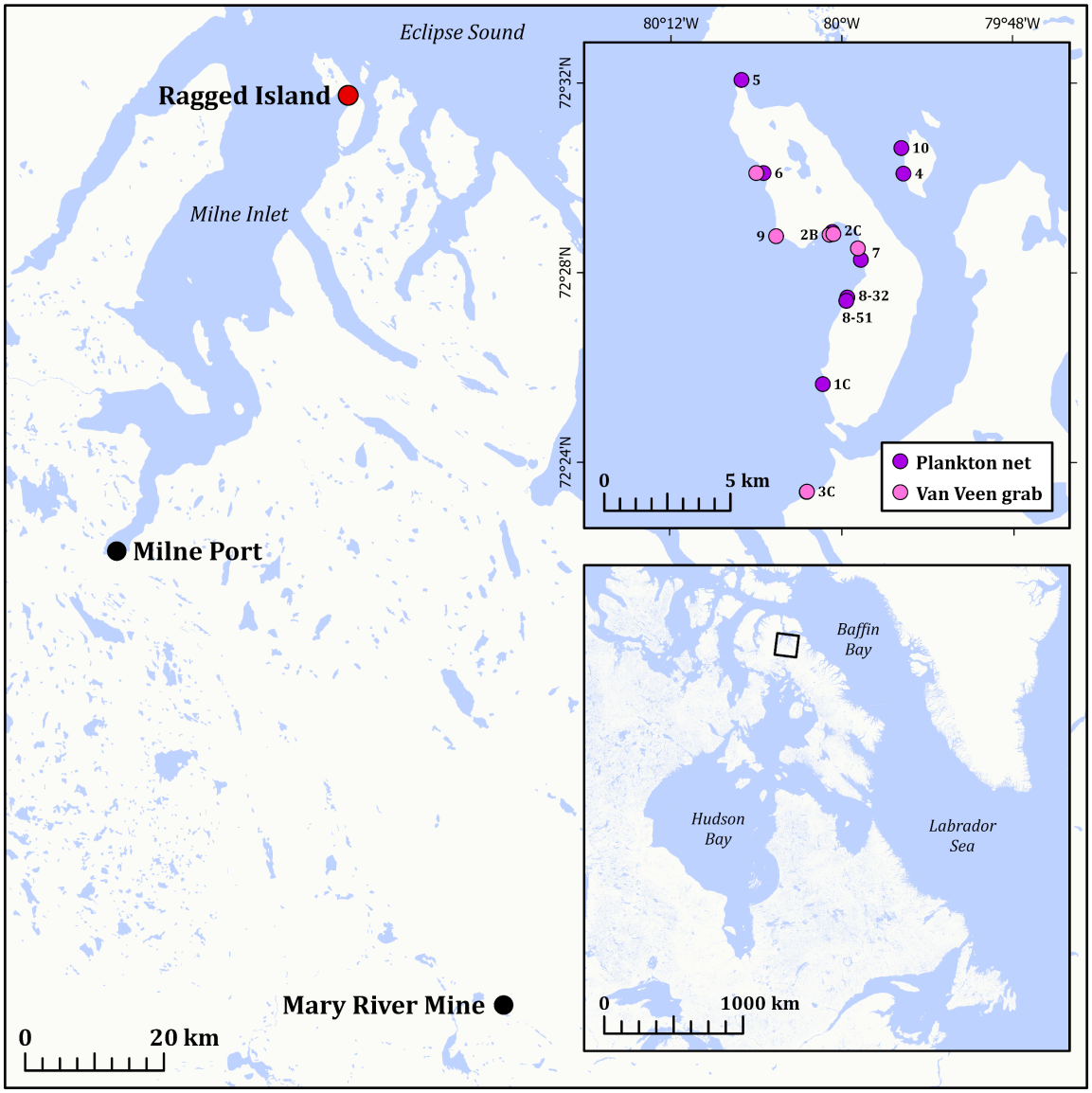


Figure 1. Map of the study area. The purple and pink dots represent the stations sampled

1.2.3 Environmental data

The number of days between sea ice melt and sampling (DSIMS) was calculated using the daily ice charts produced by the Canadian Ice Service (see **Annex II**). The study area was considered ice-free when the ice concentration was < 1/10. The daily ice charts were

extracted from the Canadian Ice Service Archive. The sampling depth (Z) is the depth to which the net was lowered and was measured using a graduated rope. The Secchi depth (Z_{Secchi}) is based on three replicates. Surface temperature ($T_{surface}$) and salinity ($S_{surface}$) as well as bottom temperature (T_{bottom}) and salinity (S_{bottom}) were measured using a Conductivity Temperature Depth probe (CastAway-CTD). The mean temperature (T_{mean}) and salinity (S_{mean}) of the water column were also calculated.

1.2.4 Sample processing and analysis

Phytoplankton samples were prepared using the Utermöhl method (Edler & Elbrächter, 2010; Utermöhl, 1958). Before filling the sedimentation chamber, the bottle was gently shaken for 3 minutes to homogenize the content. After homogenization, the sedimentation chamber was filled with a 2.973 mL subsample and sealed with a cover glass. In some cases, a 10 mL subsample was used due to low cell abundance. Sedimentation took place at room temperature, away from light sources, and required between 24 and 40 hours (Edler & Elbrächter, 2010; Hasle, 1978). Dinoflagellate concentrations are expressed in cells per litre (cells L^{-1}) and were calculated using the following equations:

$$(1)\ C = (\pi * B^2) * A * 1000$$

$$(2)\ I = D * {}^{1}/_{C} * {}^{E}/_{F} * {}^{G}/_{H}$$

where A is the depth to which the net was lowered (m), B is the net radius (m), C is the volume of water sampled (L), D is the number of cells counted per sample, E is the volume of the plastic bottle (L), F is the volume analyzed (L), G is the total area of the sedimentation chamber (mm^2), H is the area of the sedimentation chamber analyzed (mm^2), and I is the cell concentration (cells L^{-1}).

After homogenizing the sediment samples, a volume of 5 cm^3 was collected by displacement of an equal volume of distilled water in a graduated cylinder and weighted. The

subsample was then transferred to a 50 mL beaker. The beaker was placed in an ultrasonic bath for two minutes to break up sediment and organic matter particles before sieving. The subsample was then gently agitated, resulting in the suspension of fine particles. The supernatant was poured onto a 20 µm Nitex membrane to remove fine silt and clay. Any sand and coarse silt remaining in the beaker were discarded. This operation was repeated three times. The processed subsample was then transferred to a 15 mL conical bottom tube and centrifuged at 2 500 revolutions per minute for 10 minutes. The supernatant was removed, and the volume was adjusted to 10 mL. The subsample was homogenized for a few seconds on a vortex mixer, and a 1 mL aliquot was removed using a micropipette. The 1 mL aliquot was then dispensed into a Sedgewick-Rafter counting chamber. All manipulations surrounding the Sedgewick-Rafter counting chamber were carried out in accordance with LeGresley & McDermott (2010). A few drops of formaldehyde were added to the 15 mL conical bottom tube, which was then stored at 4°C for later analysis. A second sediment subsample was air-dried and weighted to calculate its water content. Concentrations are expressed in cysts per gram of dry sediment (cysts g^{-1} dry sed.) and were calculated using the following equations:

$$(3)\ X = {}^{Y}/_{K}$$

$$(4)\ U = V - W$$

$$(5)\ M = {}^{U}/_{W} * 100$$

$$(6)\ N = \frac{X * (100 - M)}{100}$$

$$(7)\ O = R * Q * {}^{1}/_{P} * {}^{1}/_{K} * {}^{1}/_{N}$$

where K is the volume of the first sediment subsample (cm^3), Y is the wet weight of the first sediment subsample (g), X is the wet weight of the first sediment subsample per cubic centimetre (g cm^{-3}), W is the wet weight of the second sediment subsample (g), V is the dry weight of the second sediment subsample (g), U is the weight of water in the sediment

(g), M is the percentage of water in the sediment, N is the dry weight of the sediment per cubic centimetre (g of dry sed. cm^{-3}), R is the number of cysts counted per sample, Q is the volume of solution prepared (mL), P is the volume of solution analyzed (mL), and O is the cyst concentration (cysts g^{-1} of dry sed.). The equations were modified from Lacasse (2011).

Dinoflagellates and dinocysts were observed using an inverted microscope (Nikon Eclipse TE-2000-U) at 200 × and 400 × magnification, and at least 400 cells or cysts were enumerated in each subsample. Identifications were based on several taxonomic references, including Hoppenrath et al. (2009), Bérard-Therriault et al. (1999), Throndsen et al. (2007) and Van Nieuwenhove et al. (2020a). Cysts and cells of uncertain identification were analyzed using a scanning electron microscope (SEM; SNE-4500M Plus). Samples were prepared according to the protocol described by Dhifallah et al. (2022). Cells were isolated and individually picked using a glass Pasteur pipet, then transferred onto 10 μm pore-size Nuclepore membranes. The samples were dehydrated through a graded ethanol series (from 10% to 100%) and critical-point dried in liquid CO_2 using a critical point dryer (Polaron E3000) and a water-recirculating heater and chiller (Quorum E4860). Once dry, the membranes were mounted on aluminum stubs and sputter-coated with gold-palladium using a sputter coater (Polaron SC7640 Sputter Coater).

1.2.5 Statistical analyses

The data collected in 2017 and analyzed by Dhifallah et al. (2022) were used for comparative purposes and to identify trends in community composition. The dinoflagellate concentration and the Shannon diversity index were calculated for each sample. Student's *t*-tests were performed on the mean dinoflagellate concentrations and the mean Shannon diversity indices to determine whether there were significant differences between 2017 and 2019. Additionally, a permutational multivariate analysis of variance (PERMANOVA) was used to test for differences in community composition across those two years. This analysis was based on the Hellinger-transformed data and the Hellinger distance (Legendre &

Gallagher, 2001; Legendre & Legendre, 2012; Rao, 1995). The Hellinger transformation converts raw count data into relative abundances, which are subsequently square-root transformed. The data were Hellinger-transformed prior to analysis to give greater weight to rare taxa. A non-metric multidimensional scaling (NMDS) and a similarity percentage (SIMPER) analysis were also performed. The SIMPER analysis showed which taxa were responsible for the observed differences in community composition.

A redundancy analysis (RDA) was used to model and analyze the relationships between dinoflagellate taxa and environmental variables. The RDA was based on the Hellinger-transformed data and the standardized environmental variables. Stations 7 and 8-32, sampled in 2019, were excluded from the analysis due to a lack of environmental data. DSIMS, Z, Z_{Secchi}, S_{mean}, $S_{surface}$, S_{bottom}, T_{mean}, $T_{surface}$, and T_{bottom} were the environmental variables used in the global RDA. A forward selection was carried out using the adjusted R^2 as the second stopping criterion (Blanchet et al., 2008). DSIMS, Z_{Secchi}, and S_{bottom} were found to be the best explanatory variables. A second RDA was carried out using DSIMS, Z_{Secchi}, and S_{bottom} as the environmental variables. Collinearity was assessed by computing variance inflation factors (VIF). A VIF > 5 suggests collinearity (Neter et al., 1996).

Cluster analyses were carried out on the dinoflagellate and the dinocyst assemblages. They were performed using the unweighted pair group method with arithmetic mean, and the clusters were computed based on the Hellinger distance.

A second PERMANOVA was performed to see if there were any differences between the cyst-producing dinoflagellate communities and the observed dinocyst communities. The analysis was based on the presence-absence data and the Jaccard distance (Legendre & Legendre, 2012). The analysis was limited to dinoflagellate and dinocyst taxa identified to the species level. *Polarella glacialis* was also excluded from the analysis as it is a sympagic species and, therefore, unlikely to be found in our plankton samples (Montresor et al., 1999; 2003). A SIMPER analysis was also performed. All statistical analyses were performed using R version 4.2.1.

1.3 RESULTS

1.3.1 Dinoflagellate assemblage composition

A total of 32 dinoflagellate taxa from 16 families were identified (see **Figures 2** and **3**). The most abundant dinoflagellate taxa observed were *Protoperidinium pellucidum* (17–39%; see **Annexes III** and **IX**), *Protoperidinium brevipes* (6.0–28%; see **Annexes V** and **IX**), *Protoperidinium mite* (7.7–17%; see **Annexes IX** and **X**), *Tripos arcticus* (2.6–18%; see **Annex IV**), *Pentapharsodinium dalei* (0.4–13%), *Protoceratium reticulatum* (0–19%; see **Figure 4**), *Dinophysis acuminata* (0.4–6.5%; see **Figure 4** and **Annex VIII**), *Peridiniella danica* (0.5–5.6%), and *Alexandrium* cf. *catenella* (0–5.4%; see **Figure 4** and **Annex VII**). Of the 32 taxa identified, five are known to produce toxins: *Alexandrium* cf. *catenella*, *Dinophysis acuminata*, *Gonyaulax* cf. *spinifera* (0–1.1%; see **Figure 4**), *Phalacroma rotundatum* (0–0.6%; see **Figure 4**, **Annexes VIII** and **X**), and *Protoceratium reticulatum*. These five species collectively accounted for between 3.7 and 24% of the assemblages.

Species of the family Protoperidiniaceae dominated the assemblages (47–78%; see **Figure 5**). The family Protoperidiniaceae includes, among others, the genera *Preperidinium* and *Protoperidinium*. Species of the family Protoperidiniaceae identified in 2019 belonged almost exclusively to the genus *Protoperidinium*. Therefore, the assemblages were primarily composed of *Protoperidinium* species (46–78%; see **Figure 6**). *Tripos* was the second most abundant genus (2.8–20%).

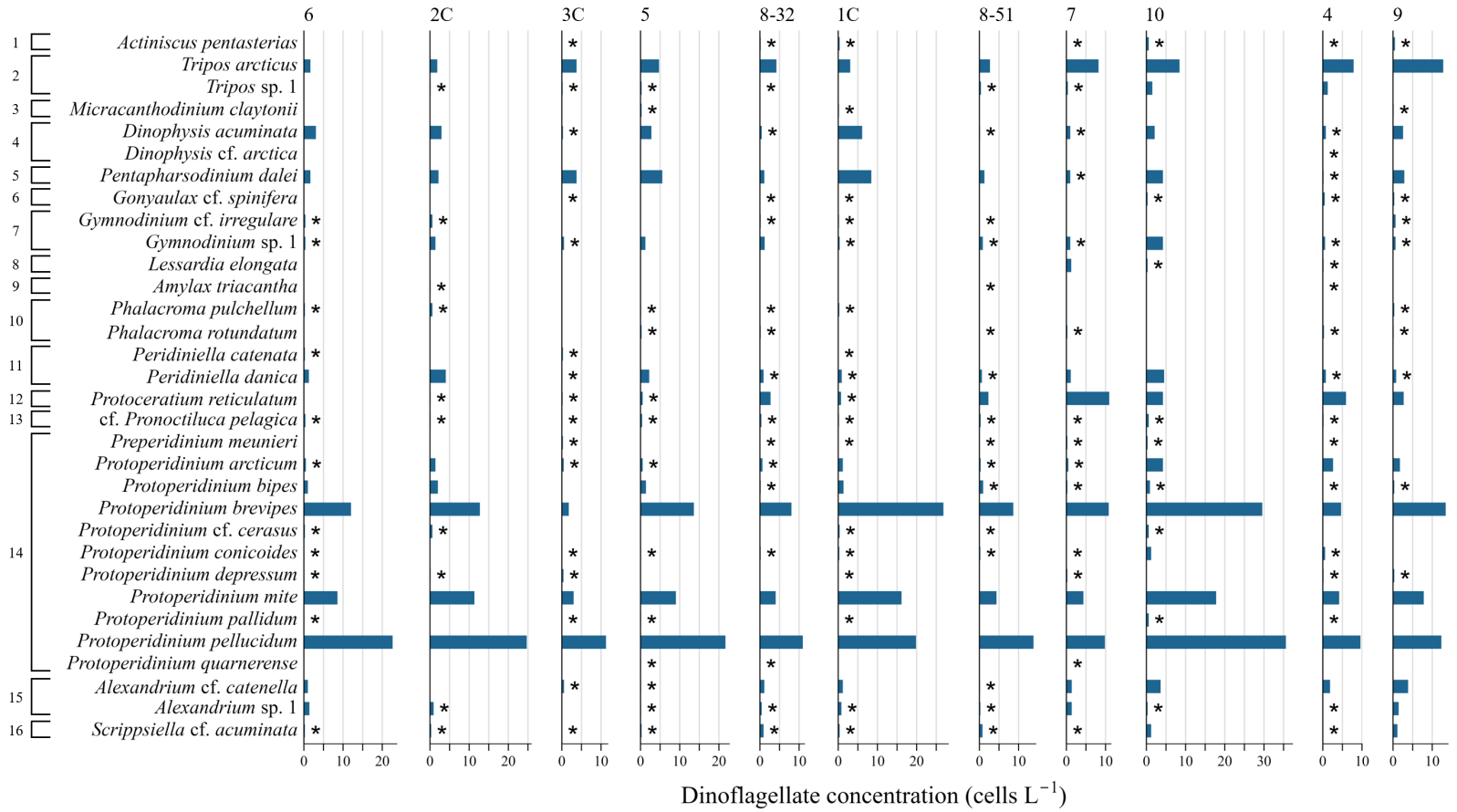


Figure 2. Concentration (cells L^{-1}) of the 32 identified dinoflagellate taxa across all phytoplankton samples. Taxa marked with an asterisk (*) have a concentration of less than 1 cell L^{-1}. Stations are listed in the order in which they were sampled. Taxa sharing the same number belong to the same family: 1) Actiniscaceae, 2) Ceratiaceae, 3) Cladopyxidaceae, 4) Dinophysaceae, 5) Ensiculiferaceae, 6) Gonyaulacaceae, 7) Gymnodiniaceae, 8) Lessardiaceae, 9) Lingulodiniaceae, 10) Oxyphysaceae, 11) Peridiniales familia incertae sedis, 12) Protoceratiaceae, 13) Protodiniferaceae, 14) Protoperidiniaceae, 15) Pyrophacaceae, 16) Thoracosphaeraceae

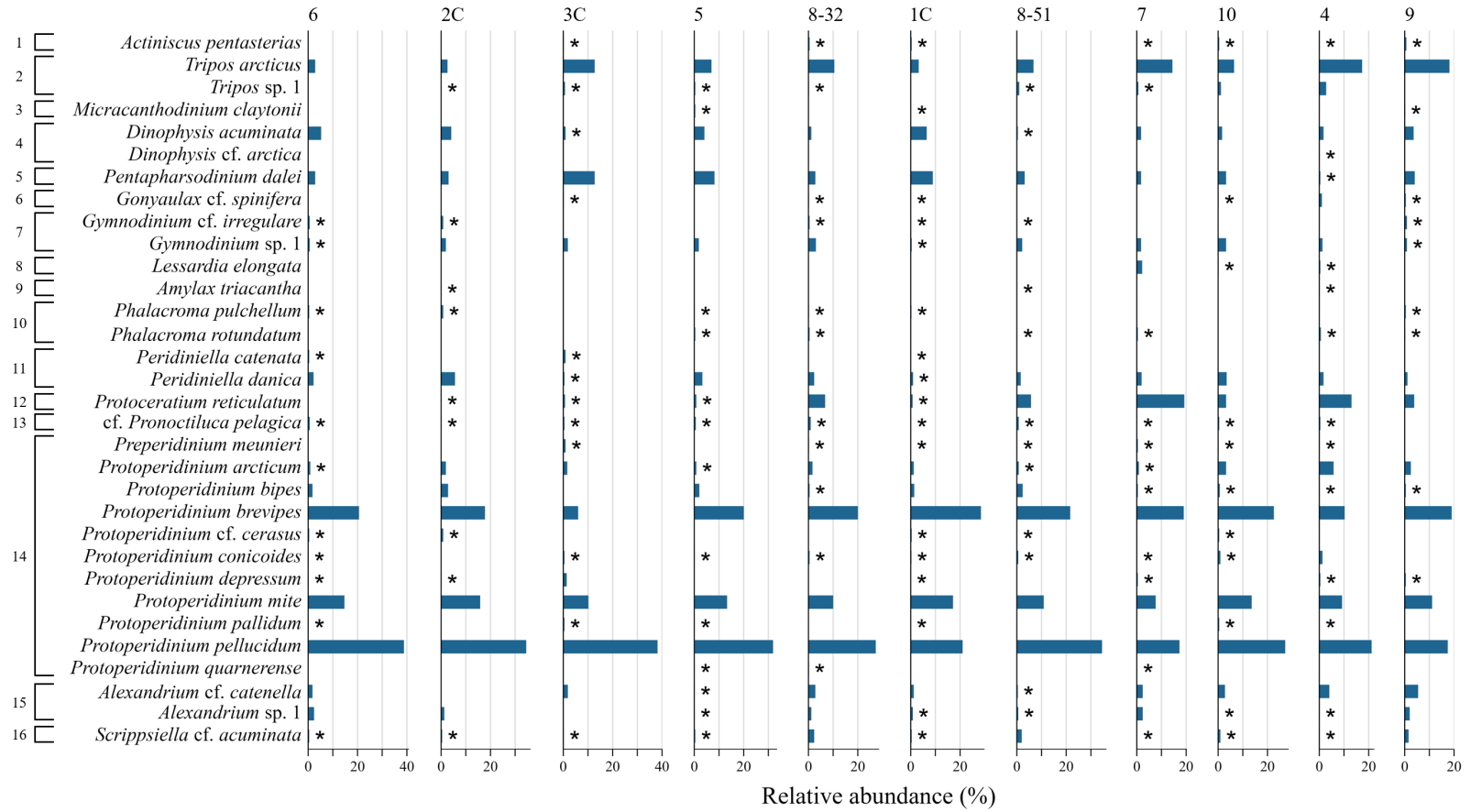


Figure 3. Relative abundance (%) of the 32 identified dinoflagellate taxa across all phytoplankton samples. Taxa marked with an asterisk (*) have a relative abundance of less than 1%. Stations are listed in the order in which they were sampled. Taxa sharing the same number belong to the same family: 1) Actiniscaceae, 2) Ceratiaceae, 3) Cladopyxidaceae, 4) Dinophysaceae, 5) Ensiculiferaceae, 6) Gonyaulacaceae, 7) Gymnodiniaceae, 8) Lessardiaceae, 9) Lingulodiniaceae, 10) Oxyphysaceae, 11) Peridiniales familia incertae sedis, 12) Protoceratiaceae, 13) Protodiniferaceae, 14) Protoperidiniaceae, 15) Pyrophacaceae, 16) Thoracosphaeraceae

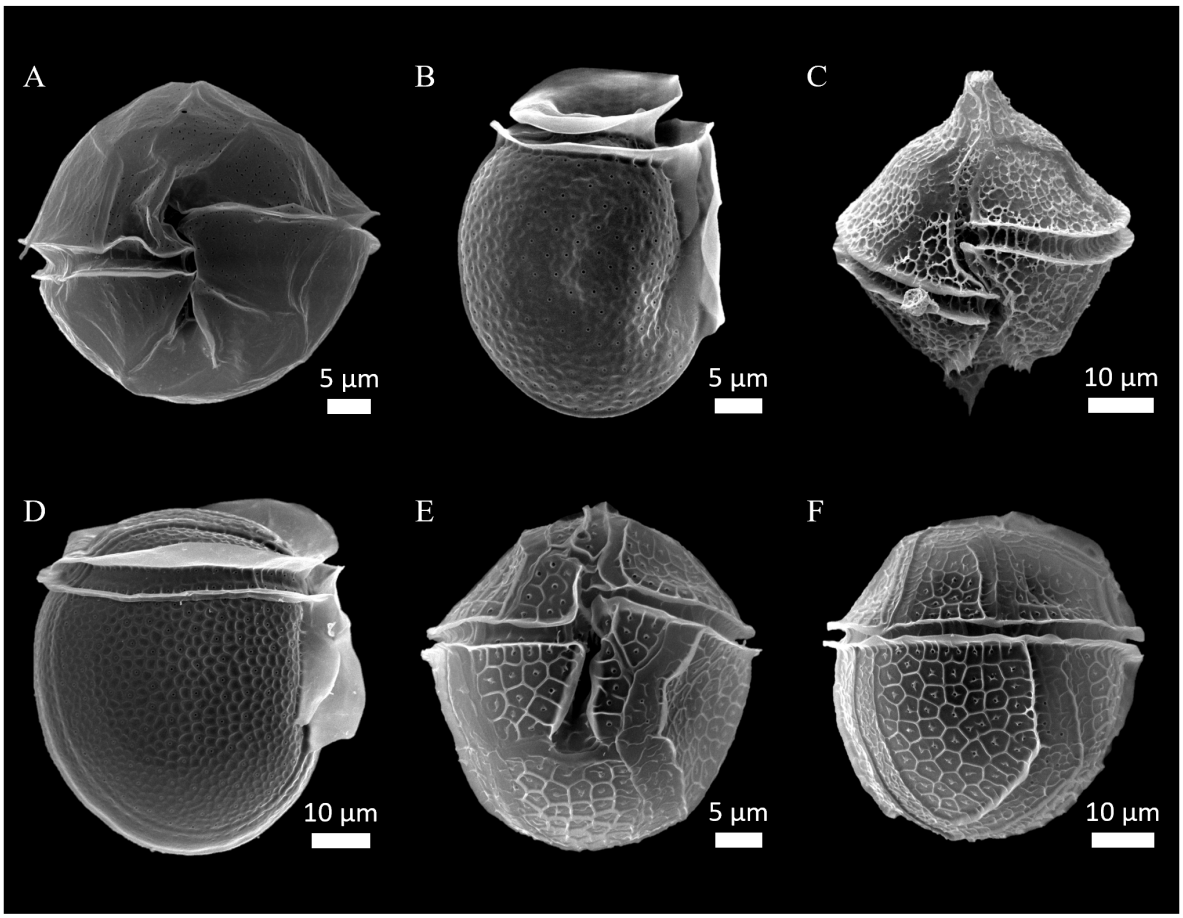


Figure 4. SEM micrographs of potentially harmful dinoflagellate taxa identified in Milne Inlet. A. *Alexandrium* cf. *catenella*, ventral view. B. *Dinophysis acuminata*, right lateral view. C. *Gonyaulax* cf. *spinifera*, ventral view. D. *Phalacroma rotundatum*, right lateral view. E-F. *Protoceratium reticulatum*. E. Ventral view. F. Right lateral view

Figure 5. Relative abundance (%) of the 16 dinoflagellate families across all phytoplankton samples

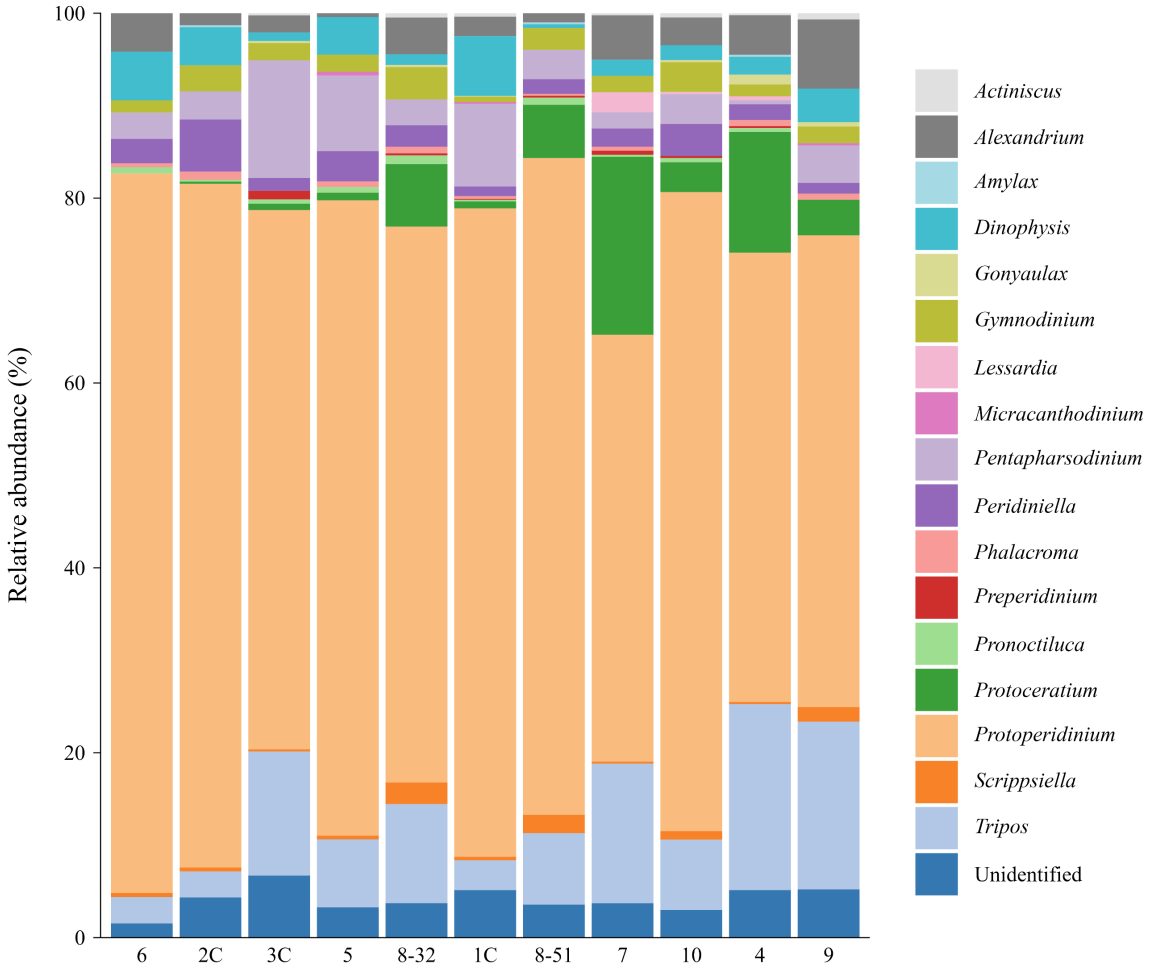


Figure 6. Relative abundance (%) of the 17 dinoflagellate genera across all phytoplankton samples

The cluster analysis revealed two distinct dinoflagellate assemblage zones (see **Figures 7** and **8A**). Zone I is further divided into two sub-zones: Ia and Ib. Zone Ia consists of eight samples collected between 16 and 25 August 2019 (23–32 DSIMS). The dinoflagellate concentrations in Zone Ia ranged from 40.0 to 131 cells L^{-1}, with a mean of 71.7 cells L^{-1}. *Protoperidinium pellucidum* (17–39%), *Protoperidinium brevipes* (18–28%), and *Protoperidinium mite* (10–17%) were the most abundant species. *Tripos arcticus* (2.6–18%) and *Protoceratium reticulatum* (0–6.8%) were found in relatively low abundances. The concentrations of toxin-producing taxa averaged 5.8 cells L^{-1}.

Zone Ib consists of a single sample (3C) collected on 18 August 2019 (25 DSIMS), in which the dinoflagellate concentration was 29.5 cells L^{-1}. This was the lowest dinoflagellate concentration observed in this study. The main taxa observed in this sub-zone were *Protoperidinium pellucidum* (38%), *Tripos arcticus* (13%), and *Pentapharsodinium dalei* (13%). *Protoceratium reticulatum* was not observed. The concentration of toxin-producing taxa reached 1.1 cells L^{-1}.

Zone II consists of two samples collected between 22 and 24 August 2019 (29–31 DSIMS). The dinoflagellate concentrations in Zone II averaged 51.1 cells L^{-1}. *Protoperidinium pellucidum* (17–21%), *Protoceratium reticulatum* (13–19%), *Tripos arcticus* (14–17%), and *Protoperidinium brevipes* (10–19%) were the dominant species. The mean concentration of toxin-producing taxa was highest in this zone, at 11.4 cells L^{-1}.

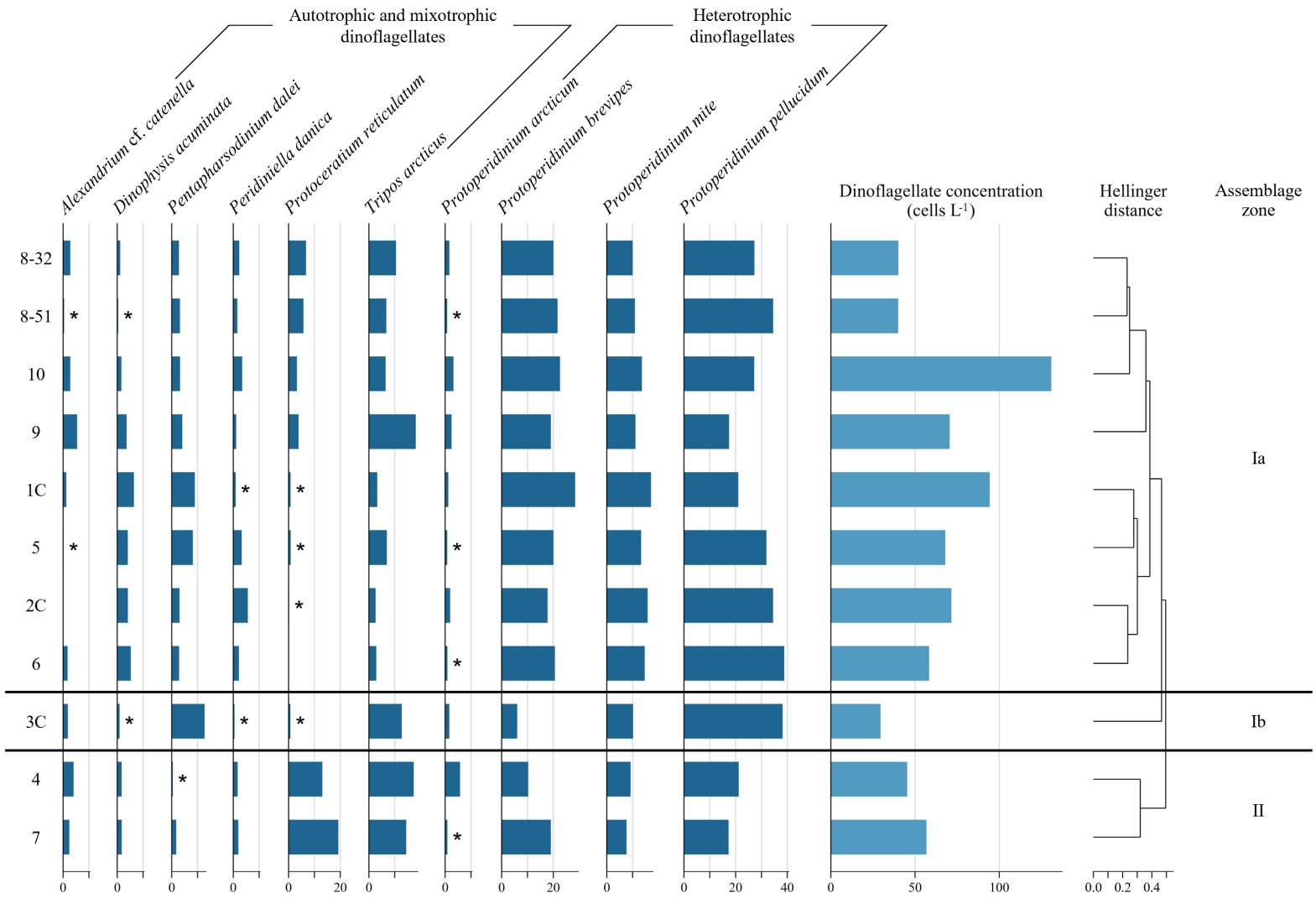


Figure 7. Relative abundance (%) of the main dinoflagellate taxa and hierarchical clustering of the phytoplankton samples. Taxa marked with an asterisk (*) have a relative abundance of less than 1%

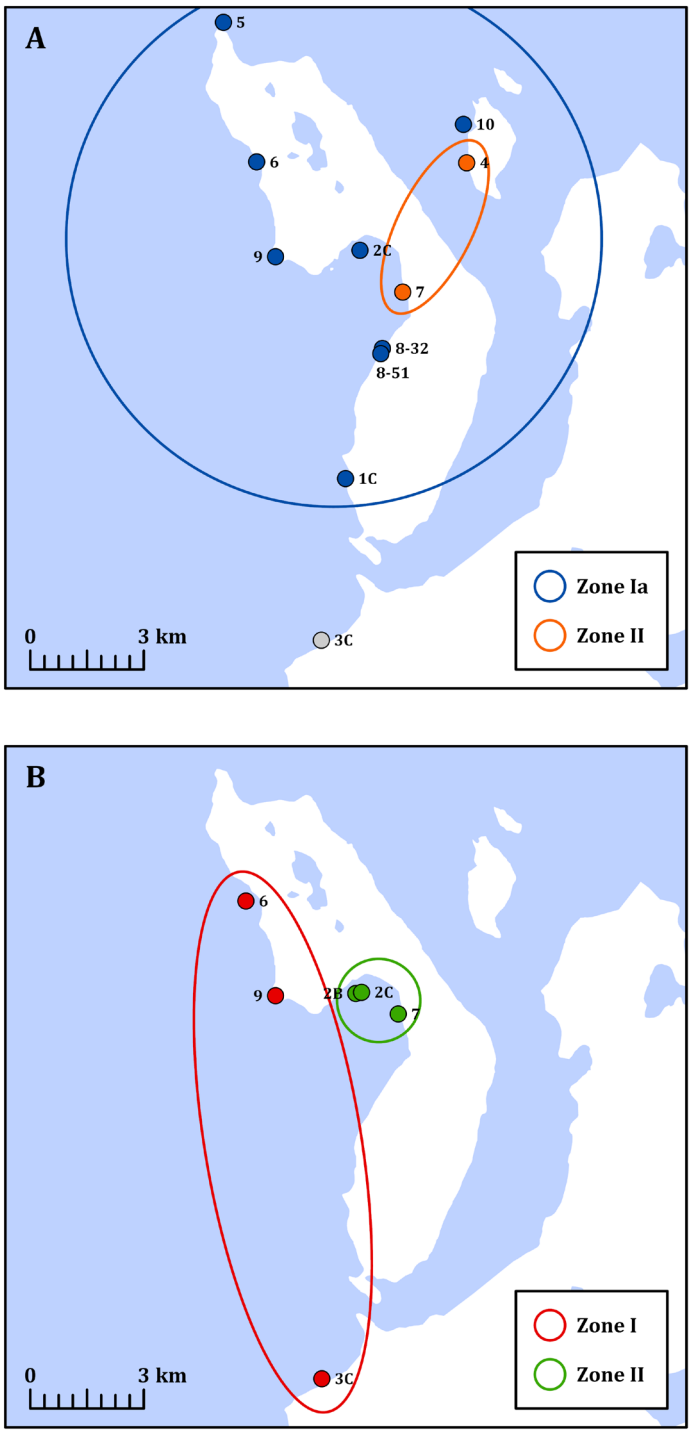


Figure 8. Map of the spatial distribution of the dinoflagellate and dinocyst assemblage zones near Ragged Island. A. Dinoflagellate assemblage zones; Zone Ib is not shown as it includes only station 3C. B. Dinocyst assemblage zones

1.3.2 Comparison of the 2017 and 2019 dinoflagellate communities

As part of their research project, Dhifallah et al. (2022) sampled 11 random sites within a 17 km radius of Milne Port in August 2017. Their key findings are summarized below. A total of 22 taxa from 9 families were identified. The main dinoflagellate taxa observed were *Tripos arcticus* (34–79%), *Alexandrium* cf. *catenella* (3.6–14%), *Protoperidinium brevipes* (1.0–34%), *Tripos longipes* (0–14%), *Tripos lineatus* (1.4–14%), *Preperidinium meunieri* (0–8.7%), *Protoperidinium pellucidum* (0.8–4.4%), and *Protoperidinium ovatum* (0–5.8%). *Alexandrium* cf. *catenella*, *Dinophysis acuminata* (0–0.7%), *Phalacroma rotundatum* (0–2.0%), and *Protoceratium reticulatum* (0–0.3%) are known to produce toxins.

Mean concentrations significantly differed between sampling years (*t*-test; p-value ≤ 0.0001; see **Figure 9**). In 2017, the dinoflagellate concentrations ranged between 1.9 and 21.0 cells L^{-1}, with a mean concentration of 10.7 cells L^{-1}. The dinoflagellate concentrations in 2019 ranged between 29.5 and 131 cells L^{-1}, averaging 64.1 cells L^{-1}. Mean diversity indices also significantly differed between sampling years (*t*-test, p-value ≤ 0.0001; see **Figure 9**). The diversity indices ranged between 0.96 and 1.72 in 2017 and 1.94 and 2.35 in 2019. The mean diversity indices were 1.34 in 2017 and 2.13 in 2019.

Dinoflagellate communities significantly differed in Milne Inlet in 2017 and 2019 (PERMANOVA; p-value ≤ 0.001). The NMDS showed that the stations clustered by sampling year, supporting the PERMANOVA results (see **Figure 10**). Five species accounted for 40% of the differences observed between sampling years (see **Table 3**). *Tripos arcticus* was the most abundant species in 2017. While it was also relatively abundant in 2019, it was observed in lower abundances. *Tripos lineatus* was not observed in 2019 but reached relatively high abundances in 2017. *Protoperidinium pellucidum*, *Protoperidinium brevipes*, and *Protoperidinium mite* were the most abundant species in 2019. *Protoperidinium mite* was not observed in 2017 but reached high abundances in 2019, whereas *Protoperidinium pellucidum* and *Protoperidinium brevipes* were observed in 2017, yet in relatively low abundances.

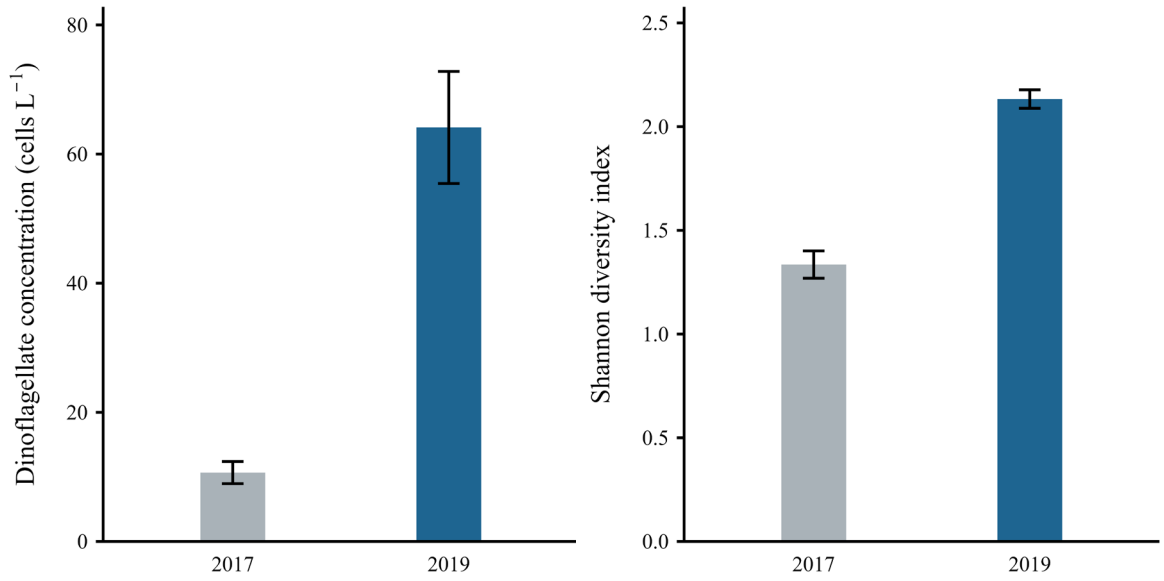


Figure 9. Mean (± standard error) dinoflagellate concentrations (cells L^{-1}) and Shannon diversity indices

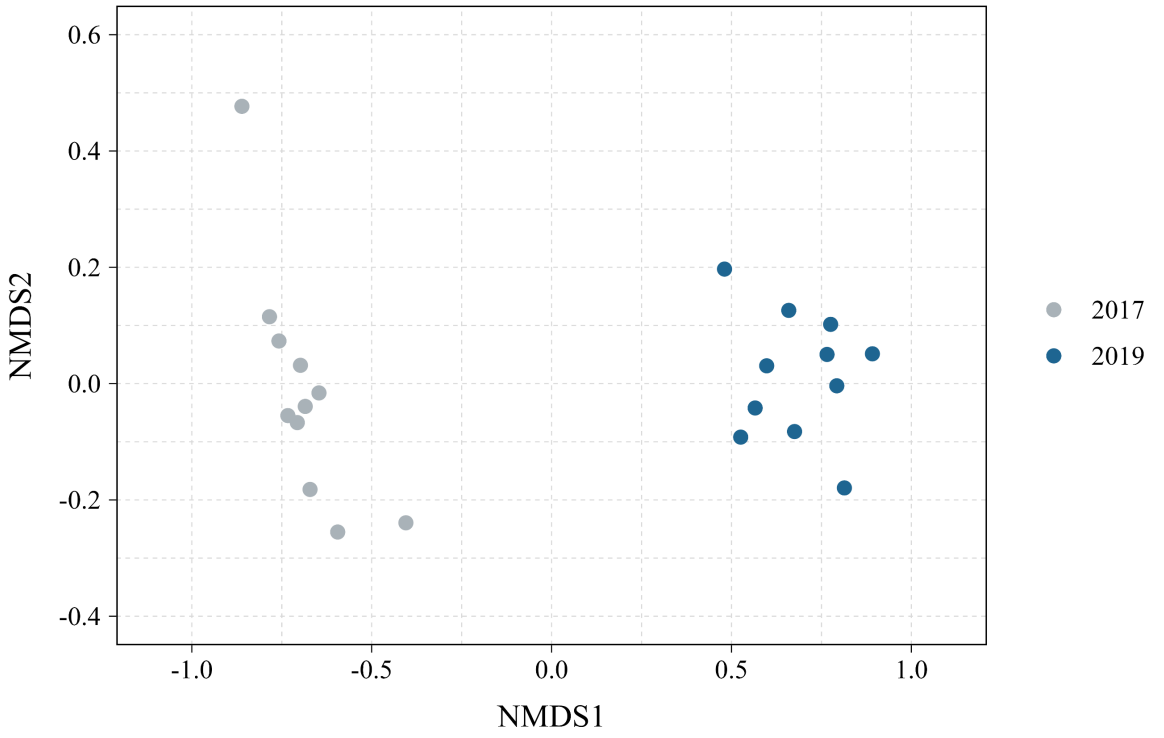


Figure 10. NMDS biplot

Table 3. Results of the SIMPER analysis for the 2017 and 2019 dinoflagellate communities

	Average dissimilarity (%)	**Standard deviation (%)**	**Individual contribution (%)**	**Cumulative contribution (%)**
Tripos arcticus	8.353	2.246	12.0	12.0
Protoperidinium pellucidum	6.158	1.499	8.8	20.8
Protoperidinium mite	5.722	0.793	8.2	29.0
Protoperidinium brevipes	3.776	1.576	5.4	34.4
Tripos lineatus	3.688	1.180	5.3	39.7
Tripos longipes	3.354	1.767	4.8	44.5
Pentapharsodinium dalei	3.339	1.377	4.8	49.3
Protoceratium reticulatum	2.869	2.082	4.1	53.4
Alexandrium cf. *catenella*	2.546	1.366	3.6	57.0
Dinophysis acuminata	2.496	1.070	3.6	60.6
Peridiniella danica	2.361	0.759	3.4	64.0
Protoperidinium arcticum	2.164	0.687	3.1	67.1
Preperidinium meunieri	2.151	1.160	3.1	70.2
Gymnodinium sp. 1	2.090	0.566	3.0	73.2
Protoperidinium ovatum	1.596	1.024	2.2	75.4
Tripos fusus	1.585	0.234	2.3	77.7
Alexandrium sp. 1	1.475	0.812	2.1	79.8
Protoperidinium bipes	1.435	0.874	2.1	81.9
Scrippsiella cf. *acuminata*	1.367	0.607	1.9	83.8
Tripos sp. 1	1.036	0.802	1.5	85.3
cf. *Pronoctiluca pelagica*	1.026	0.427	1.5	86.8
Protoperidinium depressum	0.941	0.586	1.3	88.1
Protoperidinium conicoides	0.916	0.536	1.3	89.4

A lower standard deviation indicates that the taxon consistently contributed to the observed differences across all samples (Clarke, 1993).

DSIMS, Z_{Secchi}, and S_{bottom} explained 61% (adjusted $R^2 = 0.612$) of the variation in community composition (see **Figures 11** and **12**). RDA1 explained 39% of the total variation and was significant (p-value ≤ 0.001). DSIMS and Z_{Secchi} were also significant (p-value ≤ 0.05). DSIMS, $Z_{Secchi,}$ and S_{bottom} were not collinear (VIF < 5) and were positively correlated with RDA1 (r = 0.79, r = 0.78, and r = 0.70, respectively). DSIMS, $Z_{Secchi,}$ and S_{bottom} explained 7.5, 6.5, and 4.2% of the total variation in community composition, respectively (see **Annex XI**). Samples were collected 14–22 days after the sea ice melt in 2017 (18.0 ± 0.8 DSIMS) and more than three weeks after in 2019 (27.4 ± 0.9 DSIMS). Z_{Secchi} was 5.93 ± 0.36 m in 2017 and 9.95 ± 0.72 m in 2019, meaning the water was less turbid in 2019 than in 2017. S_{bottom} was 27.1 ± 0.7 in 2017 and 30.3 ± 0.3 in 2019. Most taxa were positively related to DSIMS, $Z_{Secchi,}$ and S_{bottom}. However, *Tripos arcticus*, *Tripos lineatus*, *Tripos longipes*, and *Alexandrium* cf. *catenella* were negatively related to the explanatory variables.

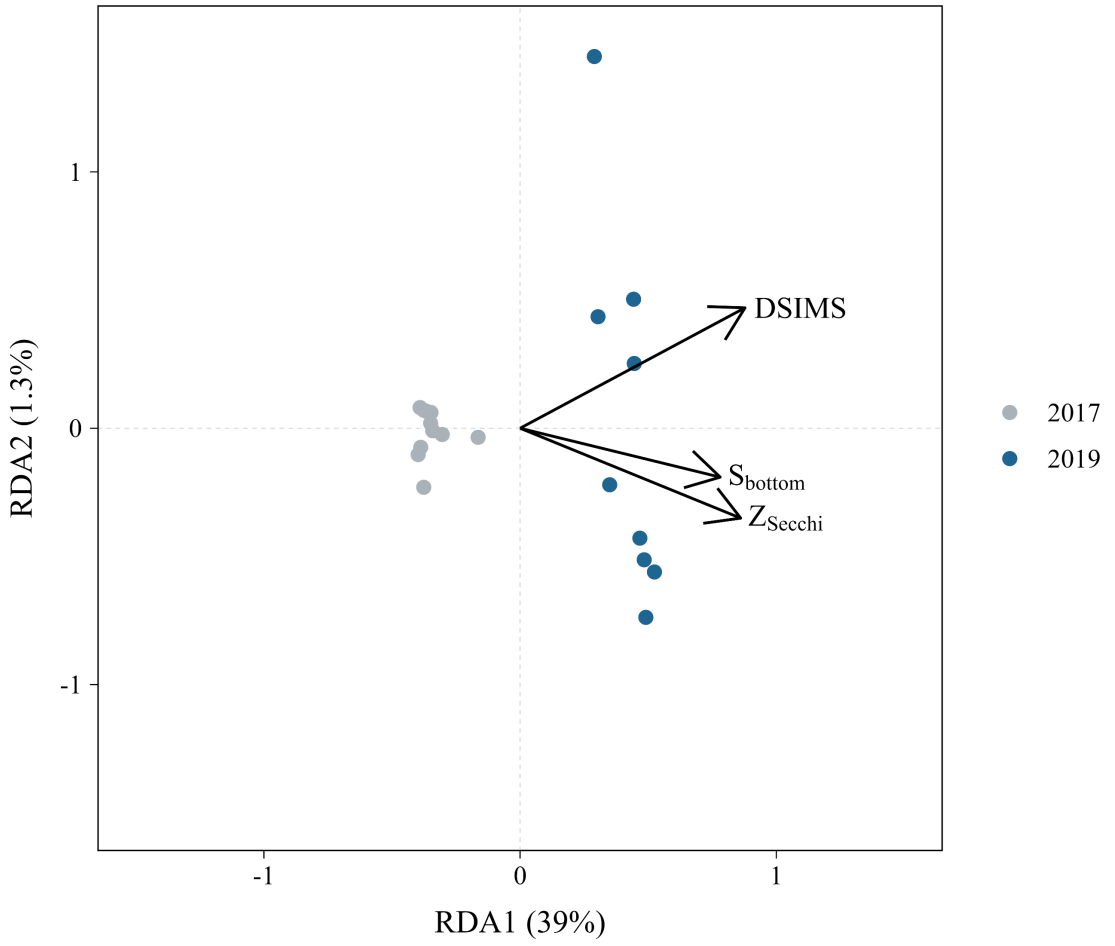


Figure 11. RDA biplot showing the relationships between the sampling stations and the environmental variables

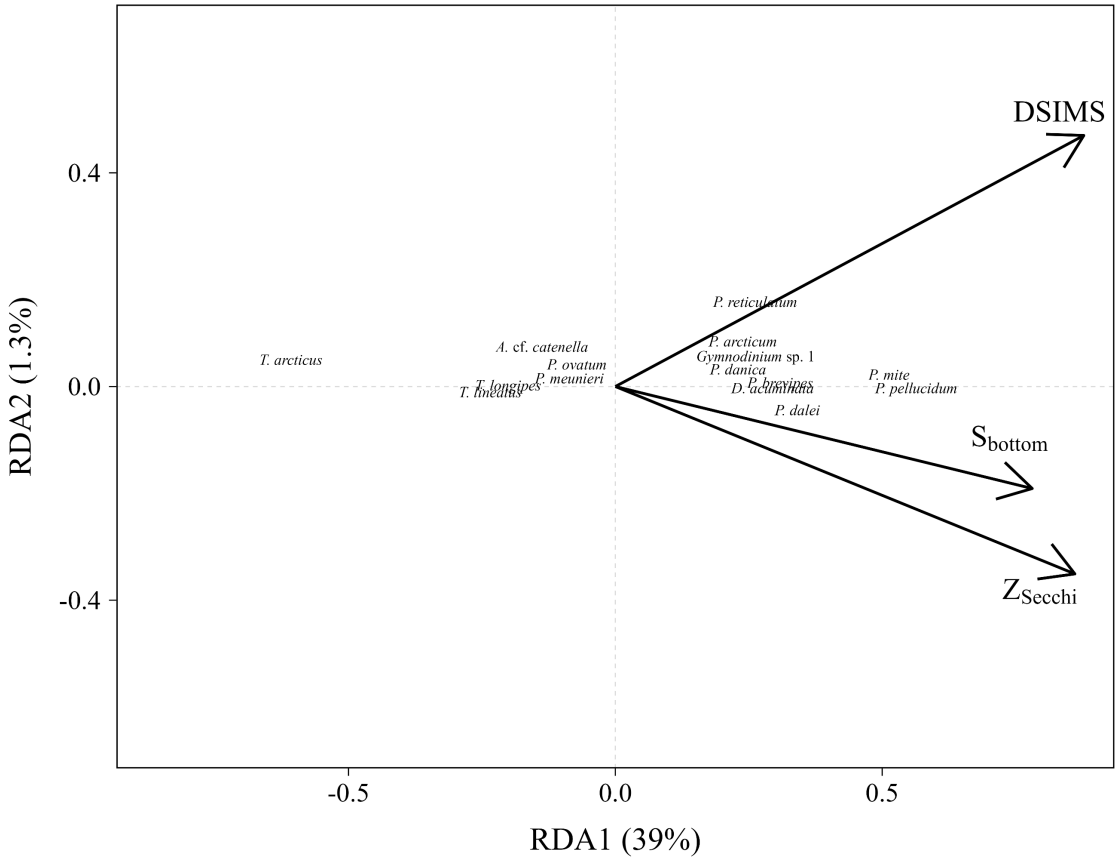


Figure 12. RDA biplot showing the relationships between the dinoflagellate taxa identified in 2017 and 2019 and the environmental variables

1.3.3 Dinocyst assemblage composition

A total of 12 dinocyst taxa were identified (see **Figure 13**). Dinocyst concentrations ranged between 212 and 807 cysts g^{-1} of dry sed., with a mean of 501 cysts g^{-1} of dry sed. The heterotrophic dinocyst taxa observed were *Brigantedinium* spp. (22–58%), *Islandinium minutum* (10–50%), *Brigantedinium simplex* (8.7–12%), *Brigantedinium cariacoense* (2.9–4.4%), *Dubridinium caperatum* (0–8.9%), *Islandinium*? *cezare* (0–4.0%), cysts of *Polykrikos*? sp. of Kunz-Pirrung (1998; 0.3–2.3%), and *Selenopemphix quanta* (<1%). The autotrophic taxa observed were *Operculodinium centrocarpum* (2.7–19%), cyst of *Pentapharsodinium dalei* (0.6–7.7%), *Spiniferites elongatus* (0.3–1.2%), and cyst of *Polarella glacialis* (<1%). The autotrophic to heterotrophic ratios ranged between 0.064 and 0.32, revealing a clear dominance of heterotrophic dinocysts.

The cluster analysis grouped the sediment samples into two zones (see **Figures 8B** and **13**). Zone I consists of three samples: 3C, 6, and 9. The dinocyst concentrations ranged between 212 and 795 cysts g^{-1} of dry sed. and averaged 445 cysts g^{-1} of dry sed. The main dinocyst taxa observed in Zone I were *Islandinium minutum* (33–50%) and *Brigantedinium* spp. (22–37%). *Brigantedinium simplex* (8.7-12%) and *Operculodinium centrocarpum* (5.6–19%) were also relatively abundant in these samples.

Zone II also consists of three samples: 2C, 2B, and 7. The dinocyst concentrations ranged between 394 and 807 cysts g^{-1} of dry sed., with a mean concentration of 557 cysts g^{-1} of dry sed. As in Zone I, *Brigantedinium* spp. (47–58%) and *Islandinium minutum* (10–14%) were the most abundant taxa. However, *Islandinium minutum* was significantly less abundant in these three samples. The relative abundance of *Brigantedinium simplex* (9.3–10%) and *Operculodinium centrocarpum* (2.7–13%) were slightly lower in Zone II, but both taxa remained relatively abundant. The relative abundances of *Dubridinium caperatum* were higher in Zone II (3.2–8.9%) than in Zone I (0–1.4%). The autotrophic to heterotrophic ratios were low in both zones, and no trend was apparent.

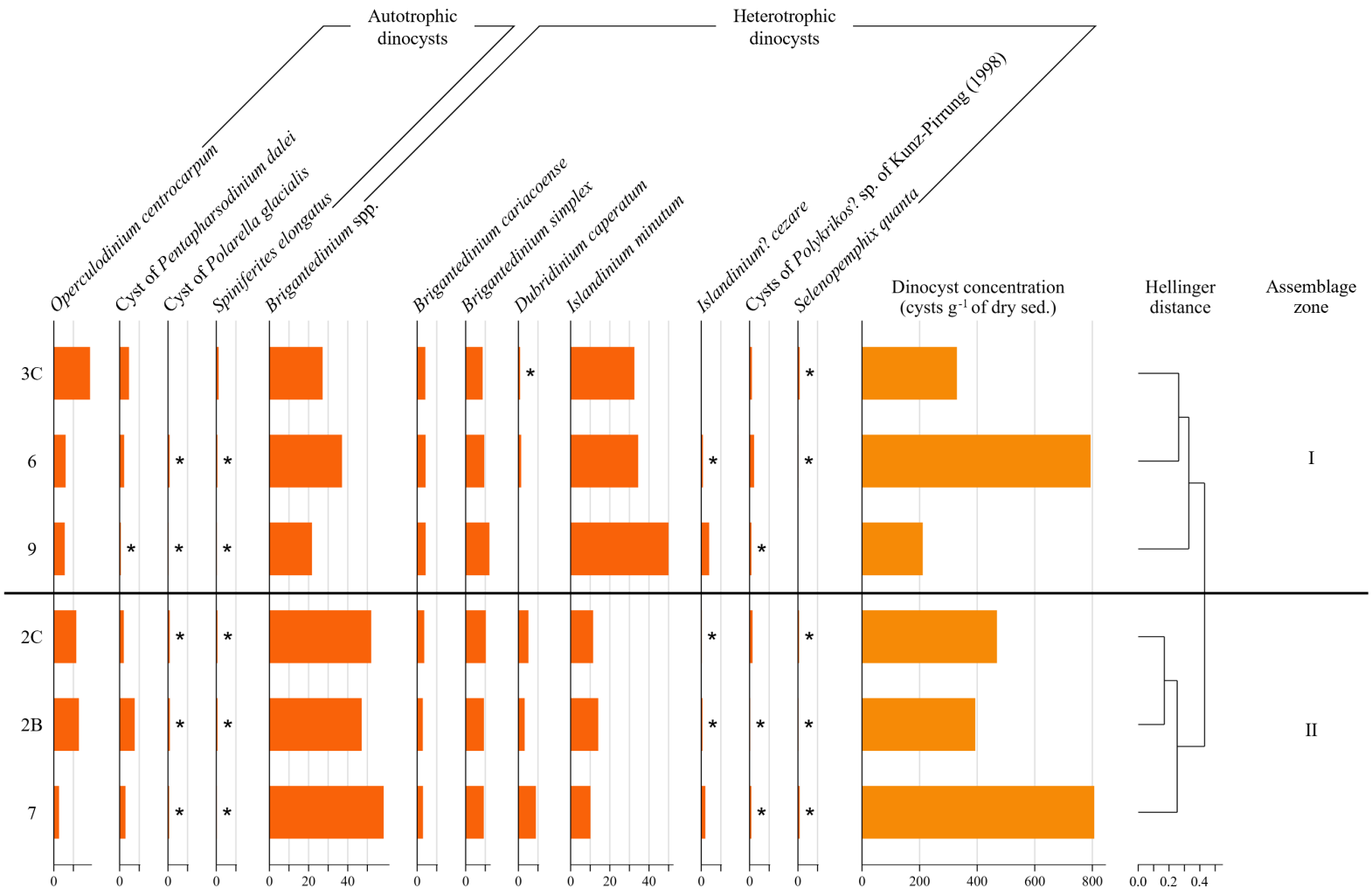


Figure 13. Relative abundance (%) of the observed dinocyst taxa and hierarchical clustering of the surface sediment samples. Taxa marked with an asterisk (*) have a relative abundance of less than 1%

1.3.4 Comparison of the cyst-producing dinoflagellate and dinocyst communities

The cyst-producing dinoflagellate and dinocyst communities were significantly different (PERMANOVA; p-value ≤ 0.001). The SIMPER analysis revealed that six taxa accounted for 79% of the observed differences between the communities (see **Table 4**). Of the 32 dinoflagellate taxa identified, eight produce cysts as part of their life cycle: *Alexandrium catenella* (= cyst of *Alexandrium catenella*), *Gonyaulax spinifera* (= *Spiniferites ramosus*), *Pentapharsodinium dalei* (= cyst of *Pentapharsodinium dalei*), *Peridiniella catenata* (= cyst of *Peridiniella catenata*), *Preperidinium meunieri* (= *Dubridinium caperatum*), *Protoceratium reticulatum* (= *Operculodinium centrocarpum*), *Protoperidinium conicoides* (= *Brigantedinium simplex*), and *Scrippsiella acuminata* (= cyst of *Scrippsiella acuminata*). Cyst of *Alexandrium catenella*, cyst of *Peridiniella catenata*, cyst of *Scrippsiella acuminata*, and *Spiniferites ramosus* were not observed in our sediment samples. Moreover, the motile affinity is known for nine of the 10 dinocyst taxa identified to the species level: *Brigantedinium cariacoense* (= *Protoperidinium avellana*), *Brigantedinium simplex*, *Dubridinium caperatum*, *Islandinium minutum* (= *Islandinium minutum*), *Operculodinium centrocarpum*, cyst of *Pentapharsodinium dalei*, cyst of *Polarella glacialis* (= *Polarella glacialis*; *Polarella glacialis* was excluded from the statistical analyses), *Selenopemphix quanta* (= *Protoperidinium conicum*), and *Spiniferites elongatus* (= *Gonyaulax elongata*). *Gonyaulax elongata*, *Islandinium minutum*, *Protoperidinium avellana*, and *Protoperidinium conicum* were not observed in our phytoplankton samples.

Table 4. Results of the SIMPER analysis for the cyst-producing dinoflagellate and dinocyst communities

	Average dissimilarity (%)	Standard deviation (%)	Individual contribution (%)	Cumulative contribution (%)
Scrippsiella acuminata **(= cyst of *Scrippsiella acuminata*)**	7.382	0.977	13.9	13.9
Gonyaulax elongata **(= *Spiniferites elongatus*)**	7.382	0.977	13.8	27.7
Protoperidinium avellana **(= *Brigantedinium cariacoense*)**	7.382	0.977	13.9	41.6
Islandinium minutum	7.382	0.977	13.9	55.5
Alexandrium catenella **(= cyst of *Alexandrium catenella*)**	6.524	2.191	12.2	67.7
Protoperidinium conicum **(= *Selenopemphix quanta*)**	5.981	2.793	11.3	79.0
Gonyaulax spinifera (= *Spiniferites ramosus*)	3.749	3.482	7.0	86.0
Preperidinium meunieri (= *Dubridinium caperatum*)	3.256	3.961	6.1	92.1
Peridiniella catenata (= cyst of *Peridiniella catenata*)	1.884	3.130	3.6	95.7
Protoperidinium conicoides (= *Brigantedinium simplex*)	1.578	3.399	3.0	98.7
Protoceratium reticulatum (= *Operculodinium centrocarpum*)	0.720	2.301	1.3	100

Taxa in bold significantly contributed to the observed differences between the communities (p-value ≤ 0.05). A lower standard deviation indicates that the taxon consistently contributed to the observed differences across all samples (Clarke, 1993).

1.4 DISCUSSION

1.4.1 Newly recorded dinoflagellate taxa in the Canadian Arctic and insights into the dinoflagellate community

The coastal and shelf areas of the world are divided into 232 ecoregions, each characterized by a relatively homogeneous species composition that differs from neighbouring areas (Spalding et al., 2007). Nineteen of these ecoregions make up the Arctic, with Milne Inlet located in the Baffin Bay–Davis Strait ecoregion. Of the 32 dinoflagellate taxa identified, two were recorded in this ecoregion for the first time: *Gymnodinium* cf. *irregulare* and *Phalacroma pulchellum* (see **Figure 14**, **Annexes VIII** and **XII**). *Gymnodinium irregulare* was described by Hope (1954) from samples collected in Nordåsvannet, Norway, and was later reported in British waters (Parke & Dixon, 1976). This species had never been observed in the Arctic. The identification of *Gymnodinium* cf. *irregulare* was based on Hope's (1954) short original description and drawings, and since no modern micrographs were found, its identification remains uncertain. *Phalacroma pulchellum* was initially described based on samples collected from the southern British Isles (Lebour, 1922) and was later observed in the Kara and Barents seas (Okolodkov, 1998), as well as in multiple regions outside the Arctic (e.g., Gómez & Boicenco, 2004; Parke & Dixon, 1976). Although they have not yet been recorded, both species may be indigenous to the Canadian Arctic. Molecular biological approaches have revealed that microbial plankton communities consist of a few abundant taxa and a large pool of rare taxa, which are often overlooked in traditional microscopic investigations (Cermeño et al., 2014; Pedrós-Alió, 2006; Sogin et al., 2006). Only a few specimens of each species were observed, and both species exhibited a mean concentration of less than 0.2 cells L^{-1}. Arctic ecosystems are largely understudied, and conventional sample volumes substantially underestimate phytoplankton diversity (Arctic Council, 2009; Cermeño et al., 2014). This suggests that some species have yet to be recorded in the Arctic. *Gymnodinium* cf. *irregulare* and *Phalacroma pulchellum* may therefore be rare indigenous species. However, we could hypothesize that both species are NIS and might have been introduced into Milne Inlet via

ballast water discharges. Ballast water dispersion modelling has revealed that ballast water released at Milne Port is undetectable beyond Ragged Island and is usually negligible within 5 km (BIMC, 2024). This would mean that the observed specimens are from established populations rather than from direct ballast water discharges. The possibility that *Gymnodinium* cf. *irregulare* and *Phalacroma pulchellum* are NIS cannot be ruled out at this time. However, both species are more likely to be indigenous.

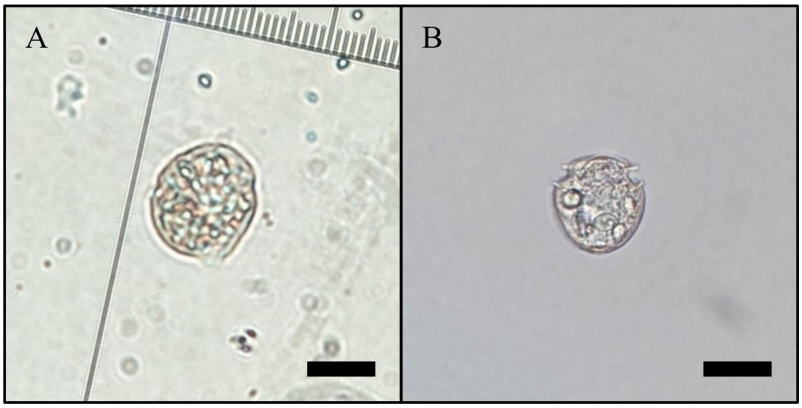


Figure 14. Micrographs of two newly recorded taxa in the Baffin Bay–Davis Strait ecoregion. A. *Gymnodinium* cf. *irregulare*. B. *Phalacroma pulchellum*. Scale bars are 20 µm

Protoperidinium is typically considered the most abundant genus in the Arctic Ocean, which aligns with our findings (Okolodkov & Dodge, 1996). However, a recent study by Schiffrine et al. (2024) suggests that *Tripos* and *Gyrodinium* are the most abundant genera in the Canadian Arctic. This is consistent with the results obtained by Dhifallah et al. (2022) for the samples collected in Milne Inlet in 2017, where *Tripos* spp. accounted for between 35 and 86% of the assemblages. However, *Tripos* and *Gyrodinium* may be overrepresented in Schiffrine et al. (2024). These two genera are mainly observed in the North Atlantic, where most samples were collected by the Continuous Plankton Recorder Survey (Schiffrine et al., 2024). The Continuous Plankton Recorder Survey uses a large mesh size (i.e., 270 µm), which undersamples smaller phytoplankton taxa (Richardson et al., 2006). This results in an underrepresentation of smaller taxa and, consequently, in an overrepresentation of larger taxa, such as *Tripos* and *Gyrodinium* (Schiffrine et al., 2024).

Species of the family Gymnodiniaceae are undoubtedly underrepresented in our samples. This is likely due to the use of formaldehyde as a fixative. Members of this family are athecate (i.e., unarmored or naked) dinoflagellates, and formaldehyde distorts most athecate cells, making identification impossible (Edler & Elbrächter, 2010). In some cases, it may even cause complete disintegration of the cells (European Committee for Standardization, 2011). Lugol's iodine solution is a better choice of fixative for preserving the typical morphology of athecate dinoflagellates (Gómez, 2007). However, it discolours phytoplankton cells, making identification much more complex (Edler & Elbrächter, 2010).

The two dinoflagellate assemblage zones appeared to be primarily determined by the DSIMS. Zone I exhibited the highest abundances of the heterotrophic species *Protoperidinium brevipes*, *Protoperidinium mite*, and *Protoperidinium pellucidum*. These species are pallium feeders, selectively feeding on diatoms over dinoflagellates (e.g., Buskey, 1997; Hansen & Calado, 1999; Jacobson & Anderson, 1986). This indicates that diatoms were abundant in the area at the time of sampling or, at least, sufficiently abundant to sustain the heterotrophic dinoflagellate communities. This was confirmed by the high abundance of diatoms in our samples. Diatoms thrive in turbulent, nutrient-rich waters and typically bloom following sea ice break-up when nutrient concentrations are high (Margalef, 1978). Autotrophic dinoflagellates, on the other hand, thrive in stratified, nutrient-poor waters and often bloom after diatoms, when nutrients are depleted (Margalef, 1978; Rochon, 2009). In contrast, heterotrophic dinoflagellates thrive when and where diatoms are abundant. Based on the DSIMS, the dinoflagellate concentrations, and the observed number of taxa, it is likely that the sampling occurred after a diatom bloom. The dinoflagellate assemblage in Zone Ib differed slightly from that in Zone Ia, showing the highest abundance of *Pentapharsodinium dalei* and the lowest abundance of *Protoperidinium brevipes*. These differences might be linked to the station's location, as station 3C is the southernmost and furthest station from Ragged Island. Zone II exhibited significantly higher abundances of the autotrophic dinoflagellates *Protoceratium reticulatum* and *Tripos arcticus*. This suggests a shift within the dinoflagellate community 25 to 30 days after the sea ice break-up as autotrophic species become more abundant.

1.4.2 Observed differences between the 2017 and 2019 communities

As mentioned by Dhifallah et al. (2022), monitoring the evolution of phytoplankton community composition serves as an excellent barometer of environmental changes in areas like Milne Inlet, where large amounts of ballast water are released annually. Mean concentrations and diversity indices both differed significantly between sampling years, which may be partially explained by the DSIMS. Autotrophic dinoflagellates thrive in stratified, nutrient-poor waters, meaning that both species richness and abundance increase as the ice-free season progresses (Margalef, 1978). The observed differences were, therefore, expected as sampling took place 14–22 days after the sea ice melt in 2017 and 23–32 days after in 2019. Moreover, the mean diversity index observed in 2019 is comparable to those reported for Churchill (western Hudson Bay) in 2007 and 2015, where sampling occurred nearly two months after the sea ice melt (Dhifallah et al., 2022).

Dinoflagellate communities in Milne Inlet differed significantly between 2017 and 2019. The RDA revealed that DSIMS, Z_{Secchi}, and S_{bottom} partially explained the observed differences in community composition. *Tripos lineatus* and *Tripos longipes* were not observed in 2019, while *Tripos arcticus* and *Alexandrium* cf. *catenella* were reported in both sampling years but were significantly more abundant in 2017. This suggests that these four species thrive earlier in the ice-free season under conditions of higher turbidity and lower salinity. This is somewhat unexpected, as these species are autotrophic, and lower turbidity, which increases irradiance, typically supports their growth. These species could, therefore, be adapted to low-light conditions, as suggested by Johns et al. (2003) for *Tripos arcticus*. Moreover, *Tripos longipes* can use light more efficiently than other dinoflagellate species, as it exhibits a higher rate of photosynthesis (Watt, 1971). Species such as *Protoperidinium pellucidum*, *Protoperidinium mite*, *Protoperidinium brevipes*, and *Protoceratium reticulatum*, which are positively related to the explanatory variables, appear to thrive later in the ice-free season under conditions of lower turbidity and higher salinity. However, *Protoperidinium* species are heterotrophic and feed on diatoms. As such, they are more likely linked to the abundance of diatoms, which bloom once or twice a year, than to the DSIMS.

At high latitudes, an initial phytoplankton bloom occurs in spring when light availability is high and nutrients are abundant, while a second bloom sometimes occurs in late summer or early fall when light availability is still high and wind mixing replenishes nutrients in the upper layer of the water column (e.g., Ardyna et al., 2013; 2014; Simo-Matchim et al., 2017). The occurrence of a second bloom in August 2019 may explain the high abundance of diatoms and heterotrophic dinoflagellates in our samples and could partially explain the observed differences between the 2017 and 2019 dinoflagellate communities. Furthermore, sampling in 2017 and 2019 took place in two distinct areas of Milne Inlet, which may explain some of the observed differences in community composition. Ragged Island is located approximately 75 km north-northeast of Milne Port, where the 2017 samples were collected. Both areas are likely influenced by different physicochemical parameters, which shape the composition of the phytoplankton communities.

1.4.3 Insights into the dinocyst community

Of the 12 dinocyst taxa identified, 10 are commonly observed in the surface sediments of the Baffin Bay–Davis Strait ecoregion and are considered common taxa in the Northern Hemisphere (de Vernal et al., 2020; Van Nieuwenhove et al., 2020b): *Brigantedinium* spp., *Brigantedinium cariacoense*, *Brigantedinium simplex*, *Islandinium minutum*, *Islandinium*? *cezare*, *Operculodinium centrocarpum*, cyst of *Pentapharsodinium dalei*, cysts of *Polykrikos*? sp. of Kunz-Pirrung (1998), *Selenopemphix quanta*, and *Spiniferites elongatus*.

Brigantedinium spp. (here including *Brigantedinium cariacoense* and *Brigantedinium simplex*) and *Operculodinium centrocarpum* are cosmopolitan species found in nearshore and offshore settings (de Vernal et al., 2020). *Brigantedinium* cysts are sensitive to oxidative degradation, and the absence of degradation signs suggests that our samples were well-preserved (Zonneveld et al., 2001; Zonneveld et al., 2008). *Islandinium minutum*, cyst of *Pentapharsodinium dalei*, and *Spiniferites elongatus* are abundant in subpolar-polar environments (de Vernal et al., 2020). *Islandinium*? *cezare* and cysts of *Polykrikos*? sp. of

Kunz-Pirrung (1998) are also commonly observed in subpolar-polar environments. *Selenopemphix quanta* is mainly reported in low-latitude to subpolar environments but also sparsely occurs in polar regions.

Dubridinium caperatum and cyst of *Polarella glacialis* are considered rare species in the Northern Hemisphere (Limoges et al., 2020; Mertens et al., 2020; Van Nieuwenhove et al., 2020b). *Dubridinium caperatum* has been reported in recent sediments collected from coastal upwelling regions off northwestern Africa and western North and South America, as well as from the Irish and East China seas, but it had never been observed in the Arctic (Zonneveld et al., 2013). However, *Dubridinium* spp. has been reported in surface sediment samples from Baffin Bay, the northern Labrador Sea, and the Greenland Sea (de Vernal et al., 2020). *Preperidinium meunieri*, the dinoflagellate that produces this dinocyst species, has been observed throughout the Arctic. It is, therefore, not surprising to find its cysts in the underlying sediments. Moreover, *Dubridinium caperatum* can withstand chemical treatments and should thus be more frequently reported in palynological studies (Reid, 1978). Mertens et al. (2020), a recent and well-illustrated taxonomic reference focusing on rare dinocyst taxa, facilitated the identification of *Dubridinium caperatum*. Rochon et al. (1999) served as the preferred taxonomic reference for many years, primarily focusing on common taxa and excluding rare taxa such as *Dubridinium caperatum*. We suspect that *Dubridinium caperatum* is more often than not incorrectly classified as *Brigantedinium* spp. or Protoperidinioids. The inability of observers to identify it accurately and, consequently, its absence from Arctic palynological studies, may partially result from the limited availability of taxonomic references focused on rare taxa and from the species' low occurrence.

Polarella glacialis and its cysts were initially discovered and described from sea-ice brine collected in McMurdo Sound, Antarctica (Montresor et al., 1999; Stoecker et al., 1997; 1998). *Polarella glacialis* was subsequently found in phytoplankton samples collected adjacent to a 63 cm-thick ice edge in northern Baffin Bay (Montresor et al., 2003). This species is not commonly identified in surface sediment samples and is rarely reported in palaeoceanographic studies (Heikkilä et al., 2014; 2016; Limoges et al., 2018). The organic

wall of cyst of *Polarella glacialis* does not contain dinosporin, and its long-term preservation and resistance to palynological sample preparation are unknown (Heikkilä et al., 2014; 2016; Limoges et al., 2018; Montresor et al., 1999). Furthermore, the cysts of *Polarella glacialis* are small (i.e., 12-17 µm long and 8-15 µm wide) and could easily be lost when sieved through the 10-20 µm mesh size commonly used in sample preparation (Heikkilä et al., 2014; Montresor et al., 1999). Palynological sample preparation typically involves chemical treatments with HCl and HF to dissolve carbonates and silicates (de Vernal et al., 2020). However, these chemical treatments can damage the cellular contents and membranes of the cysts, and species such as cyst of *Polarella glacialis* may be altered. We did not use chemical treatments, opting for an alternative protocol that better preserves cellular contents and cysts that do not withstand palynological treatments, which explains the presence of cyst of *Polarella glacialis* in our samples. Some recent studies have identified cyst of *Polarella glacialis* in sediment samples collected in the Arctic. Heikkilä et al. (2014; 2016), Limoges et al. (2018), and Koerner et al. (2021) have reported this species in samples from Hudson Bay, the Mandel Sea and northern Baffin Bay, respectively. Meunier (1910), Ikävalko & Gradinger (1997), Okolodkov (1998), and Kunz-Pirrung (1998) identified spiny cysts similar to cyst of *Polarella glacialis* in samples from the Kara and Barents seas, the Greenland Sea, the Russian Arctic seas, and the Laptev Sea, respectively (Montresor et al., 2003). As it is a sympagic species, cyst of *Polarella glacialis* only occurs in polar environments.

Stations 2B, 2C, and 7 were located only a few hundred metres away within the same bay and exhibited highly similar assemblages. The strong similarity between these three samples can be attributed to their spatial proximity. Samples 6 and 9 were collected west of Ragged Island, 2.5 km apart. Sample 3C was collected south of Ragged Island, more than 10 km from station 9 and nearly 13 km from station 6. Although some differences were observed, all three samples exhibited similar assemblages. Station 6 assemblages were more similar to those of station 3C than to those of station 9 despite station 3C being located more than 10 km farther away. These similarities are not attributed to spatial proximity but rather to sediment texture. Dinocysts behave as silt particles and tend to concentrate in fine sediments (e.g., Dale, 1976; Evitt, 1985). Samples 3C and 6 were composed of fine

sediments, while sample 9 was primarily made up of sand. This explains the similarities and some of the differences observed between these three samples. It also explains the relatively low dinocyst concentration observed in sample 9.

1.4.4 Dinoflagellate and dinocyst taxa contributing to the differences in community composition

The absence of four dinoflagellate species (*Gonyaulax elongata*, *Islandinium minutum*, *Protoperidinium avellana*, and *Protoperidinium conicum*) and four dinocyst species (cyst of *Alexandrium catenella*, cyst of *Peridiniella catenata*, cyst of *Scrippsiella acuminata*, and *Spiniferites ramosus*) explains the statistically significant differences in community composition.

Gonyaulax elongata and *Islandinium minutum* are rarely, if ever, recorded in plankton surveys. Both species were described based on thecae obtained from cyst incubation experiments (Ellegaard et al., 2003; Potvin et al., 2013). *Spiniferites elongatus* is commonly reported in regions characterized by seasonal sea ice and high salinities, which suggests that *Gonyaulax elongata* thrives in highly saline waters (de Vernal et al., 2020). S_{mean} in Milne Inlet in 2019 was 27.9 ± 0.2, which may partially explain the absence of *Gonyaulax elongata* in our samples. Although *Islandinium minutum* has never been recorded in any published plankton surveys, it has been identified by Rochon (unpublished data) in several plankton samples throughout the Canadian Arctic. This indicates that *Islandinium minutum* is neither rare nor a sympagic species. Its absence from plankton surveys suggests that it may have been confused with morphologically similar species over the years.

Protoperidinium avellana is rarely reported in plankton samples from the Arctic. It was described by Meunier (1919) in the southern North Sea. However, most subsequent descriptions have been based on thecae obtained from cysts (Lewis et al., 1984; Matsuoka, 1984; Wall & Dale, 1968). *Protoperidinium avellana* is morphologically similar to

Protoperidinium thorianum, a species commonly observed in the Arctic, and may have been misidentified as the latter in previous studies (Hoppenrath et al., 2009).

Protoperidinium conicum is commonly observed in the Arctic, especially in the Northern Grand Banks–Southern Labrador, Northern Labrador, and Hudson Complex ecoregions (Rochon, unpublished data). However, its absence from our samples suggests that it may be a rare species in the Baffin Bay–Davis Strait ecoregion. It may also be more abundant later in the ice-free season (Dhifallah et al., 2022).

Alexandrium catenella, *Peridiniella catenata*, and *Scrippsiella acuminata* are commonly observed in the Baffin Bay–Davis Strait ecoregion and throughout the Arctic. However, their cysts are rarely, if ever, reported in palynological studies. Cyst of *Alexandrium catenella* rapidly decomposes and disappears in sediments due to the macromolecular composition of its cysts and the thinness of the cyst wall (Ando et al., 2024). Its absence from our samples is, therefore, not surprising as our sediment samples could represent several years to a few decades. Cyst of *Peridiniella catenata*, on the other hand, is not clearly recognizable when empty, which explains the species' absence in palynological studies (Kremp, 2001). Cyst of *Scrippsiella acuminata*, in contrast, is rarely reported, as its calcareous structure does not withstand the chemical treatments involved in palynological preparations (Head et al., 2006). In our case, the sediment samples were processed with slightly acidic distilled water, which may explain the absence of this species in our samples.

The cyst-theca relationship between *Gonyaulax spinifera* and its cysts is poorly understood (e.g., Rochon et al., 2009). At least three dinocyst species are attributed to *Gonyaulax spinifera*: *Ataxiodinium choane*, *Spiniferites mirabilis*, and *Spiniferites ramosus* (Van Nieuwenhove et al., 2020a). *Ataxiodinium choane* and *Spiniferites mirabilis* are mainly observed in low-latitude to subpolar environments and were excluded from the statistical analyses. *Spiniferites ramosus* was not observed in our samples but has been reported in surface sediment samples from Pond Inlet (0–0.9%; de Vernal et al., 2020). A more thorough sampling effort might reveal some specimens.

1.4.5 Ballast water and NIS

Between 2016 and 2018, more than five million cubic metres of ballast water were released into Milne Inlet, none of which was treated prior to release (see **Annex XIII**; BIMC, 2017; 2018; 2019). Nevertheless, all bulk carriers servicing Milne Port completed a mid-ocean ballast water exchange (BWE) as required by the Convention and the Canadian Shipping Act (BIMC, 2019). In 2019, 22% of all bulk carriers that called to Milne Port (28% of all voyages) completed a BWE and treated their ballast water prior to release (see **Table 5**; BIMC, 2020). This means that between 2016 and 2019, only approximately 10% of all ballast water released into Milne Inlet was treated (BIMC, 2017; 2018; 2019; 2020). Several studies have shown that the effectiveness of mid-ocean BWE depends on the ship, oceanic conditions, technique, and taxonomic group (e.g., Dickman & Zhang, 1999; Roy et al., 2012; Simard et al., 2011). Simard et al. (2011) found that BWE eliminates only 29 to 40% of phytoplankton cells, and Roy et al. (2012) stated that BWE is not an effective measure for limiting the introduction of non-indigenous dinoflagellate taxa. Milne Inlet was, therefore, at high risk of introduction between 2016 and 2019. However, there is no evidence to suggest that non-indigenous dinoflagellate species were introduced into Milne Inlet through ballast water discharge. As previously mentioned, *Gymnodinium* cf. *irregulare* and *Phalacroma pulchellum* were recorded for the first time in the Baffin Bay–Davis Strait ecoregion. Although not previously observed, both taxa may be indigenous to the Canadian Arctic. Moreover, no connection can be made between the dinocyst community and the ballast water released into Milne Inlet following the opening of the Mary River Mine. Since 2019, the number of bulk carriers equipped with an IMO-approved BWMS has significantly increased (see **Table 5**; BIMC, 2021; 2022; 2023a; 2024). In 2023, 97% of all bulk carriers that called to Milne Port (99% of all voyages) were equipped with a BWMS and actively used it (BIMC, 2024). The risk of introduction of NIS has, therefore, considerably decreased since 2019.

Table 5. Data on iron ore export and the bulk carriers involved

Year	Million tonnes of iron ore shipped	Number of round-trip voyages	Number of voyages that completed both ballast water exchange and treatment	Number of bulk carriers	Number of bulk carriers equipped with an IMO-approved BWMS
2016	2.8	37	0	18	0
2017	4.1	56	0	23	0
2018	5.1	71	0	31	0
2019	5.9	82	23	41	9
2020	5.5	72	42	36	18
2021	5.6	74	56	39	28
2022	4.7	62	56	35	30
2023	6.1	75	74	39	38

The information presented in this table is sourced from BIMC (2017; 2018; 2019; 2020; 2021; 2022; 2023a; 2024).

1.5 CONCLUSION

Here, we provide new insights into the diversity, structure, and distribution of dinoflagellate communities in one of the busiest ports in the Canadian Arctic. Our data show that two taxa, *Gymnodinium* cf. *irregulare* and *Phalacroma pulchellum*, were recorded for the first time in the Baffin Bay–Davis Strait ecoregion. Both species may be indigenous to the Canadian Arctic. Furthermore, five of the 32 taxa identified are known to produce toxins. Our data also reveal significant differences between the 2017 and 2019 dinoflagellate communities, which were partially explained by DSIMS, Z_{Secchi}, S_{bottom}, and the presence and absence of diatoms. Additionally, significant differences were observed between the cyst-producing dinoflagellate and dinocyst communities. However, these differences are not surprising and can be explained. Moreover, although millions of cubic metres of untreated ballast water were released into Milne Inlet between 2016 and 2019, there is no evidence to suggest that non-indigenous dinoflagellate species were introduced.

Since 8 September 2024, all ships are required to be equipped with an IMO-approved BWMS, significantly reducing the risks associated with ballast water discharge. However, the risks of introduction of NIS cannot be overlooked, and monitoring programs must continue. Moreover, new approaches, such as metagenomics, could and should be used in conjunction with traditional methods to better detect potential NIS and better assess the diversity, structure, and distribution of phytoplankton communities (Dhifallah et al., 2022). Monitoring and research programs are crucial to protect Arctic ecosystems from the introduction and spread of NIS and to understand how they may respond to ongoing environmental changes.

CONCLUSION GÉNÉRALE

Dans l'Arctique, l'augmentation du trafic maritime depuis plus d'une décennie accroît les risques de contamination, notamment par l'introduction d'ENI. Ces risques sont d'autant plus importants dans le bras de mer Milne, où des millions de mètres cubes d'eau de ballast sont rejetés chaque année.

Ce projet de recherche propose une nouvelle évaluation de la diversité, de la structure et de la répartition des communautés de dinoflagellés dans l'un des ports les plus fréquentés de l'Arctique canadien. Nos données montrent que deux taxa, *Gymnodinium* cf. *irregulare* et *Phalacroma pulchellum*, ont été observés pour la première fois dans l'Arctique canadien. Bien qu'ils n'aient pas encore été recensés, ces deux taxa pourraient bien être indigènes à l'Arctique canadien. Les écosystèmes de l'Arctique sont largement sous-étudiés et les volumes d'eau analysés sous-estiment considérablement la diversité du phytoplancton (Arctic Council, 2009; Cermeño et al., 2014). Ceci laisse supposer que certaines espèces n'y ont pas encore été recensées. De plus, cinq des 32 taxa identifiés sont des producteurs potentiels de toxines : *Alexandrium* cf. *catenella*, *Dinophysis acuminata*, *Gonyaulax* cf. *spinifera*, *Phalacroma rotundatum* et *Protoceratium reticulatum*. Nos données révèlent également des différences significatives entre les communautés de dinoflagellés de 2017 et de 2019. Ces différences peuvent en partie être expliquées par le DSIMS, la Z_{Secchi}, la S_{bottom} et la présence et l'absence de diatomées lors de l'échantillonnage. Des différences significatives ont aussi été observées entre la communauté de dinoflagellés producteurs de kystes et la communauté de kystes de dinoflagellés. Toutefois, ces différences ne sont pas surprenantes et peuvent être expliquées. Finalement, bien que des millions de mètres cubes d'eau de ballast non traitée aient été rejetés dans le bras de mer Milne entre 2016 et 2019, rien n'indique que des espèces de dinoflagellés non indigènes y aient été introduites.

À la lumière de ces résultats, il convient d'apporter quelques précisions sur les techniques d'échantillonnage utilisées dans le cadre de ce projet de recherche. Comme mentionné précédemment, les échantillons de phytoplancton ont été prélevés à l'aide d'un filet à plancton de 20 µm. Ce type de filet ne permet pas de retenir le pico- et le nanoplancton, ce qui introduit un biais dans la représentation des communautés phytoplanctoniques. Une approche multi-maille permettrait de corriger ce biais, mais alourdirait considérablement la charge de travail. Il est également important de souligner que notre démarche repose sur un échantillonnage ponctuel. Ce type d'échantillonnage ne nous permet pas de recenser l'ensemble des espèces présentes dans ce bras de mer. Les communautés de phytoplancton évoluent au fil des saisons, et en l'absence d'un suivi hebdomadaire ou mensuel, il demeure impossible d'obtenir un portrait complet de ces communautés.

Depuis le 8 septembre 2024, tous les navires doivent être équipés d'un SGEB approuvé par l'OMI, ce qui réduit considérablement les risques associés au rejet d'eau de ballast. Toutefois, les risques d'introduction d'ENI ne peuvent être négligés et les programmes de surveillance doivent continuer. De plus, de nouvelles approches, telles que la métagénomique, pourraient et devraient être utilisées conjointement aux méthodes traditionnelles afin de mieux détecter de potentielles ENI et de mieux évaluer la diversité, la structure et la répartition des communautés de phytoplancton (Dhifallah et al., 2022). Les programmes de surveillance et de recherche sont essentiels afin de protéger les écosystèmes arctiques de l'introduction et de la propagation des ENI et afin de comprendre comment ces écosystèmes répondront aux changements environnementaux actuels.

ANNEXE I

SAMPLING INFORMATION

Gear	Station	Date	Sampling depth (m)	Latitude	Longitude
Plankton net	6	2019-08-16	38.0	72.5019	-80.0915
	2C	2019-08-17	17.0	72.4812	-80.0110
	3C	2019-08-18	15.0	72.3898	-80.0408
	5	2019-08-19	19.0	72.5346	-80.1175
	8-32	2019-08-19	32.0	72.4582	-79.9935
	1C	2019-08-20	19.0	72.4277	-80.0221
	8-51	2019-08-21	51.0	72.4570	-79.9948
	7	2019-08-22	19.0	72.4714	-79.9777
	10	2019-08-23	20.0	72.5107	-79.9304
	4	2019-08-24	22.0	72.5017	-79.9279
	9	2019-08-25	16.5	72.4797	-80.0767
Van Veen grab	6	2019-08-16	38.0	72.5019	-80.0998
	2B	2019-08-17	10.0	72.4802	-80.0143
	2C	2019-08-17	20.0	72.4805	-80.0095
	3C	2019-08-18	17.6	72.3899	-80.0404
	7	2019-08-22	18.1	72.4754	-79.9812
	9	2019-08-25	17.0	72.4797	-80.0767

ANNEXE II

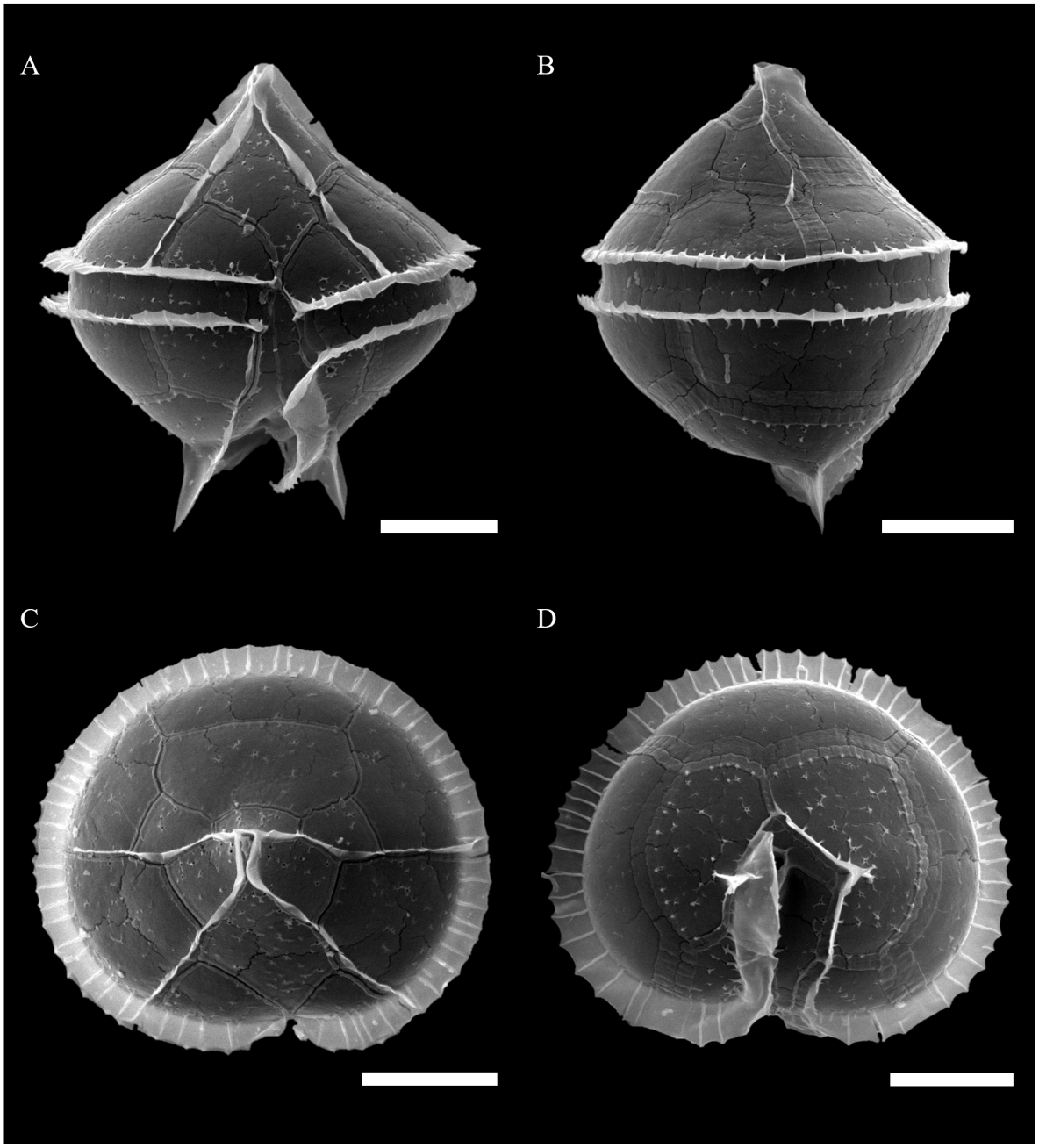
ENVIRONMENTAL DATA

Station	DSIMS	Z (m)	Z_{Secchi} (m)	$T_{surface}$ (°C)	T_{bottom} (°C)	T_{mean} (°C)	$S_{surface}$	S_{bottom}	S_{mean}
6	23	38.0	9.5	3.70	-0.30	1.60	27.15	29.86	28.28
2C	24	17.0	10.5	4.60	1.90	3.13	27.38	29.75	28.72
3C	25	15.0	10.0	7.70	1.50	3.95	22.90	30.88	27.61
5	26	19.0	11.0	6.70	1.20	3.77	23.61	31.32	27.70
8-32	26	32.0	N/A	N/A	N/A	N/A	N/A	N/A	N/A
1C	27	19.0	14.6	6.20	0.50	3.20	25.59	31.08	28.51
8-51	28	51.0	9.8	6.80	2.10	4.90	25.34	30.40	27.32
7	29	19.0	N/A	N/A	N/A	N/A	N/A	N/A	N/A
10	30	20.0	6.5	5.30	2.50	3.95	27.13	30.36	28.61
4	31	22.0	8.7	5.40	3.90	4.85	26.24	28.53	27.05
9	32	16.5	9.0	6.30	1.70	3.53	23.34	30.38	27.45

DSIMS: number of days between sea ice melt and sampling. Z: sampling depth. Z_{Secchi}: Secchi depth. $T_{surface}$: surface temperature. T_{bottom}: bottom temperature. T_{mean}: mean temperature. $S_{surface}$: surface salinity. S_{bottom}: bottom salinity. S_{mean}: mean salinity. N/A: not available.

ANNEXE III

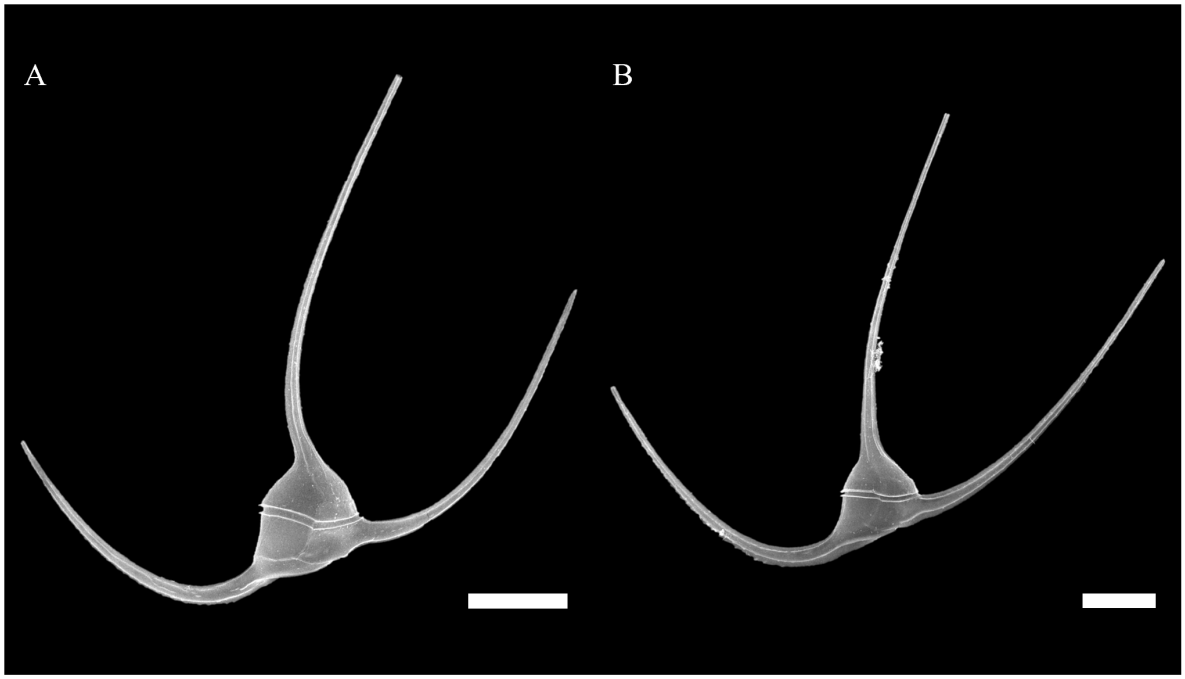
SEM MICROGRAPHS OF *PROTOPERIDINIUM PELLUCIDUM*



A-D. *Protoperidinium pellucidum*. A. Ventral view. B. Right lateral view. C. Epitheca. D. Hypotheca. Scale bars are 10 µm.

ANNEXE IV

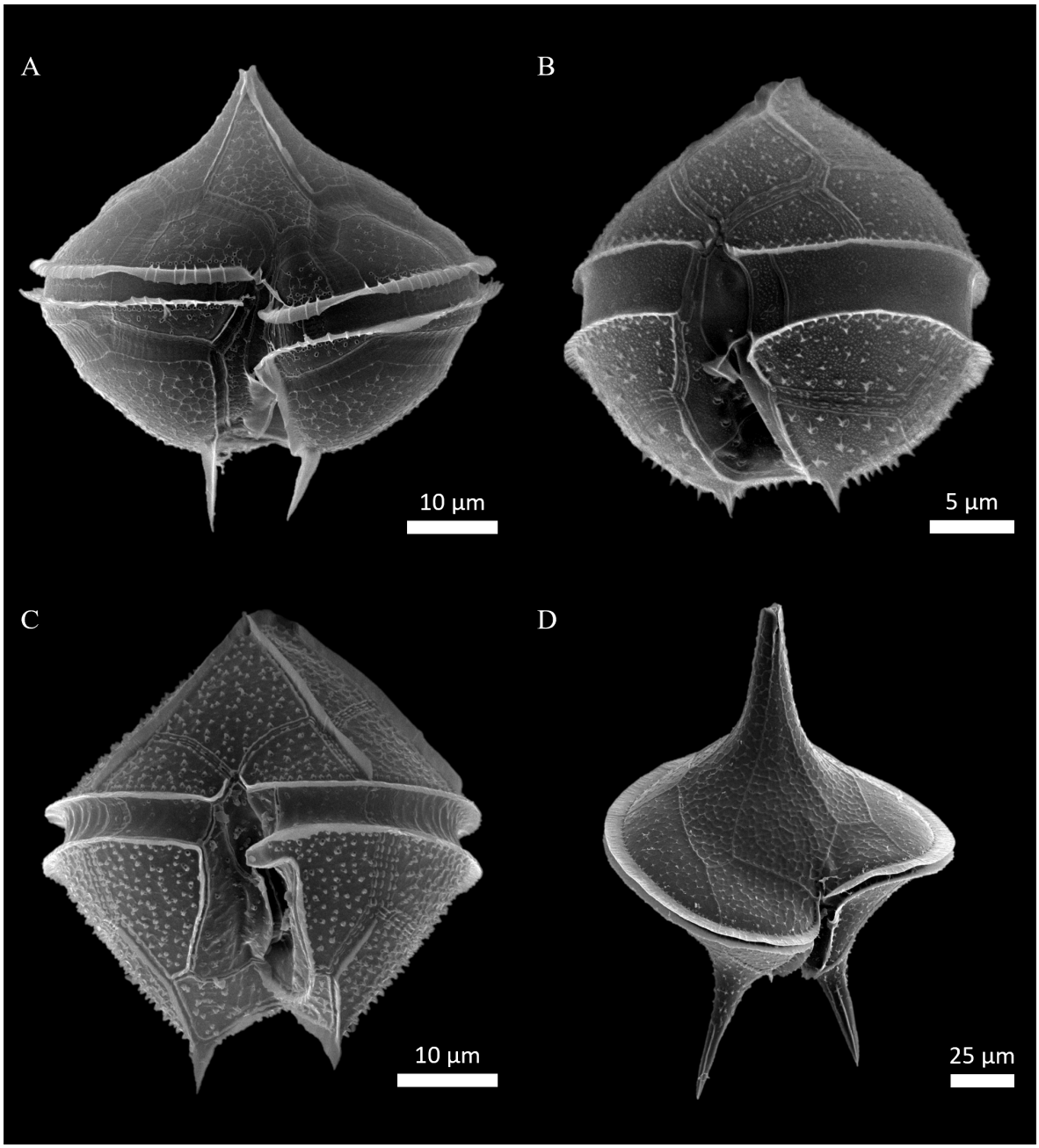
SEM MICROGRAPHS OF *TRIPOS ARCTICUS*



A-B. *Tripos arcticus*, dorsal view. Scale bars are 50 µm.

ANNEXE V

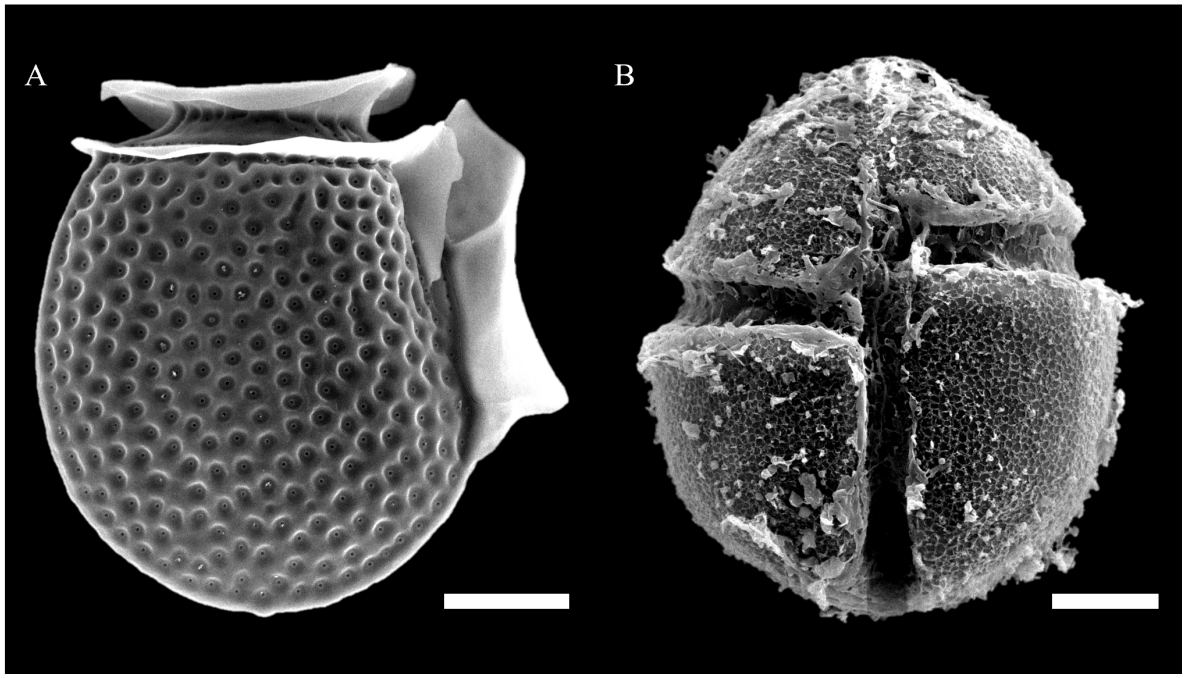
SEM MICROGRAPHS OF FOUR *PROTOPERIDINIUM* TAXA



A. *Protoperidinium arcticum*, ventral view. B. *Protoperidinium brevipes*, ventral view. C. *Protoperidinium conicoides*, ventral view. D. *Protoperidinium depressum*, ventral view.

ANNEXE VI

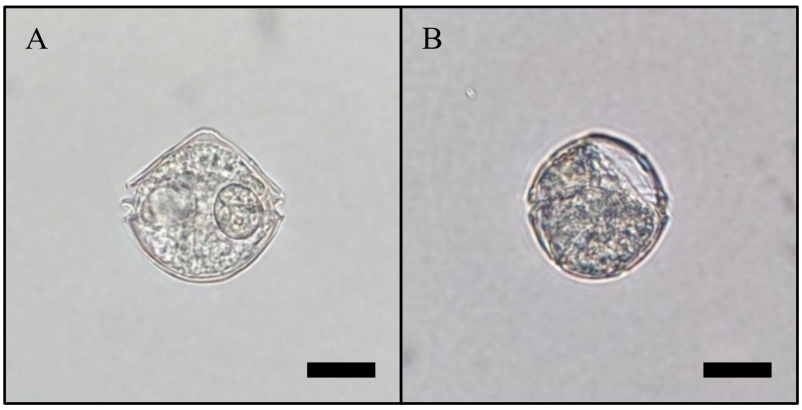
SEM MICROGRAPHS OF TWO PROBLEMATIC TAXA



A. *Dinophysis* cf. *arctica*, right lateral view. B. *Gymnodinium* sp. 1, ventral view. Scale bars are 10 µm.

ANNEXE VII

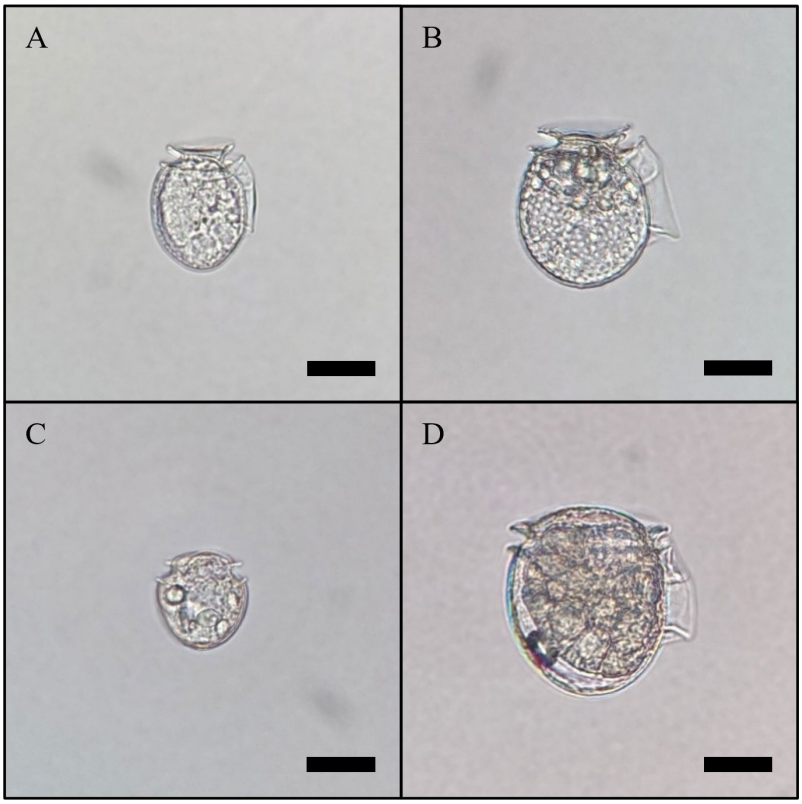
MICROGRAPHS OF TWO *ALEXANDRIUM* TAXA



A. *Alexandrium* cf. *catenella*. B. *Alexandrium* sp. 1. Scale bars are 20 µm.

ANNEXE VIII

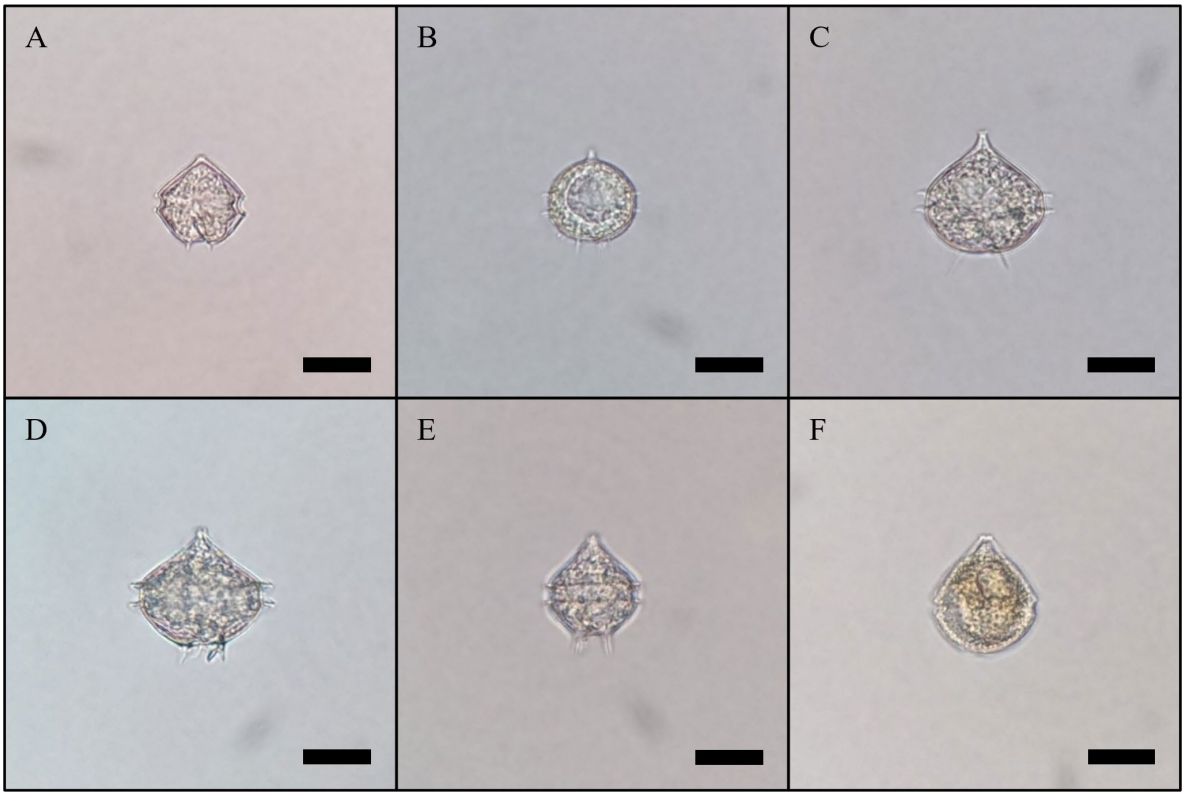
MICROGRAPHS OF TWO *DINOPHYSIS* TAXA AND TWO *PHALACROMA* TAXA



A. *Dinophysis acuminata*. B. *Dinophysis* cf. *arctica*. C. *Phalacroma pulchellum*. D. *Phalacroma rotundatum*. Scale bars are 20 µm.

ANNEXE IX

MICROGRAPHS OF THE THREE MOST ABUNDANT DINOFLAGELLATE TAXA IDENTIFIED IN MILNE INLET AND TWO PROBLEMATIC TAXA



A. *Protoperidinium brevipes*. B. *Protoperidinium* cf. *cerasus*. C. *Protoperidinium mite*. D. *Protoperidinium pellucidum*. E. *Protoperidinium pellucidum*, small specimen. F. *Scrippsiella* cf. *acuminata*. Scale bars are 20 µm.

ANNEXE X
PROBLEMATIC TAXA AND INSIGHTS ON SPECIFIC SPECIES

Protoperidinium mite

Protoperidinium mite is often confused with *Protoperidinium cerasus* and *Protoperidinium quarnerense* but can be reliably identified through detailed observations of its anterior intercalary plate 2a. We confidently identified this species as *Protoperidinium mite* based on SEM analysis of 10 specimens.

Protoperidinium cf. *cerasus*

Under an optical microscope, *Protoperidinium* cf. *cerasus*, although smaller, resembles *Protoperidinium mite*. However, its thecal plates are smooth and lack the characteristic ornamentation observed on those of *Protoperidinium mite*. Its first precingular plate is relatively small, and its apical horn is distinct from the epitheca. Both characteristics align with the original description of *Protoperidinium cerasus* by Paulsen (1907). *Protoperidinium* cf. *cerasus* also resembles the original drawings. However, the specimens observed in our samples are smaller and do not match the descriptions of *Protoperidinium cerasus* in recent identification keys (e.g., Hoppenrath et al., 2009). We, therefore, referred to this taxon as *Protoperidinium* cf. *cerasus*.

Tripos sp. 1

Dinoflagellates have a complex life cycle, comprising a haploid phase and a diploid phase (Evitt, 1985). During the haploid phase, cells divide mitotically. A single cell divides into two daughter cells, then two into four, four into eight, and so on. Under specific conditions, some vegetative cells function as gametes. These gametes fuse in pairs to form a motile diploid zygote, known as a planozygote. Some species, such as *Ceratium cornutum* and *Tripos longipes*, are anisogamous, meaning that male and female gametes differ

morphologically (Pfiester & Anderson, 1987). Male gametes are usually smaller and differ morphologically from vegetative cells (von Stosch, 1964). *Tripos* sp. 1 is likely a male gamete of the species *Tripos arcticus*. *Tripos* sp. 1 measures 125–135 μm in length and 60–70 μm in width.

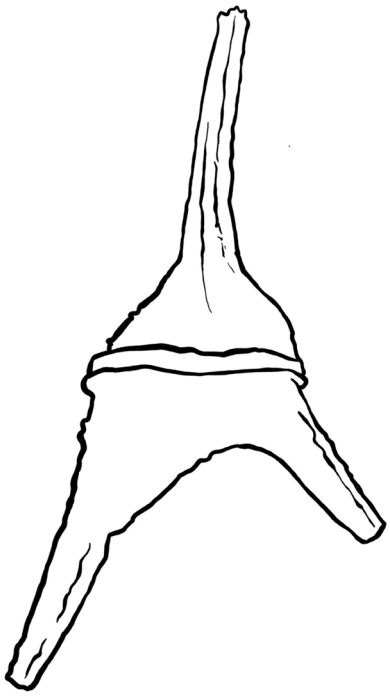


Figure 15. *Tripos* sp. 1, dorsal view

Dinophysis* cf. *arctica

The identification of *Dinophysis* species is typically based on the size and shape of the cells, the thecal ornamentation, and the morphology of the left sulcal list (Larsen & Moestrup, 1992). *Dinophysis* cf. *arctica* is morphologically similar to several *Dinophysis* species. The specimens observed in our samples are roughly the same size as *Dinophysis arctica* and *Dinophysis recurva* and share morphological characteristics with both species. Given that *Dinophysis arctica* is commonly observed in the Canadian Arctic, we referred to this taxon as *Dinophysis* cf. *arctica*.

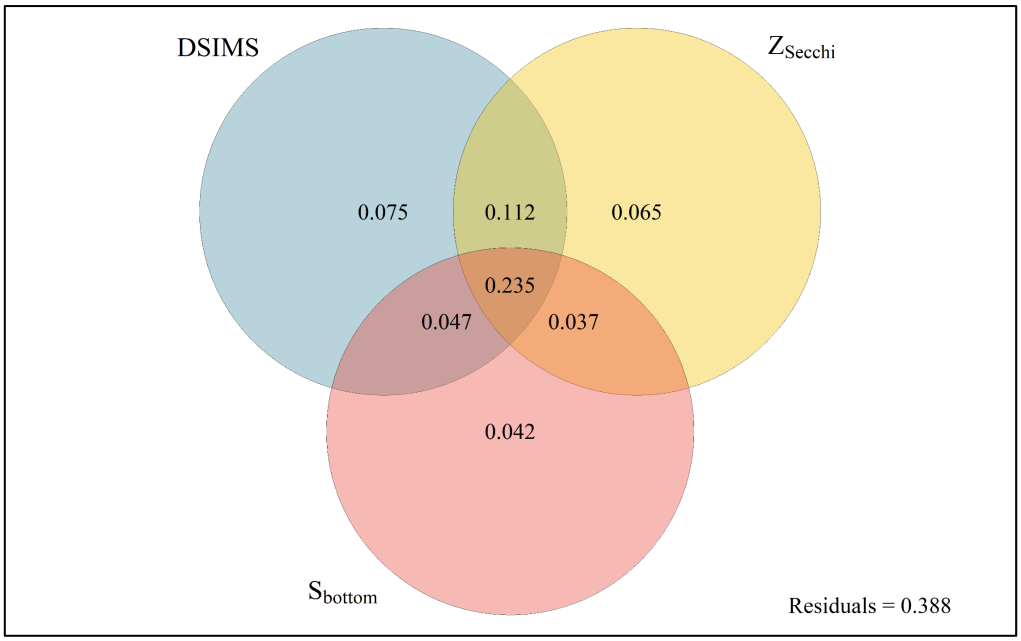
Phalacroma rotundatum

Worldwide, 112 species have been assigned to the genus *Dinophysis* and 60 to the genus *Phalacroma* (Guiry, 2020; 2024). However, only 10 species of *Dinophysis* and two species of *Phalacroma* have been shown to produce okadaic acid and pectenotoxins (Bates et al., 2020; Reguera et al., 2012; 2014). Of these 12 species, only seven have been linked to diarrhetic shellfish poisoning outbreaks (Reguera et al., 2012). While *Dinophysis acuminata*, *Dinophysis acuta*, *Dinophysis caudata*, *Dinophysis fortii*, *Dinophysis norvegica*, and *Phalacroma rotundatum* occur in Canadian waters, none have been shown to produce okadaic acid (Bates et al., 2020). *Prorocentrum lima* is the only confirmed producer of okadaic acid in Canadian waters.

Doubts remain regarding the toxigenic nature of *Phalacroma rotundatum* (Reguera et al., 2012; 2014). Although *Phalacroma rotundatum* has been shown to contain okadaic acid and pectenotoxins, it does not appear to produce them. Evidence suggests that this heterotrophic dinoflagellate does not synthesize toxins *de novo*, but instead acquires them from its prey (González-Gil et al., 2011). *Phalacroma rotundatum* preys on ciliates, which in turn feed on other planktonic cells, including toxic *Dinophysis* species (Hansen, 1991). Consequently, *Phalacroma rotundatum* may serve as a vector of toxins rather than as a toxin producer (González-Gil et al., 2011).

ANNEXE XI

RESULTS OF THE VARIATION PARTITIONING ANALYSIS



ANNEXE XII
LIST OF THE DINOFLAGELLATE TAXA IDENTIFIED AND THEIR CURRENT DISTRIBUTION IN THE ARCTIC

Actiniscus pentasterias (Ehrenberg) Ehrenberg, 1844

Okolodkov (1998) and the references therein; Barents Sea, Chukchi Sea

Lovejoy et al. (2002); northern Baffin Bay

Jensen and Veland (2006); Disko Bay

Dhifallah et al. (2022); mouth of the Churchill River, Deception Bay, Milne Inlet

Alexandrium catenella (Whedon & Kofoid) Balech, 1985

Bursa (1963); Point Barrow [as *Gonyaulax tamarensis* Lebour, 1925]

Hsiao (1983) and the references therein; Igloolik [as *Gonyaulax tamarensis* Lebour, 1925]

Hsiao et al. (1984); mouth of the Great Whale River [as *Gonyaulax tamarensis* Lebour, 1925]

Hsiao and Pinkewycz (1985b); Frobisher Bay [as *Gonyaulax tamarensis* Lebour, 1925]

Percy et al. (1992); Sugluk Inlet, Wakeham Bay [as *Gonyaulax tamarensis* Lebour, 1925]

Okolodkov (1998) and the references therein; Barents Sea [as *Alexandrium tamarense* (Lebour) Balech, 1995]

Okolodkov (2005) and the references therein; Barents Sea [as *Alexandrium tamarense* (Lebour) Balech, 1995]

Jensen and Veland (2006); Disko Bay [as *Alexandrium tamarense* (Lebour) Balech, 1995]

Niemi et al. (2011); Thesiger Bay [as *Alexandrium tamarense* (Lebour) Balech, 1995]

Anderson et al. (2021); eastern Bering Sea, Bering Strait, Chukchi Sea, southwestern Beaufort Sea [as cyst of *Alexandrium catenella* (Whedon & Kofoid) Balech, 1985]

Dhifallah et al. (2022); mouth of the Churchill River, Deception Bay, Milne Inlet

Amylax triacantha (Jørgensen) Sournia, 1984

Bursa (1963); Point Barrow [as *Gonyaulax triacantha* Jørgensen, 1899]

Hsiao (1983) and the references therein; western Baffin Bay, off Bylot Island, off Exeter Sound [as *Gonyaulax triacantha* Jørgensen, 1899]

Pinkewycz and Hsiao (1987); southern Beaufort Sea [as *Gonyaulax triacantha* Jørgensen, 1899]

Falk-Petersen et al. (1997); northern Barents Sea

Okolodkov (1998) and the references therein; Barents Sea, Chukchi Sea, Kara Sea, Laptev Sea, White Sea

Melnikov et al. (2002); Canada Basin of the Arctic Ocean

Jensen and Veland (2006); Disko Bay

Kubiszyn et al. (2014); off Spitsbergen

Krawczyk et al. (2015); Godthåbsfjord system

Laget (2017); Deception Bay

Dhifallah et al. (2022); mouth of the Churchill River, Deception Bay, Frobisher Bay, Milne Inlet

Dinophysis acuminata Claparède & Lachmann, 1859

Bursa (1963); Point Barrow

Hsiao (1976); southern Beaufort Sea, Eskimo Lakes

Anderson et al. (1981); Hudson Bay

Hsiao (1983) and the references therein; off Merchants Bay, off Exeter Sound, Beaufort Sea, Eskimo Lakes

Hsiao et al. (1984); mouth of the Great Whale River

Hsiao and Pinkewycz (1985a); Frobisher Bay

Hsiao and Pinkewycz (1985b); Frobisher Bay

Percy et al. (1992); Sugluk Inlet

Falk-Petersen et al. (1997); northern Barents Sea

Okolodkov (1998) and the references therein; Barents Sea, Chukchi Sea, East Siberian Sea, Kara Sea, White Sea

Lovejoy et al. (2002); northern Baffin Bay, off Murchison Sound

Melnikov et al. (2002); Canada Basin of the Arctic Ocean

Okolodkov (2005) and the references therein; Barents Sea, Chukchi Sea, East Siberian Sea, Kara Sea, White Sea

Jensen and Veland (2006); Disko Bay

Różańska et al. (2009); Franklin Bay

Seuthe et al. (2011); Kongsfjorden

Kubiszyn et al. (2014); off Spitsbergen

Krawczyk et al. (2015); Godthåbsfjord system

Laget (2017); Deception Bay

Simo-Matchim et al. (2017); Nachvak Fjord, Okak Fjord, Anaktalak Fjord

Dhifallah et al. (2022); mouth of the Churchill River, Deception Bay, Frobisher Bay, Milne Inlet, Wager Bay

Dinophysis arctica Mereschkowsky, 1879

Anderson et al. (1981); Hudson Bay

Hsiao (1983) and the references therein; Davis Strait, western Baffin Bay, Exeter Sound, Balaena Bay, Lancaster Sound

Hsiao et al. (1984); mouth of the Great Whale River

Hsiao and Pinkewycz (1985a); Frobisher Bay

Hsiao and Pinkewycz (1985b); Frobisher Bay

Hsiao and Pinkewycz (1987); western Baffin Bay, Jones Sound, Lancaster Sound

Pinkewycz et al. (1987); northeastern Ungava Bay

Okolodkov (1998) and the references therein; Barents Sea, Chukchi Sea, Kara Sea, Laptev Sea, White Sea

Melnikov et al. (2002); Canada Basin of the Arctic Ocean

Gonyaulax spinifera (Claparède & Lachmann) Diesing, 1866

Bursa (1963); Point Barrow

Falk-Petersen et al. (1997); northern Barents Sea

Okolodkov (1998) and the references therein; Barents Sea, Chukchi Sea, Kara Sea, Laptev Sea, White Sea

Lovejoy et al. (2002); northern Baffin Bay, off Murchison Sound, Smith Sound

Laget (2017); Deception Bay

de Vernal et al. (2020); Greenland Sea, Iceland Sea, Denmark Strait, eastern Baffin Bay, Disko Bay, Labrador Sea, Okak Fjord, Anaktalak Fjord, Davis Strait, Baffin Bay, Pond Inlet, Clyde Inlet, Hudson Bay, Hudson Strait, Nares Strait, Amundsen Gulf, Thesiger Bay, Beaufort Sea, Barents Sea, Chukchi Sea, East Siberian Sea, Kara Sea, eastern Bering Sea, Norwegian Sea [as *Spiniferites ramosus* (Ehrenberg) Mantell, 1854]

Dhifallah et al. (2022); mouth of the Churchill River, Deception Bay

Gymnodinium irregulare Hope, 1954

Lessardia elongata Saldarriaga & Taylor, 2003

Lovejoy et al. (2002); northern Baffin Bay, off Murchison Sound, Smith Sound [as *Gymnodinium elongatum* Hope, 1954]

Niemi et al. (2011); Thesiger Bay [as *Gymnodinium elongatum* Hope, 1954]

Seuthe et al. (2011); Kongsfjorden

Kubiszyn et al. (2014); off Spitsbergen

Simo-Matchim et al. (2017); Nachvak Fjord, Saglek Fjord, Anaktalak Fjord [as *Gymnodinium elongatum* Hope, 1954]

Micracanthodinium claytonii (Holmes) Dodge, 1982

Booth and Horner (1997); Arctic Ocean

Falk-Petersen et al. (1997); northern Barents Sea

Okolodkov (1998) and the references therein; Barents Sea

Lovejoy et al. (2002); northern Baffin Bay, Smith Sound

Kubiszyn et al. (2014); off Spitsbergen

Simo-Matchim et al. (2017); Nachvak Fjord, Saglek Fjord, Okak Fjord, Anaktalak Fjord

Dhifallah et al. (2022); Wager Bay

Pentapharsodinium dalei Indelicato & Loeblich, 1986

Jensen and Veland (2006); Disko Bay

de Vernal et al. (2020); Greenland Sea, Iceland Sea, Denmark Strait, eastern Baffin Bay, Disko Bay, Labrador Sea, Nachvak Fjord, Saglek Fjord, Okak Fjord, Anaktalak Fjord, Davis Strait, Baffin Bay, Pond Inlet, Clyde Inlet, Hudson Bay, Hudson Strait, Foxe Channel, Lancaster Sound, Admiralty Inlet, M'Clintock Channel, Queen Maud Gulf, Dease Strait, Prince of Wales Strait, Amundsen Gulf, Thesiger Bay, Beaufort Sea, Barents Sea, Chukchi Sea, East Siberian Sea, Kara Sea, Laptev Sea, eastern Bering Sea, Norwegian Sea [as cyst of *Pentapharsodinium dalei* Indelicato & Loeblich, 1986]

Dhifallah et al. (2022); mouth of the Churchill River, Deception Bay

Peridiniella catenata (Levander) Balech, 1977

Bursa (1963); Point Barrow [as *Gonyaulax catenata* (Levander) Kofoid, 1911]

Hsiao (1976); southern Beaufort Sea, Eskimo Lakes [as *Gonyaulax catenata* (Levander) Kofoid, 1911]

Hsiao (1983) and the references therein; Jones Sound, Lancaster Sound, western Baffin Bay, Clarence Head, Cape Isabella, Cape Faraday, off Bylot Island, Igloolik, Balaena Bay, Beaufort Sea, Eskimo Lakes, Wellington Channel, Resolute Passage [as *Gonyaulax catenata* (Levander) Kofoid, 1911]

Hsiao et al. (1984); mouth of the Great Whale River [as *Gonyaulax catenata* (Levander) Kofoid, 1911]

Hsiao and Pinkewycz (1985a); Frobisher Bay [as *Gonyaulax catenata* (Levander) Kofoid, 1911]

Hsiao and Pinkewycz (1985b); Frobisher Bay [as *Gonyaulax catenata* (Levander) Kofoid, 1911]

Hsiao and Pinkewycz (1987); western Baffin Bay, Jones Sound [as *Gonyaulax catenata* (Levander) Kofoid, 1911]

Pinkewycz and Hsiao (1987); southern Beaufort Sea [as *Gonyaulax catenata* (Levander) Kofoid, 1911]

Percy et al. (1992); Sugluk Inlet, Wakeham Bay [as *Gonyaulax catenata* (Levander) Kofoid, 1911]

Falk-Petersen et al. (1997); northern Barents Sea

Okolodkov (1998) and the references therein; Arctic Basin, Barents Sea, Chukchi Sea, East Siberian Sea, Kara Sea, Laptev Sea, White Sea

Jensen and Veland (2006); Disko Bay

Różańska et al. (2009); Franklin Bay

Stewart (2013); southern Beaufort Sea

Krawczyk et al. (2015); Godthåbsfjord system

Laget (2017); Deception Bay

Dhifallah et al. (2022); Deception Bay, Frobisher Bay, Milne Inlet

Peridiniella danica (Paulsen) Okolodkov & Dodge, 1995

Hsiao (1983) and the references therein; Jones Sound, Lancaster Sound, western Baffin Bay, Clarence Head, Cape Isabella, Cape Faraday, off Bylot Island, off Merchants Bay [as *Glenodinium danicum* Paulsen, 1907]

Falk-Petersen et al. (1997); northern Barents Sea

Okolodkov (1998) and the references therein; Barents Sea, Kara Sea, White Sea

Riedel et al. (2003); McDougall Sound

Niemi et al. (2011); Thesiger Bay

Laget (2017); Deception Bay

Simo-Matchim et al. (2017); Nachvak Fjord, Saglek Fjord, Okak Fjord, Anaktalak Fjord

Dhifallah et al. (2022); Wager Bay

Phalacroma pulchellum Lebour, 1922

Falk-Petersen et al. (1997); northern Barents Sea [as *Dinophysis pulchella* (Lebour) Balech, 1967]

Okolodkov (1998) and the references therein; Barents Sea, Kara Sea [as *Dinophysis pulchella* (Lebour) Balech, 1967]

Phalacroma rotundatum (Claparède & Lachmann) Kofoid & Michener, 1911

Bursa (1963); Point Barrow

Anderson et al. (1981); Hudson Bay [as *Dinophysis rotundata* Claparède & Lachmann, 1859]

Hsiao (1983) and the references therein; Cape Isabella, off Clarence Head, off Bylot Island, off Exeter Sound [as *Prodinophysis rotundata* (Claparède & Lachmann) Balech, 1944]

Percy et al. (1992); Sugluk Inlet [as *Dinophysis rotundata* Claparède & Lachmann, 1859]

Falk-Petersen et al. (1997); northern Barents Sea [as *Dinophysis rotundata* Claparède & Lachmann, 1859]

Okolodkov (1998) and the references therein; Barents Sea, Chukchi Sea, Kara Sea, Laptev Sea, White Sea [as *Dinophysis rotundata* Claparède & Lachmann, 1859]

Melnikov et al. (2002); Canada Basin of the Arctic Ocean [as *Dinophysis rotundata* Claparède & Lachmann, 1859]

Okolodkov (2005) and the references therein; Barents Sea, Chukchi Sea, Kara Sea, Laptev Sea, White Sea [as *Dinophysis rotundata* Claparède & Lachmann, 1859]

Jensen and Veland (2006); Disko Bay [as *Dinophysis rotundata* Claparède & Lachmann, 1859]

Kubiszyn et al. (2014); off Spitsbergen [as *Dinophysis rotundata* Claparède & Lachmann, 1859]

Krawczyk et al. (2015); Godthåbsfjord system [as *Dinophysis rotundata* Claparède & Lachmann, 1859]

Laget (2017); Deception Bay [as *Dinophysis rotundata* Claparède & Lachmann, 1859]

Simo-Matchim et al. (2017); Okak Fjord [as *Dinophysis rotundata* Claparède & Lachmann, 1859]

Dhifallah et al. (2022); mouth of the Churchill River, Deception Bay, Milne Inlet

Preperidinium meunieri (Pavillard) Elbrächter, 1993

Okolodkov (1998) and the references therein; Barents Sea, Chukchi Sea, Kara Sea, White Sea

Jensen and Veland (2006); Disko Bay

Krawczyk et al. (2015); Godthåbsfjord system

Laget (2017); Deception Bay

Dhifallah et al. (2022); mouth of the Churchill River, Deception Bay, Milne Inlet, Wager Bay

Pronoctiluca pelagica Fabre-Domergue, 1889

Hsiao (1983) and the references therein; Davis Strait

Lovejoy et al. (2002); northern Baffin Bay

Riedel et al. (2003); McDougall Sound

Niemi et al. (2011); Thesiger Bay

Seuthe et al. (2011); Kongsfjorden

Kubiszyn et al. (2014); off Spitsbergen

Dhifallah et al. (2022); Wager Bay

Protoceratium reticulatum (Claparède & Lachmann) Bütschli, 1885

Bursa (1963); Point Barrow

Hsiao (1983) and the references therein; Lancaster Sound, Igloolik

Percy et al. (1992); Sugluk Inlet

Okolodkov (1998) and the references therein; Barents Sea, Chukchi Sea, Kara Sea, White Sea

Melnikov et al. (2002); Canada Basin of the Arctic Ocean [as *Gonyaulax grindleyi* Reinecke, 1967]

Okolodkov (2005) and the references therein; Barents Sea, Chukchi Sea, Kara Sea, White Sea

Jensen and Veland (2006); Disko Bay

Krawczyk et al. (2015); Godthåbsfjord system

Laget (2017); Deception Bay

de Vernal et al. (2020); Greenland Sea, Iceland Sea, Denmark Strait, eastern Baffin Bay, Disko Bay, Labrador Sea, Nachvak Fjord, Saglek Fjord, Davis Strait, Baffin Bay, Pond Inlet, Clyde Inlet, Hudson Bay, Hudson Strait, Foxe Channel, Lancaster Sound, Barrow Strait, Admiralty Inlet, Prince Gustaf Adolf Sea, Norwegian Bay, Nares Strait, M'Clintock

Channel, Queen Maud Gulf, Dease Strait, Prince of Wales Strait, Amundsen Gulf, Thesiger Bay, Beaufort Sea, Barents Sea, Chukchi Sea, East Siberian Sea, Kara Sea, Laptev Sea, eastern Bering Sea, Norwegian Sea [as *Operculodinium centrocarpum* sensu Wall & Dale, 1966]

Dhifallah et al. (2022); mouth of the Churchill River, Deception Bay, Frobisher Bay, Milne Inlet, Wager Bay

Protoperidinium arcticum (Grøntved & Seidenfaden) Okolodkov, 1997

Hsiao (1983) and the references therein; Jones Sound, Lancaster Sound, western Baffin Bay, Clarence Head, Cape Isabella, Cape Faraday, off Bylot Island [as *Peridinium ovatum* f. *arctica* Grøntved & Seidenfaden, 1938]

Okolodkov (1998) and the references therein; Barents Sea, Chukchi Sea

Laget (2017); Deception Bay

Protoperidinium bipes (Paulsen) Balech, 1974

Anderson et al. (1981); Hudson Bay

Percy et al. (1992); Sugluk Inlet

Falk-Petersen et al. (1997); northern Barents Sea

Okolodkov (1998) and the references therein; Arctic Basin, Barents Sea, Chukchi Sea, Kara Sea, Laptev Sea, White Sea

Lovejoy et al. (2002); northern Baffin Bay

Melnikov et al. (2002); Canada Basin of the Arctic Ocean

Riedel et al. (2003); McDougall Sound

Jensen and Veland (2006); Disko Bay

Seuthe et al. (2011); Kongsfjorden

Stewart (2013); southern Beaufort Sea

Kubiszyn et al. (2014); off Spitsbergen

Krawczyk et al. (2015); Godthåbsfjord system

Laget (2017); Deception Bay

Simo-Matchim et al. (2017); Nachvak Fjord, Okak Fjord, Anaktalak Fjord

Dhifallah et al. (2022); Deception Bay, Frobisher Bay, Milne Inlet, Wager Bay

Protoperidinium brevipes (Paulsen) Balech, 1974

Bursa (1963); Point Barrow [as *Peridinium brevipes* Paulsen, 1908]

Anderson et al. (1981); Hudson Bay [as *Peridinium brevipes* Paulsen, 1908]

Hsiao (1983) and the references therein; Jones Sound, Lancaster Sound, western Baffin Bay, Clarence Head, Cape Isabella, Cape Faraday, off Bylot Island, off Merchants Bay, off Exeter Sound, Igloolik, Davis Strait [as *Peridinium brevipes* Paulsen, 1908]

Booth and Horner (1997); Arctic Ocean

Falk-Petersen et al. (1997); northern Barents Sea

Okolodkov (1998) and the references therein; Barent Sea, Chukchi Sea, East Siberian Sea, Kara Sea, Laptev Sea, White Sea

Lovejoy et al. (2002); northern Baffin Bay

Melnikov et al. (2002); Canada Basin of the Arctic Ocean

Riedel et al. (2003); McDougall Sound

Jensen and Veland (2006); Disko Bay

Seuthe et al. (2011); Kongsfjorden

Kubiszyn et al. (2014); off Spitsbergen

Krawczyk et al. (2015); Godthåbsfjord system

Laget (2017); Deception Bay

Simo-Matchim et al. (2017); Nachvak Fjord

Dhifallah et al. (2022); mouth of the Churchill River, Deception Bay, Frobisher Bay, Milne Inlet, Wager Bay

Protoperidinium cerasus (Paulsen) Balech, 1973

Bursa (1963); Point Barrow [as *Peridinium cerasus* Paulsen, 1907]

Hsiao (1976); southern Beaufort Sea, Eskimo Lakes [as *Peridinium cerasus* Paulsen, 1907]

Hsiao and Pinkewycz (1987); western Baffin Bay, Lancaster Sound [as *Peridinium cerasus* Paulsen, 1907]

Falk-Petersen et al. (1997); northern Barents Sea

Okolodkov (1998) and the references therein; Barents Sea, Chukchi Sea, Kara Sea, Laptev Sea, White Sea

Jensen and Veland (2006); Disko Bay

Kubiszyn et al. (2014); off Spitsbergen

Krawczyk et al. (2015); Godthåbsfjord system

Laget (2017); Deception Bay

Dhifallah et al. (2022); Wager Bay

Protoperidinium conicoides (Paulsen) Balech, 1973

Hsiao (1983) and the references therein; Davis Strait, Jones Sound, western Baffin Bay, Clarence Head, off Bylot Island, off Merchants Bay, off Exeter Sound, Lancaster Sound [as *Peridinium conicoides* Paulsen, 1905]

Okolodkov (1998) and the references therein; Barents Sea, Chukchi Sea, East Siberian Sea, Kara Sea, Laptev Sea, White Sea

Laget (2017); Deception Bay

Dhifallah et al. (2022); mouth of the Churchill River, Deception Bay, Frobisher Bay

Protoperidinium depressum (Bailey) Balech, 1974

Bursa (1963); Point Barrow [as *Peridinium depressum* Bailey, 1854]

Anderson et al. (1981); Hudson Bay [as *Peridinium depressum* Bailey, 1854]

Hsiao (1983) and the references therein; Davis Strait. Jones Sound, Lancaster Sound, western Baffin Bay, Clarence Head, Cape Isabella, Cape Faraday, off Resolution Island, off Bylot Island, off Merchants Bay, in and off Exeter Sound, Igloolik, Creswell Bay, Wellington Channel, Resolute Passage [as *Peridinium depressum* Bailey, 1854]

Hsiao and Pinkewycz (1985b); Frobisher Bay [as *Peridinium depressum* Bailey, 1854]

Percy et al. (1992); Sugluk Inlet

Okolodkov (1998) and the references therein; Arctic Basin, Barents Sea, Chukchi Sea, Kara Sea, Laptev Sea, White Sea

Melnikov et al. (2002); Canada Basin of the Arctic Ocean

Jensen and Veland (2006); Disko Bay

Kubiszyn et al. (2014); off Spitsbergen

Krawczyk et al. (2015); Godthåbsfjord system

Laget (2017); Deception Bay

Dhifallah et al. (2022); mouth of the Churchill River, Deception Bay, Frobisher Bay, Milne Inlet

Protoperidinium mite (Pavillard) Balech, 1974

Bursa (1963); Point Barrow [as *Peridinium mite* Pavillard, 1916]

Hsiao (1983) and the references therein; Davis Strait [as *Peridinium granii* f. *mite* (Pavillard) Schiller, 1937]

Okolodkov (1998) and the references therein; Laptev Sea

Riedel et al. (2003); McDougall Sound

Laget (2017); Deception Bay

Protoperidinium pallidum (Ostenfeld) Balech, 1973

Bursa (1963); Point Barrow [as *Peridinium pallidum* Ostenfeld, 1900]

Anderson et al. (1981); Hudon Bay [as *Peridinium pallidum* Ostenfeld, 1900]

Hsiao (1983) and the references therein; Davis Strait, Jones Sound, Lancaster Sound, western Baffin Bay, Clarence Head, Cape Isabella, Cape Faraday, off Resolution Island, Wellington Channel, Assistance Bay, off Bylot Island, off Merchants Bay, Exeter Sound, Igloolik [as *Peridinium pallidum* Ostenfeld, 1900]

Percy et al. (1992); Sugluk Inlet

Falk-Petersen et al. (1997); northern Barents Sea

Okolodkov (1998) and the references therein; Arctic Basin, Barents Sea, Chukchi Sea, East Siberian Sea, Kara Sea, Laptev Sea, White Sea

Melnikov et al. (2002); Canada Basin of the Arctic Ocean

Jensen and Veland (2006); Disko Bay

Seuthe et al. (2011); Kongsfjorden

Krawczyk et al. (2015); Godthåbsfjord system

Laget (2017); Deception Bay

Dhifallah et al. (2022); mouth of the Churchill River, Deception Bay, Frobisher Bay, Milne Inlet

Protoperidinium pellucidum Bergh, 1882

Bursa (1963); Point Barrow [as *Peridinium pellucidum* (Bergh) Schütt, 1895]

Hsiao (1976); southern Beaufort Sea, Eskimo Lakes [as *Peridinium pellucidum* (Bergh) Schütt, 1895]

Anderson et al. (1981); Hudson Bay [as *Peridinium pellucidum* (Bergh) Schütt, 1895]

Hsiao (1983) and the references therein; Jones Sound, Lancaster Sound, western Baffin Bay, Clarence Head, Cape Isabella, Cape Faraday, off Bylot Island, off Merchants Bay,

off Exeter Sound, Igloolik, Creswell Bay, Beaufort Sea [as *Peridinium pellucidum* (Bergh) Schütt, 1895]

Hsiao and Pinkewycz (1987); western Baffin Bay, Jones Sound [as *Peridinium pellucidum* (Bergh) Schütt, 1895]

Pinkewycz et al. (1987); northeastern Ungava Bay [as *Peridinium pellucidum* (Bergh) Schütt, 1895]

Percy et al. (1992); Sugluk Inlet

Booth and Horner (1997); Arctic Ocean

Falk-Petersen et al. (1997); northern Barents Sea

Okolodkov (1998) and the references therein; Arctic Basin, Barents Sea, Chukchi Sea, East Siberian Sea, Kara Sea, Laptev Sea, White Sea

Lovejoy et al. (2002); northern Baffin Bay

Melnikov et al. (2002); Canada Basin of the Arctic Ocean

Jensen and Veland (2006); Disko Bay

Seuthe et al. (2011); Kongsfjorden

Stewart (2013); southern Beaufort Sea

Kubiszyn et al. (2014); off Spitsbergen

Krawczyk et al. (2015); Godthåbsfjord system

Laget (2017); Deception Bay

Simo-Matchim et al. (2017); Nachvak Fjord

Dhifallah et al. (2022); mouth of the Churchill River, Deception Bay, Frobisher Bay, Milne Inlet, Wager Bay

Protoperidinium quarnerense (Schröder) Balech, 1974

Hsiao (1983) and the references therein; Jones Sound, western Baffin Bay, Clarence Head, off Bylot Island, off Exeter Sound, Beaufort Sea, Lancaster Sound

Okolodkov (1998) and the references therein; Barents Sea, Kara Sea, White Sea

Krawczyk et al. (2015); Godthåbsfjord system

Laget (2017); Deception Bay

Scrippsiella acuminata (Ehrenberg) Kretschmann, Elbrächter, Zinssmeister, Soehner, Kirsch, Kusber & Gottschling, 2015

Bursa (1963); Point Barrow [as *Peridinium trochoideum* (Stein) Lemmermann, 1910]

Hsiao (1983) and the references therein; Igloolik, Davis Strait [as *Peridinium trochoideum* (Stein) Lemmermann, 1910]

Okolodkov (1998) and the references therein; Barents Sea, Chukchi Sea, Kara Sea, White Sea [as *Scrippsiella trochoidea* (Stein) Loeblich, 1976]

Melnikov et al. (2002); Canada Basin of the Arctic Ocean [as *Scrippsiella trochoidea* (Stein) Loeblich, 1976]

Riedel et al. (2003); McDougall Sound [as *Scrippsiella trochoidea* (Stein) Loeblich, 1976]

Seuthe et al. (2011); Kongsfjorden [as *Scrippsiella trochoidea* (Stein) Loeblich, 1976]

Kubiszyn et al. (2014); off Spitsbergen [as *Scrippsiella trochoidea* (Stein) Loeblich, 1976]

Laget (2017); Deception Bay [as *Scrippsiella trochoidea* (Stein) Loeblich, 1976]

Simo-Matchim et al. (2017); Anaktalak Fjord [as *Scrippsiella trochoidea* (Stein) Loeblich, 1976]

Tripos arcticus (Ehrenberg) Gómez, 2021

Bursa (1963); Point Barrow [as *Ceratium arcticum* (Ehrenberg) Maggi, 1880]

Anderson et al. (1981); Hudson Bay [as *Ceratium arcticum* (Ehrenberg) Maggi, 1880]

Percy et al. (1992); Sugluk Inlet [as *Ceratium arcticum* (Ehrenberg) Maggi, 1880]

Falk-Petersen et al. (1997); northern Barents Sea [as *Ceratium arcticum* (Ehrenberg) Maggi, 1880]

Okolodkov (1998) and the references therein; Arctic Basin, Barents Sea, Chukchi Sea, Kara Sea, Laptev Sea, White Sea [as *Ceratium arcticum* (Ehrenberg) Maggi, 1880]

Lovejoy et al. (2002); northern Baffin Bay, off Murchison Sound [as *Ceratium arcticum* (Ehrenberg) Maggi, 1880]

Jensen and Veland (2006); Disko Bay [as *Ceratium arcticum* (Ehrenberg) Maggi, 1880]

Seuthe et al. (2011); Kongsfjorden [as *Ceratium arcticum* (Ehrenberg) Maggi, 1880]

Krawczyk et al. (2015); Godthåbsfjord system [as *Ceratium arcticum* (Ehrenberg) Maggi, 1880]

Laget (2017); Deception Bay

Dhifallah et al. (2022); mouth of the Churchill River, Deception Bay, Frobisher Bay, Milne Inlet

ANNEXE XIII

SUPPLEMENTARY DATA ON THE BULK CARRIERS INVOLVED IN IRON ORE EXPORT

Vessel name	Ballast capacity (m^3)	2016	2017	2018	2019	2020	2021	2022	2023
ADMIRAL SCHMIDT	59 978	0	0	0	0	3	3	0	0
AM BUCHANAN	N/A	0	1	0	1	0	1	1	0
AM GHENT	26 379	0	0	0	0	0	0	0	1
AM HAMBURG	N/A	0	0	1	1	0	1	0	0
AM KRAKOW	N/A	0	0	0	0	0	0	1	1
AM QUEBEC	N/A	0	0	0	1	1	1	1	0
ARKADIA	33 437	2	2	3	3	0	2	2	0
BULK DESTINY	34 021	0	0	1	2	1	0	0	0
BULK ENDURANCE	34 021	0	0	0	2	0	0	0	0
CONRAD OLDENDORFF	27 749	0	0	0	0	0	1	0	0
CORNELIE OLDENDORFF	26 379	0	0	0	0	0	0	1	0
DESPINA V	34 837	0	0	0	2	1	3	0	1
ELENA VE	34 547	0	0	0	2	0	1	2	2
FEDERAL TAMBO	30 467	0	0	1	0	0	0	0	0
FLAG METTE	34 837	0	0	0	2	2	1	1	3
GEBE OLDENDORFF	N/A	0	0	0	2	1	1	0	0
GEORG OLDENDORFF	N/A	0	0	0	1	1	0	0	2
GERDT OLDENDORFF	N/A	0	0	0	0	0	0	1	0
GISELA OLDENDORFF	N/A	0	0	0	2	2	1	0	1

GOLDEN AMBER	35 054	0	2	3	2	1	2	2	2
GOLDEN BRILLIANT	35 116	2	2	2	2	3	2	2	2
GOLDEN BULL	35 116	2	2	2	2	2	2	2	2
GOLDEN DIAMOND	N/A	2	3	2	2	1	2	1	2
GOLDEN ENTERPRISE	20 481	0	0	0	1	0	0	0	0
GOLDEN FAST	N/A	0	0	0	0	0	0	2	2
GOLDEN FORTUNE	N/A	0	0	0	0	0	0	1	1
GOLDEN FORWARD	N/A	0	0	0	0	0	0	1	1
GOLDEN FREEZE	N/A	0	0	0	0	0	2	2	2
GOLDEN FROST	N/A	0	0	0	0	0	2	2	2
GOLDEN FURIOUS	N/A	0	0	0	0	0	0	2	2
GOLDEN ICE	35 949	2	3	3	2	2	2	2	0
GOLDEN OPAL	N/A	0	3	2	2	2	2	2	2
GOLDEN OPPORTUNITY	35 949	2	2	3	2	2	2	0	0
GOLDEN PEARL	N/A	2	2	3	2	0	0	0	0
GOLDEN ROSE	36 072	0	0	0	0	1	1	0	0
GOLDEN RUBY	N/A	1	2	2	2	3	2	2	2
GOLDEN SAGUENAY	35 949	2	2	2	2	2	0	0	0
GOLDEN STRENGTH	35 950	2	2	2	2	0	1	2	0
GOLDEN SUEK	35 054	0	0	2	2	0	0	2	1
GRETKE OLDENDORFF	N/A	0	0	0	0	0	0	0	1
HAUKE OLDENDORFF	N/A	0	0	0	0	0	0	0	2
HEIDE OLDENDORFF	N/A	0	0	0	0	0	0	0	1
KAI OLDENDORFF	N/A	0	0	0	1	0	1	0	0

KENDRA OLDENDORFF	N/A	0	0	0	0	0	1	0	0
KIRA OLDENDORFF	N/A	0	0	0	0	0	0	1	0
KUMPULA	33 438	0	0	0	2	1	0	0	2
LOWLANDS HOPE	34 005	0	0	0	0	0	0	0	1
NORDIC BOTHNIA	16 486	0	0	1	0	0	0	0	0
NORDIC NULUUJAAK	42 816	0	0	0	0	0	1	2	3
NORDIC OASIS	34 023	2	3	3	3	4	3	2	2
NORDIC ODIN	34 023	2	3	3	3	3	3	2	3
NORDIC ODYSSEY	33 661	2	2	3	3	3	2	2	3
NORDIC OLYMPIC	34 023	2	3	3	3	3	3	3	3
NORDIC ORION	33 661	2	3	3	3	3	3	3	3
NORDIC OSHIMA	34 023	2	3	4	3	3	3	2	3
NORDIC QINNGUA	42 816	0	0	0	0	0	3	2	3
NORDIC SANNGIJUQ	42 816	0	0	0	0	0	0	2	2
NORDIC SIKU	42 816	0	0	0	0	0	0	2	3
NORDKAP	34 705	0	0	2	0	0	0	0	0
NORDPOL	22 079	0	0	2	0	0	0	0	0
NS ENERGY	20 704	3	3	2	3	3	1	0	0
NS YAKUTIA	20 704	3	2	3	3	3	2	0	0
PABAL	33 599	0	0	0	1	1	2	0	0
PABUR	33 599	0	0	0	1	2	2	0	0
PATRICIA V	21 009	0	0	0	2	0	0	0	2
REGINA OLDENDORFF	N/A	0	0	0	0	0	0	0	1
RIO GRITA	21 369	0	0	2	0	1	0	0	0

RIO TAMARA	21 366	0	3	2	0	3	0	0	0
RIXTA OLDENDORFF	N/A	0	0	0	0	0	0	0	1
SAGAR SAMRAT	21 220	0	3	3	3	2	3	2	2
SEA EXPRESS	N/A	0	0	0	0	1	0	0	0
SEA NEPTUNE	35 694	0	0	1	1	1	0	0	0
SEA ORPHEUS	20 481	0	0	0	1	0	0	0	0
SEA PLUTO	N/A	0	0	0	0	1	0	0	0
VITUS BERING	N/A	0	0	0	0	3	3	0	0

Columns 3 to 10 show the number of round-trip voyages completed by each vessel over the course of a single shipping season. The information presented in this table is sourced from BIMC (2017; 2018; 2019; 2020; 2021; 2022; 2023a; 2024) and MarineTraffic (www.marinetraffic.com). N/A: not available.

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