INFLUENCE DES ACTIVITÉS HUMAINES SUR LES ÉCOSYSTÈMES BENTHIQUES D'UNE ZONE INDUSTRIALO-PORTUAIRE SUBARCTIQUE

APPLICATION D'INDICATEURS DE STATUT ÉCOLOGIQUE ET PRÉDICTION DE LA STRUCTURE DES COMMUNAUTÉS

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« *Oui, la mer est tout. Je l'aime ! Elle couvre les sept dixièmes du globe terrestre. Son souffle est pur et sain. C'est l'immense désert où l'homme n'est jamais seul, car il sent frémir la vie à ses côtés. Ah ! monsieur, vivez, vivez au sein des mers. Là seulement est l'indépendance ! Là, je ne reconnais pas de maîtres ; là, je suis libre !* »
Vingt mille lieues sous les mers,
Jules Verne

« *C'est dans l'action que l'on reconnait les champions.* »
Slogan d'une enseigne aperçue tous les jours à Sept-Îles,
Inconnu

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Cette thèse est divisée en quatre sections : une introduction présentant les différents concepts et principes utilisés tout au long de la thèse, le corps principal contenant trois chapitres expliquant les analyses effectuées et interprétant les résultats obtenus, une conclusion générale intégrant les différents résultats et proposant des perspectives, ainsi qu'une annexe. Cette dernière comprend un article de perspective, réalisé lors du programme de mentorat de la 4^{e} *World Conference on Marine Biodiversity* traitant de la contribution des évaluations environnementales holistiques pour atteindre les *Aichi Biodiversity Targets*, ainsi qu'un jeu de données, obtenu sur les écosystèmes intertidaux de la région de Sept-Îles lors d'une campagne d'échantillonnage en 2016.

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Dreujou, E., McKindsey, CW., Grant, C., Tréau de Coeli, L., St-Louis, R., Archambault, P. (2020). Biodiversity and habitat assessment of coastal benthic communities in a sub-Arctic industrial harbor area. *Water* 12(9):2424. DOI:10.3390/w12092424. (Choix de l'éditeur)

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Dreujou, E., McKindsey, CW., Desroy, N., Ban, N., Foveau, A., Archambault, P. (2019). How to characterize human influence on coastal benthic ecosystems? Global Change on Estuarine and Coastal Ecosystems Conference (CHEERS), Bordeaux (France), 4-8 novembre.

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Dreujou, E., Archambault, P., McKindsey, CW., Desroy, N., Foveau, A. (2018). Linking anthropogenic activities and benthic communities in industrial and harbor areas: what is the state of the ecosystems? Réunion scientifique du Canadian Healthy Oceans Network, Ottawa (Canada), 29 novembre - 3 décembre. (Prix de la meilleure affiche scientifique)

Dreujou, E., Archambault, P., McKindsey, CW., Desroy, N., Foveau, A. (2018). Relier les activités humaines et les communautés benthiques en zone industrielle-portuaire : quel est l'état des écosystèmes ? Réunion scientifique de Québec-Océan, Rivière-du-Loup (Canada), 5-6 novembre.

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Dreujou, E., Archambault, P., McKindsey, CW. (2017). Individual and cumulative impacts of anthropogenic stressors on coastal ecosystems: the case of Sept-Îles, QC. Réunion scientifique du CHONe II, Gatineau (Canada), 1-5 mai.

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RÉSUMÉ

L'ensemble des environnements côtiers et océaniques de la planète est influencé par les activités humaines, dont les impacts peuvent modifier la structure et l'intégrité des écosystèmes de façon durable. Afin de protéger adéquatement le milieu naturel et de soutenir un développement durable, notamment dans des régions concentrant de multiples activités humaines, il est nécessaire de comprendre comment évoluent les écosystèmes marins sous influence anthropique. Les communautés benthiques sont un compartiment particulièrement intéressant pour étudier ces problématiques, car de nombreuses espèces possèdent des capacités d'évitement limitées de par leur mode de vie majoritairement sessile ainsi qu'une longue espérance de vie. Alors que plusieurs travaux évaluant l'impact anthropique ont été effectués sur une large gamme d'écosystèmes à travers le monde, peu ont considéré spécifiquement des écosystèmes subarctiques, dont l'utilisation par l'homme est prévue d'augmenter en lien avec le changement climatique. C'est dans ce contexte que s'inscrit ma thèse, où l'objectif principal est de comprendre comment les écosystèmes benthiques d'une zone industrialo-portuaire subarctique sont influencés par les activités humaines. La zone d'étude considérée se trouve dans la région de Sept-Îles (Québec, Canada), plateforme économique importante pour le Québec, située dans le Golfe du Saint-Laurent. Pour répondre à ces problématiques, cette thèse est divisée en trois chapitres.

Le premier chapitre a pour but de caractériser la structure écologique des écosystèmes côtiers considérés. Lors de campagnes de terrain et d'analyses en laboratoire, un total de 289 taxons ont été échantillonnés, dont la majorité, présente dans le Golfe du Saint-Laurent, sont des nouvelles mentions dans cette région. Divers paramètres abiotiques du sédiment ont été évalués, tels que la concentration en matière organique, en métaux lourds et la distribution de fractions granulométriques. L'analyse de la similarité des assemblages d'invertébrés de taille supérieure à 0,5 mm a révélé des signes de perturbation dans certaines zones, avec un nombre accru d'espèces tolérant la pollution et d'espèces opportunistes. Des modèles de régression ont permis de mettre en évidence les variables de l'habitat qui impactent le plus la structure des communautés.

Le deuxième chapitre s'intéresse au statut écologique des écosystèmes en se basant sur la composition des communautés benthiques. Seize indicateurs du statut écologique ont été sélectionnés au moyen d'une revue de littérature, puis divisés en trois catégories selon leur méthodologie : mesures d'abondance, diversité des communautés et espèces indicatrices. Ces indicateurs ont été appliqués en utilisant les listes d'espèces obtenues lors du chapitre précédent, et la majorité a détecté des communautés diversifiées sans signe évident de perturbation. De plus, plusieurs corrélations significatives ont été détectées entre les indicateurs et les paramètres de l'habitat, notamment avec les concentrations en métaux lourds. Chaque catégorie d'indicateur apporte des informations importantes sur l'état de l'écosystème tout en présentant des limitations, en particulier à propos des références utilisées pour définir le statut écologique.

Le dernier chapitre a pris en compte les activités humaines influençant l'écosystème, afin de calculer une empreinte anthropique locale sur les communautés selon des gradients d'exposition. Un indice d'exposition pour chaque activité considérée (aquaculture, dragage, influence industrielle, influence municipale, pêcheries, rejets d'égouts, transport maritime) a été développé grâce à la distance depuis leur(s) source(s) et des données de pêche. Plusieurs liens ont été découverts entre les indices d'expositions obtenus et la distribution des invertébrés benthiques, au moyen de modèles prédictifs *Hierarchical Modelling of Species Communities.* L'indice d'exposition cumulée a mis en évidence des zones de superposition d'activité humaine. Le profil des communautés présentes dans ces régions n'est pas particulièrement perturbé, ce qui corrobore les résultats des chapitres précédents sur le statut des écosystèmes considérés.

Cette thèse de doctorat contribue à l'amélioration des connaissances sur les écosystèmes côtiers subarctiques, notamment en présentant la première étude de biodiversité benthique dans la région de Sept-Îles. Des méthodes d'évaluation du statut écologique et de l'exposition anthropique ont été développées à l'échelle locale ($<$ 100 km), ce qui constitue des outils particulièrement intéressants pour les gestionnaires afin de définir des objectifs de gestion et de soutenir des initiatives de conservation.

Mots clefs : écologie marine, écosystèmes côtiers subarctiques, invertébrés benthiques, biodiversité, prédiction des communautés, activités humaines, exposition anthropique, évaluation du statut écologique.

ABSTRACT

Coastal and ocean environments are influenced by human activities worldwide, the impacts of which can significantly modify the structure and integrity of ecosystems. In order to adequately protect the natural environment and support sustainable anthropogenic development, specifically in regions where multiple human activities co-occur, it is necessary to understand how marine ecosystems are influenced. Benthic communities are a particularly interesting compartment for studying these issues, because many species have a limited mobility due to their predominantly sessile lifestyle as well as a long life span. While studies assessing anthropogenic impacts have been carried out on a wide range of ecosystems around the world, few have specifically considered sub-Arctic ecosystems, where human activity is expected to increase in connection with climate change. In this context, my thesis's main objective is to understand how the benthic ecosystems of a sub-Arctic industrial harbour area are influenced by human activities. The study area herein considered is located in the Sept-Îles region (Quebec, Canada), an important economic hub for Quebec, located in the Gulf of St. Lawrence. To address these topics, this thesis is divided into three chapters.

The first chapter aimed to characterize the structure of the considered coastal ecosystems. During field campaigns and laboratory analyses, a total of 289 taxa were sampled, the majority of which, present in the Gulf of St.Lawrence, are new records in this region. Various abiotic parameters of the sediment were assessed, such as the concentration of organic matter, heavy metals and the distribution of particle size fractions. Similarity analysis of invertebrate assemblages larger than 0.5 mm showed signs of disturbance in some areas, with an increased number of pollution-tolerant and opportunistic species. Regression models highlighted which habitat variables had the most impact on the structure of communities.

The second chapter looked at the ecological status of ecosystems based on the composition of benthic communities. Sixteen indicators of ecological status were selected through a literature review, divided into three categories according to their methodology: measures of abundance, community diversity and indicator species. These indicators were applied using the species lists obtained in the previous chapter, and the majority of which identified diverse communities with no obvious sign of disturbance. In addition, several significant correlations were detected between indicators and habitat parameters, especially with heavy metal concentrations. Each category of indicator provided important information on the state of the ecosystem while presenting limitations, in particular about reference conditions used to define ecological status.

The last chapter examined human activities influencing the ecosystem, in order to calculate a local anthropogenic footprint on communities according to exposure gradients. An exposure index for each activity considered (aquaculture, dredging, industrial influence, city influence, fisheries, sewage discharges, shipping) was developed using the distance from their sources and fishing events. Several links were discovered between the exposure indices obtained and the distribution of benthic invertebrates, using predictive models *Hierarchical Modelling of Species Communities*. The cumulative exposure index revealed areas of superposition of human activity. The profile of the communities present in these zones is not particularly disturbed, which corroborates the results of the previous chapters on the status of the ecosystems considered.

This PhD thesis improves ecological knowledge in sub-Arctic coastal ecosystems, in particular by presenting the first benthic biodiversity census in the Sept-Îles region. Methods for assessing ecological status and anthropogenic exposure have been developed at the local scale (< 100 km), which constitute particularly interesting tools for stakeholders in order to define management targets and support conservation initiatives.

Keywords: marine ecology, sub-Arctic coastal ecosystems, benthic invertebrates, biodiversity, community prediction, human activities, anthropogenic exposure, ecological status assessment.

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INTRODUCTION GÉNÉRALE

L'augmentation sans précédent de l'empreinte anthropique sur les écosystèmes depuis le début de l'ère industrielle a motivé la volonté de comprendre la place qu'occupe l'humanité dans le milieu qu'elle habite, en particulier à propos des océans, longtemps considérés comme immarcescibles. La totalité des écosystèmes marins, qu'ils soient côtiers, pélagiques ou profonds, subit au moins une influence d'origine anthropique (Halpern et al., 2019), et l'intensité et la diversité de ces pressions sont en constante augmentation (Smith, 2011; Boonstra et al., 2015). Fortes de ce constat, plusieurs organisations internationales, comme la *Convention on Biological Diversity* sous l'égide des Nations Unies, ont mis en place des objectifs concrets de protection des écosystèmes, dans le but de guider les initiatives de conservation et de développement durable (United Nations, 1992). Ces initiatives représentent des opportunités uniques d'augmenter les interactions entre scientifiques, industriels, politiques et citoyens afin d'accroître et diffuser nos connaissances sur les écosystèmes marins.

Écosystème : communautés et habitats

Un écosystème est un ensemble constitué de deux composantes interconnectées. La première est la composante abiotique, liée aux paramètres physiques, chimiques et géologiques du milieu, qui est représentée par les habitats (ensemble de conditions environnementales particulières) (Allaby, 2010; Morin, 2011). La seconde est la composante biotique, correspondant aux espèces présentes dans ce milieu, regroupées en communautés (ensemble d'espèces vivant dans un habitat particulier) (Allaby, 2010; Morin, 2011). Cette définition inclut également les relations au sein de chaque composante, par exemple les interactions entre individus d'une même espèce (intraspécifiques) ou d'espèces différentes (interspécifiques).

Le terme “biodiversité” est employé pour désigner la diversité biologique d’un écosystème, et celle-ci peut être définie selon plusieurs perspectives, qu’elle soit spécifique (nombre de taxons différents), génétique (nombre des gènes différents), fonctionnelle (nombre de fonctions différentes) ou écosystémique (nombre d’habitats différents) (United Nations, 1992; Wilson, 1992; Gray, 1997; Gaston et Spicer, 2004; Hooper et al., 2005; Stachowicz et al., 2007). Le caractère multiple de cette définition rend son interprétation particulièrement complexe, et la diversité spécifique est généralement employée comme l’indicateur privilégié de la biodiversité (par ex. United Nations, 1992). En effet, les liens entre la diversité spécifique et la structure d’un écosystème ont été mis en évidence aussi bien en milieu terrestre (par ex. Hooper et al., 2005) qu’aquatique (par ex. Giller et al., 2004; O’Connor et Crowe, 2005).

La persistance d’une espèce dans un écosystème est intrinsèquement reliée à son habitat, où elle effectuera des compromis (*trade-offs*) en fonction de son métabolisme et de son comportement afin de maintenir sa population (Morin, 2011). Ceci constitue la théorie de la niche écologique, qui a été conceptualisée par plusieurs auteurs comme un ensemble de conditions environnementales (abiotiques et biotiques) permettant la survie d’une espèce (Grinnell, 1917; Elton, 1927; Hutchinson, 1957; Hardin, 1960; MacArthur et Wilson, 1967). En retour, les espèces induisent des modifications de l’habitat, par exemple en diminuant la disponibilité en ressources ou en modifiant l’intégrité physique du milieu (Schmitz et al., 2008; Stachowicz et al., 2007; Morin, 2011). L’étude de la structure et de l’évolution des écosystèmes est donc dépendante de l’étude de la biodiversité et de l’habitat, ainsi que de leur évolution spatiale et temporelle.

Perturbations de l'écosystème et effets sur les communautés

De nombreuses études écologiques cherchent à comprendre comment réagissent les différentes composantes de l'écosystème face à une perturbation avant, pendant et après son occurrence, notamment dans des buts de conservation ou de restauration. Plusieurs méthodes peuvent être choisies pour répondre à cet objectif, souvent reliées à des disciplines scientifiques particulières, ce qui rend complexes la définition et l'utilisation d'un lexique approprié (Borja et al., 2012; Judd et al., 2015). Tout au long de cette thèse de doctorat, les définitions suivantes, illustrées par la Figure 1, ont été utilisées.

Une "perturbation", dans le sens de *disturbance* en anglais, est une force qui affecte des processus environnementaux et/ou modifie un écosystème hors d'une situation d'équilibre (Odum et al., 1979; Rykiel Jr., 1985; Boonstra et al., 2015; Beauchesne et al., 2020b; Orr et al., 2020). Nombre de synonymes existent pour cette définition, comme "stresseur" (*stressor*), "déterminant" (*driver of change*) ou "pression" (*pressure*), et le débat n'est pas clos quant à savoir quel terme privilégier (Crain et al., 2008; Darling et Côté, 2008; Judd et al., 2015; Côté et al., 2016). L'origine d'une perturbation peut être naturelle, selon des évènements stochastiques (par ex. les tempêtes) ou cycliques (par ex. les saisons), ou bien anthropiques, c'est-à-dire en fonction des activités humaines (Preston et Shackelford, 2002; Micheli et al., 2016). Les conséquences d'une perturbation sont regroupées sous les termes "impact" ou "effet", qui vont entrainer des modifications mesurables de processus physiologiques, fonctionnels ou environnementaux au sein de l'écosystème (Rykiel Jr., 1985; Judd et al., 2015; Orr et al., 2020).

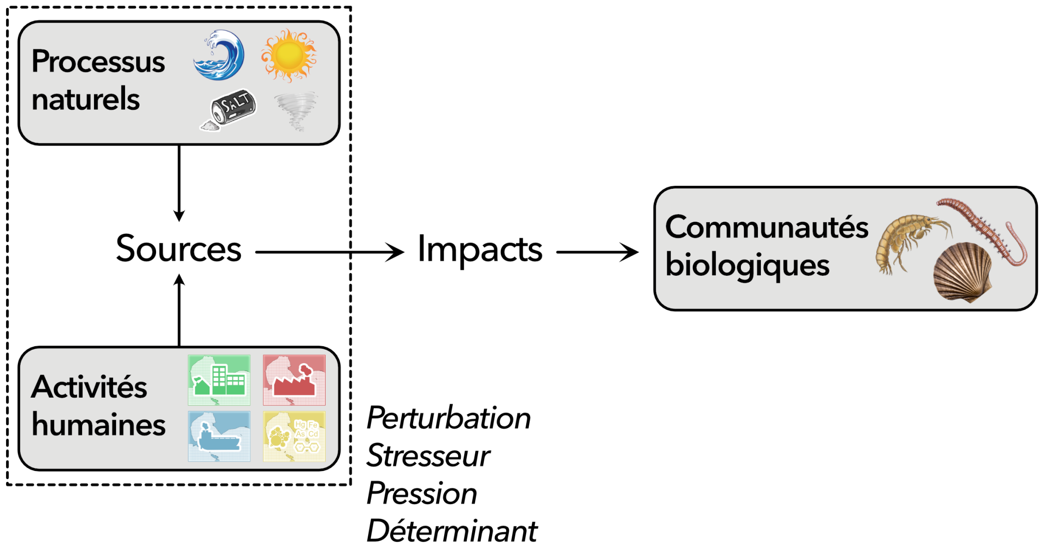

FIGURE 1 – Diagramme représentant les liens entre les sources d'une perturbation (d'origine naturelle ou anthropique) et les communautés benthiques.

Lorsqu'un écosystème subit une perturbation, celle-ci peut donc se traduire par une modification instantanée et/ou durable des composantes abiotique ou biotique (Rykiel Jr., 1985; Piggott et al., 2015). Ces phénomènes n'ont pas nécessairement les mêmes échelles spatiales et temporelles, où certaines perturbations peuvent être ponctuelles et éphémères, d'autres beaucoup plus diffuses et persistantes (Levin, 1992; Witman et al., 2015), ce qui influera sur les notions de résistance (capacité à supporter les effets d'une perturbation), de résilience (temps nécessaire pour atteindre à nouveau l'état avant la perturbation) de l'écosystème.

Au sein des communautés, le maintien de la population d'une espèce est alors conditionné par les réponses qu'elle adoptera pour s'acclimater à ces nouvelles conditions (Allaby, 2010; Morin, 2011). Par exemple, une perturbation peut favoriser des espèces adaptées au caractère instable associé à ces évènements (Pearson et Rosenberg, 1978; Grall et Glémarec, 1997). De telles espèces sont dites opportunistes, présentant une croissance rapide de leur population et une forte densité d'individus de petite taille (appelée stratégie de type r, privilégiant un fort taux de reproduction), en opposition aux espèces longévives et moins abondantes retrouvées dans des environnements plus stables (stratégie de type K, privilégiant le maintien de la population à sa capacité maximale) (MacArthur et Wilson, 1967; Pianka, 1970).

Les communautés benthiques

Avec plus de 70 % de la surface de la planète recouverte par des océans, les écosystèmes marins regroupent une vaste gamme d'environnements. Leurs limites peuvent être définies selon de nombreux critères et en fonction des échelles spatio-temporelles considérées (Webb, 2019). Parmi les critères les plus couramment utilisés se trouvent les écosystèmes pélagiques (dans la colonne d'eau), benthiques (en lien avec les fonds marins), intertidaux (influencés par le cycle des marées), subtidaux (en dessous de la zone de balancement des marées), côtiers (influencés par les apports d'origine terrestre et sur le talus continental) ou hauturiers (au large des côtes) (Webb, 2019). Il existe différents patrons de biodiversité entre ces écosystèmes, et les environnements côtiers subtidaux figurent parmi les plus riches et les plus diversifiés (par ex. Gray, 1997; Gaston, 2000).

Le compartiment benthique contient une diversité élevée d'organismes, incluant des assemblages complexes de virus, bactéries, faune et flore. Afin de répondre à des questions écologiques spécifiques, ces communautés sont souvent séparées en différents sous-ensembles partageant des caractéristiques communes (par ex. Sheldon et al., 1972; Schwinghamer, 1981). Plusieurs catégories d'organismes ont été définies en fonction de leur taille : la microfaune (organismes plus petits que 0,1 mm), la méiofaune (entre 0,1 et 0,5/1 mm), la macrofaune (entre 0,5/1 et 100 mm) et la mégafaune (plus de 100 mm)

(Figure 2) (Schwinghamer, 1981; Warwick et Clarke, 1984; Warwick et al., 2006). La limite séparant la méiofaune et la macrofaune (0,5 ou 1 mm) fait encore aujourd'hui l'objet de débats entre chercheurs ; il s'agit d'une considération importante car le choix de cette limite aura un impact direct sur la résolution taxonomique des communautés étudiées (Mckindsey et Bourget, 2001; Warwick et al., 2006).

Les invertébrés macrobenthiques, qui constituent le principal objet d'étude de cette thèse de doctorat, regroupent les espèces de nombreux phylums vivant en lien avec les fonds marins. Ceux-ci peuvent être sessiles (fixés sur un substrat) ou mobiles, évoluant dans, sur ou au-dessus du sédiment (respectivement l'endobenthos, l'épibenthos et le suprabenthos) (Figure 2). De plus, certaines espèces peuvent être benthiques tout au long de leur vie (cycle holobenthique) ou à seulement des étapes précises de leur développement (cycle bentho-pélagique). La diversité de ces modes de vie est reliée à un nombre important de stratégies évolutives mises en œuvre pour prospérer dans l'habitat, que ce soit des méthodes de nutrition (par ex. filtration de l'eau, ingestion du sédiment, recyclage de matières détritiques), des comportements (par ex. mode de vie, mobilité) ou des interactions biotiques (par ex. symbiose, parasitisme). De plus, les espèces macrobenthiques sont incluses dans des réseaux trophiques complexes, qui permettent les transferts d'énergie à travers l'écosystème.

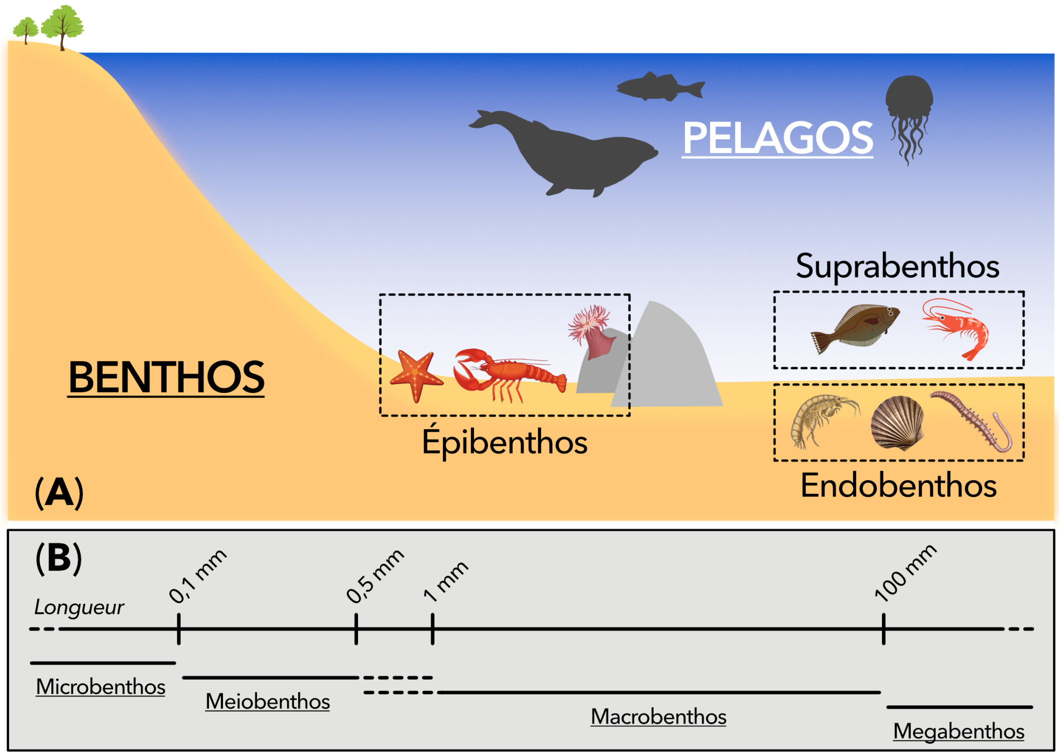

Figure 2 – Représentation schématique de différents groupes d'espèces au sein d'une communauté biologique, en fonction de leur milieu de vie (A) ou de leur taille (B).

Au sein de la macrofaune – et *a fortiori* au sein des communautés en général –, chaque espèce possède un rôle dans le fonctionnement de l'écosystème en lien avec l'utilisation qu'elle fait de son habitat. L'étude de ces rôles est particulièrement intéressante pour comprendre la structure des écosystèmes (Morin, 2011). Par exemple, les espèces ingénieures modifient directement ou indirectement les écosystèmes par leur mode de vie (par ex. en créant des structures biogéniques, pouvant jouer le rôle d'habitats pour d'autres espèces) ou leur comportement (par ex. en oxygénant le sédiment profond avec une activité de bioturbation), ce qui permettra à d'autres espèces de profiter d'habitats particuliers (Meadows et al., 1991, 2012).

Beaucoup de ces espèces possèdent des capacités d'évitement limitées, ce qui peut conduire à une dégradation de l'état des populations et à une mortalité accrue lorsque l'écosystème est perturbé (Grall et Glémarec, 1997). Ceci est notamment le cas pour les espèces sessiles (par ex. les moules ou les anémones), les espèces vivant dans le sédiment (comme les amphipodes ou les mollusques fouisseurs) ou les organismes filtreurs (tels que des polychètes tubicoles ou les éponges) qui sont particulièrement sensibles aux perturbations chimiques affectant la colonne d'eau. La présence (ou l'absence) de ces espèces, ainsi qualifiées de "sentinelles" ou "indicatrices", aura la possibilité de renseigner sur un certain état de l'écosystème (Pearson et Rosenberg, 1978; Hooper et al., 2005; Dauvin et al., 2010).

Influence anthropique et statut environnemental

À l'échelle mondiale, la population humaine a atteint 7,7 milliards d'individus en 2019, et les projections démographiques indiquent qu'elle s'élèvera à 9,7 milliards en 2050 (United Nations, 2019). Avec une proportion importante de cette population qui entretient des liens étroits avec l'océan – environ 40 % vit à moins de 100 km des côtes –, la croissance de l'influence humaine sur les milieux marins est évidente (Ban et Alder, 2008; Smith, 2011; Socioeconomic Data and Applications Center, 2020). De nombreux exemples ont été documentés à travers le monde, mettant ainsi en lumière des conséquences sur les écosystèmes comme l'extinction locale de populations (par ex. la disparition de la morue arctique due à la pêche intensive de ses stocks, Department of Fisheries and Oceans, 2007), l'introduction d'espèces exotiques (par ex. l'introduction de l'ascidie *Botrylus schlosseri* à cause du trafic maritime, Ma et al., 2017a) ou encore la destruction d'habitats (par ex. due à l'exploitation de ressources fossiles, Archambault et al., 2017).

L'étude de l'influence anthropique a souvent été focalisée sur une activité humaine (par exemple le dragage ou la pêche), un processus (telles que les réponses physiologiques à un contaminant) ou un certain type d'écosystème (comme les récifs coralliens ou les forêts de macroalgues) (Borja et al., 2008a). Le déploiement de technologies telles qu'une puissance de calcul informatique et une couverture satellitaire accrues permettent aujourd'hui de considérer l'empreinte humaine d'une façon plus holistique (Micheli et al., 2016; Dreujou et al., 2020a). En particulier, il est possible d'étudier les effets émergents dus aux interactions entre différentes activités humaines, dans le but de mieux comprendre et prédire l'évolution des écosystèmes (Crain et al., 2008; Darling et Côté, 2008; Halpern et Fujita, 2013; Brown et al., 2014; Piggott et al., 2015; Galic et al., 2018; Hodgson et al., 2019; Beauchesne et al., 2020b).

Il existe un éventail de méthodes permettant l'évaluation de l'influence anthropique sur les communautés et les habitats dans le cadre de stratégie marines. Celles-ci peuvent être basées sur des travaux de terrain (comme les plans d'échantillonnage *Multiple Before-After and Control/Impact*), des expériences (par exemple en observant les réponses de communautés selon des conditions contrôles et impactées) ou des modélisations informatiques (Hurlbert, 1984; Legendre et Legendre, 1998; Quinn et Keough, 2002; Underwood, 2002, 2012). Les résultats obtenus peuvent ainsi contribuer à des modèles intégratifs tels que le modèle *Driver-Pressure-State-Impact-Response* (DPSIR), ou les évaluations à stratégie holistique, comme la planification spatiale marine (*marine spatial planning*) ou la gestion basée sur l'écosystème (*ecosystem-based management*), permettent de bâtir des liens nécessaires entre disciplines scientifiques et acteurs environnementaux (Levin et al., 2009; Atkins et al., 2011; Hayes et al., 2015; Borja et al., 2016; Santos et al., 2019). La contribution de ces différents types de travaux aux stratégies marines a été discutée par Dreujou et al. (2020a), avec la mise en évidence de priorités de recherche pour accroître leur efficacité au sein de stratégies de gestion (article présenté en annexe de cette thèse).

En se basant sur ces résultats, il est possible de déterminer un statut environnemental pour les écosystèmes étudiés. La notion de “bon état écologique” a été définie dans l’Article 3 de la Directive-cadre stratégie milieu marin (European Commission, 2008; Borja et al., 2013) :

> *“Good Environmental Status is the environmental status of marine waters where these provide ecologically diverse and dynamic oceans and seas which are clean, healthy and productive.”*

La définition ci-dessus est à coupler avec les onze descripteurs établis par European Commission (2008) afin de considérer le statut de l’écosystème en fonction de concepts intercorrélés (comme la biodiversité, le réseau trophique, l’intégrité du fond marin ou les concentrations en contaminants). Bien que le concept de bon état écologique soit encore débattu, notamment lorsque sont considérées les notions de résistance, résilience, productivité et fonctions, celui-ci permet de fournir une base au développement de stratégies de gestion. Celles-ci définissent un certain nombre d’objectifs et de cibles à atteindre, déterminés en fonction des connaissances scientifiques, des sphères politique et diplomatique, et des écosystèmes sélectionnés. Plusieurs exemples à l’échelle internationale peuvent être cités, tels que l’Accord de Paris sur le climat (cible de 10 % d’aires marines et côtières protégées), les objectifs d’Aichi pour la biodiversité (*Aichi Biodiversity Targets*) et les objectifs de développement durable (*Sustainable Development Goals*) (Secretariat of the Convention on Biological Diversity, 2010; United Nations, 2015; SDG, 2020).

Dans cette optique, les études écologiques peuvent avoir recours à des indicateurs environnementaux, c’est-à-dire des métriques quantitatives qui se basent sur des composantes de l’écosystème pour obtenir un statut environnemental par rapport à des conditions de référence (Rice, 2003; Rees et al., 2008). Les indicateurs dits “biotiques” s’intéressent à la structure des communautés et à l’identité des espèces présentes pour calculer le statut de l’écosystème. Certains indicateurs biotiques sont calculés à partir de mesures d'abondance ou de diversité (par ex. le W-Statistic Index, l'indice de Shannon ou la diversité fonctionnelle), tandis que d'autres considèrent la prévalence de groupes d'espèces caractéristiques en fonction de leur réponse à une perturbation (par ex. le AZTI Marine Biotic Index) (Pinto et al., 2009; Teixeira et al., 2016). Même si beaucoup de ces méthodes ont fait leurs preuves, elles peuvent être complexes à appliquer pour différentes régions et perturbations, ou être dans l'incapacité de détecter adéquatement la variation des écosystèmes (Niemi et McDonald, 2004; Pinto et al., 2009).

Différentes voies d'amélioration ont été proposées, comme l'intégration de perturbations multiples, de gradients de perturbation ou la considération de réseaux d'interactions dans l'écosystème. Le score d'impact cumulé développé par Halpern et al. (2008) constitue un exemple récent (mis à jour par Halpern et al., 2015, 2019), où plusieurs stresseurs anthropiques ont été caractérisés à l'échelle globale. Cette méthode combine deux propriétés de l'écosystème pour calculer un score d'impact, l'exposition d'une part (occurrence d'une perturbation sur des composantes de l'écosystème) et la vulnérabilité d'autre part (susceptibilité des composantes de l'écosystème à une perturbation) (Wilson et al., 2005; Halpern et al., 2007, 2008). L'indice est ainsi calculé selon l'équation ci-dessous, où D est la valeur d'un stresseur, E est la présence/absence d'un écosystème, μ est un coefficient d'impact, i est un stresseur (jusqu'à n), j est un écosystème (jusqu'à m) :

$$I_C = \sum_{j=1}^{n} \frac{1}{n} \sum_{i=1}^{m} D_i.E_j.\mu_{i,j}$$

En l'état des connaissances, considérer la vulnérabilité de façon adéquate demande un large volume de données (par ex. sur les tolérances physiologiques des espèces, les interactions biotiques ou les modes d'action des perturbations) qui peuvent être spécifiques à des écosystèmes particuliers, ce qui augmente rapidement la complexité de ces évaluations (Wilson et al., 2005; Beauchesne et al., 2020b). Une solution possible est de se concentrer sur l'exposition, afin de cibler quels compartiments pourraient être influencés par des perturbations spécifiques, et ainsi focaliser les efforts d'évaluation de la vulnérabilité sur les interactions déterminées pertinentes.

L'évaluation du statut de l'écosystème peut être réalisée à différentes échelles spatiales selon les systèmes écologiques, les objectifs de conservation ainsi que les unités de gestion considérées. Bien que les études régionales et mondiales permettent d'obtenir des informations pertinentes sur les tendances à large échelle et de mobiliser un ensemble de nations pour atteindre des objectifs communs, il est important de considérer des études à plus petite échelle (par ex. à l'échelle d'un estuaire, d'une baie, d'une côte) car les processus écologiques ne sont pas identiques et d'autres types de réponses peuvent être détectés (Comín et al., 2004; Crowe et Frid, 2015; Solan et Whiteley, 2016). Ceci met en lumière la nécessité d'adapter les recommandations d'une étude écologique à l'échelle spatiale considérée pour obtenir des systèmes de gestion efficaces, où la mobilisation des parties publiques, industrielles et scientifiques permet d'atteindre les objectifs de protection environnementale.

Application dans le Golfe du Saint-Laurent

En 2019, 13,8 % des eaux du Canada étaient régulées par une forme de gestion publique et 8,9 % (511 906 km^2) étaient officiellement protégées (Environment and Climate Change Canada, 2020). Ce total pourrait augmenter à 30 % d'ici 2030, suite à la décision de rejoindre la *Global Ocean Alliance* (Government of Canada, 2020). Le long de la côte Est du Canada, le complexe de l'Estuaire et du Golfe du Saint-Laurent est l'une des régions qui concentre le plus d'activités humaines (Belley et al., 2010; Daigle et al., 2017; Schloss et al., 2017). Le transport de marchandises depuis l'Atlantique vers les Grands Lacs est un axe commercial majeur, à partir duquel sont connectées de nombreuses chaînes d'approvisionnement au Canada et aux États-Unis (Gouvernement du Québec, 2017).

Le Golfe du Saint-Laurent comprend les eaux entre Pointe-des-Monts jusqu'aux détroits de Cabot et Belle-Isle (Figure 3). Cette région est considérée subarctique, subissant ainsi de fortes variations de température et de salinité dues à la formation de glace sur les côtes et de banquise en hiver, ainsi que d'importants apports d'eau douce suite au dégel au printemps (Dutil et al., 2012; Demers et al., 2018). Il s'agit d'une zone importante de biodiversité, avec la présence d'espèces emblématiques, telles que des mammifères marins (par ex. le béluga ou la baleine à bosse) (Schloss et al., 2017; ROMM, 2020), et de nombreuses espèces d'intérêt commercial pour la pêche (par ex. le crabe des neiges, la crevette nordique, le flétan de l'Altantique) (Department of Fisheries and Oceans, 2019). Plusieurs initiatives à l'échelle du Québec et du Canada font de la sauvegarde des écosystèmes dans le système du Saint-Laurent une priorité (Gouvernement du Québec, 2015), et le Ministère des Pêches et Océans a désigné cette région comme une aire de gestion intégrée avec des objectifs de protection des écosystèmes (Department of Fisheries and Oceans, 2009, 2013).

Selon Beauchesne et al. (2020b), plusieurs régions concentrent un nombre élevé d'activités humaines, notamment le long des côtes à l'ouest du Golfe. Ceci est d'autant plus vrai pour les zones industrialo-portuaires, qui correspondent à un espace délimité servant à des fins industrielles et situé à proximité de services portuaires ainsi que d'infrastructures routières et ferroviaires (Government of Canada, 2020). Parmi les zones industrialo-portuaires présentes dans le Golfe du Saint-Laurent, celle située à Sept-Îles (région Côte-Nord) est certainement la plus importante. Sept-Îles est le 2e port québécois, le

4e canadien, et en 2019, 29,3 millions de tonnes de marchandises y ont été échangées (Binkley, 2020; Port de Sept-Îles, 2020). Les principales industries sont l'exportation du minerai acheminé par train depuis les mines du nord du Québec et du Labrador, la production d'aluminium, la pêche commerciale (crabe des neige, crevette nordique, flétan et bulot), ainsi que le tourisme (Department of Fisheries and Oceans, 2019).

La région maritime de Sept-Îles est composée de deux éléments géographiques d'importance : la Baie des Sept Îles et l'archipel à son entrée (Figure 3). La profondeur moyenne dans la baie est de 35 m et peut atteindre 150 m dans les chenaux de l'archipel (Dutil et al., 2012), et la circulation est de type estuarienne avec une forte influence de cycle des marées (Shaw, 2019). Le climat de cette région est considéré comme subarctique, avec une végétation et une hydrologie particulière au nord du Québec, ainsi que la formation de glace sur les côtes en novembre/décembre et leur dégel en avril (Demers et al., 2018).

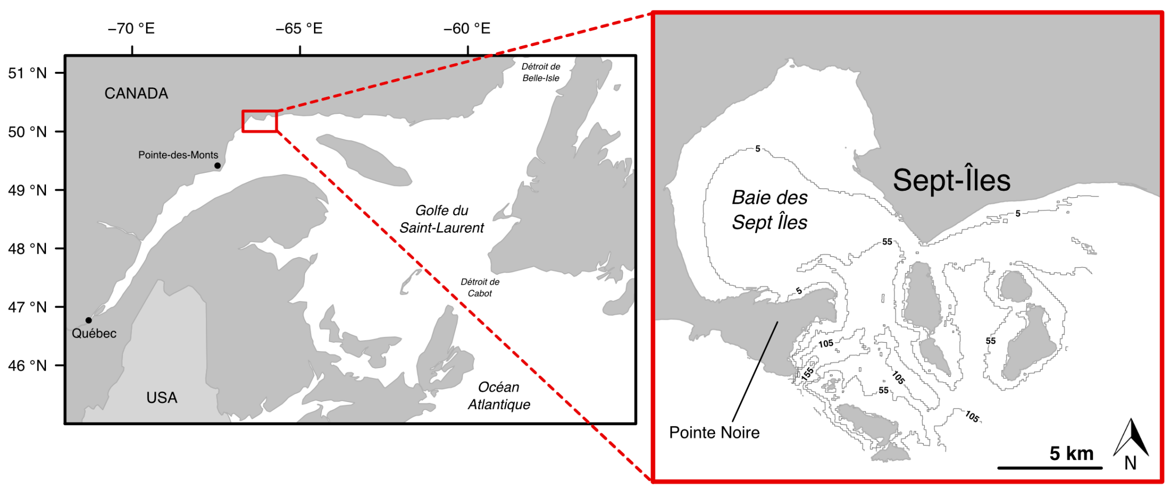

FIGURE 3 – Carte de la zone d'étude considérée tout au long de cette thèse de doctorat. Les isobathes ont été représentées dans la Baie des Sept Îles (mètres).

Peu d'études écologiques ont cherché à caractériser ces écosystèmes ainsi que leur environnement abiotique, ce qui pose un défi conséquent dans le cadre de la stratégie de protection des écosystèmes dans le système du Saint-Laurent. La zone industrialo-portuaire de Sept-Îles représente donc un cadre d'étude particulièrement intéressant afin de comprendre comment les activités humaines influencent des écosystèmes côtiers canadiens à l'échelle locale.

Objectifs et structure de la thèse

Objectifs spécifiques

L'objectif principal de cette thèse de doctorat est de décrire les relations entre communautés benthiques et activités humaines à l'échelle d'une zone industrialo-portuaire subarctique, notamment dans un contexte d'évaluation du statut environnemental. Pour répondre à cet objectif, la thèse est divisée en trois chapitres, illustrés par le schéma conceptuel de la Figure 4.

En premier lieu, j'ai réalisé la première description synthétique de la structure des écosystèmes benthiques côtiers dans la région de Sept-Îles, permettant de disposer de données importantes sur les communautés macrobenthiques et leurs habitats (chapitre 1). En me basant sur ces résultats, j'ai ensuite cherché à déterminer quel était le statut environnemental de la zone industrialo-portuaire de Sept-Îles en testant plusieurs indicateurs écologiques (chapitre 2). Enfin, partant du fait que beaucoup de ces indicateurs possèdent des limitations dans leur interprétation, j'ai développé un modèle local pour caractériser l'exposition des communautés benthiques aux activités humaines et j'ai testé l'efficacité de ce modèle à prédire la structure des communautés benthiques (chapitre 3).

Une section d'annexes vient compléter ce travail en proposant d'une part une réflexion sur la contribution des évaluations environnementales holistiques aux objectifs d'Aichi pour la biodiversité et des priorités de recherche pour augmenter leur potentiel en terme de gestion des écosystèmes marins et d'autre part les jeux de données d'une étude pilote dans la région de Sept-Îles qui s'est intéressée aux écosystèmes intertidaux.

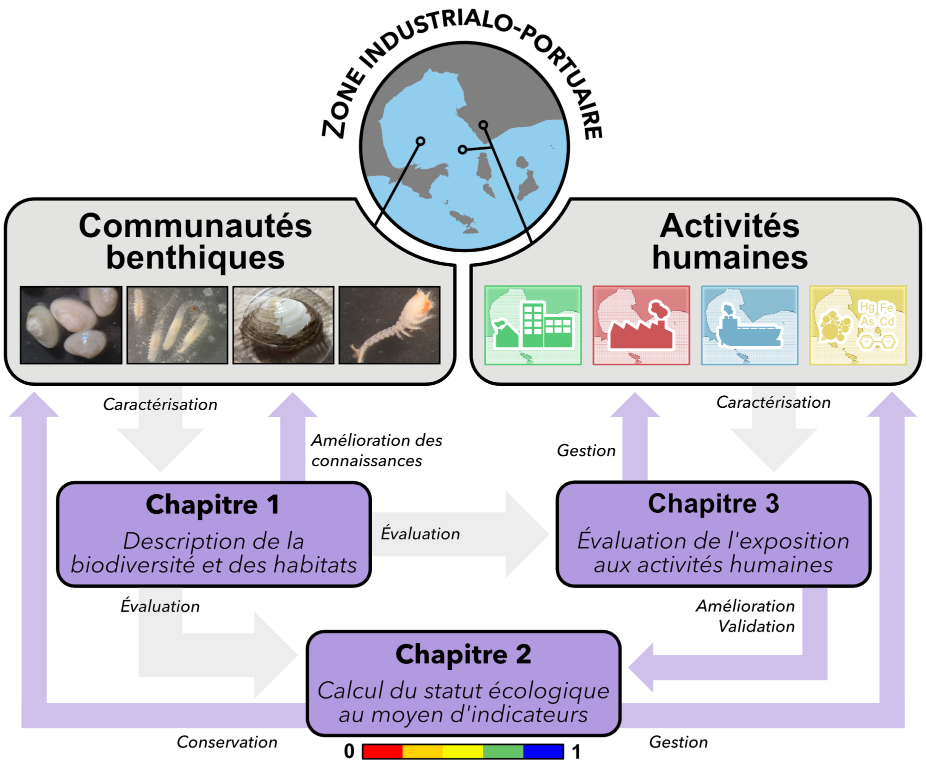

FIGURE 4 – Diagramme intégratif représentant les liens entre les différents chapitres de la thèse de doctorat. Les flèches grises correspondent aux liens entre composantes considérées des écosystèmes de la région d'étude (encadrés gris) et les chapitres (encadrés pourpres). Les flèches pourpres correspondent au débouché de chaque chapitre.

Structure des chapitres

Chapitre 1 : Évaluation de la biodiversité et de l'habitat des communautés benthiques côtiers en zone industrialo-portuaire subarctique

La région de Sept-Îles possède une faible quantité de données décrivant les communautés et habitats benthiques. Afin de soutenir l'utilisation de méthodes d'évaluation environnementale et de décrire efficacement les réponses des écosystèmes aux perturbations, ce chapitre a pour objectifs :

— Décrire les habitats et communautés macrobenthiques dans la région de Sept-Îles
— Évaluer les liens entre les communautés et les variables abiotiques de leur habitat
— Étudier la similarité des assemblages benthiques dans la zone considérée
— Évaluer comment des choix méthodologiques, tel que le choix de la taille des organismes échantillonnés, influence la description de l'écosystème

Deux campagnes de terrain ont permis de récolter les données écologiques nécessaires pour répondre à ces objectifs, en particulier grâce à l'identification des espèces macrobenthiques et la mesure de variables abiotiques. Plusieurs groupes de stations d'échantillonnage ont été formés en fonction de la similarité de leurs assemblages d'espèces, et leur relation avec l'habitat a été évaluée afin de comprendre quelles étaient les variables abiotiques structurantes.

Chapitre 2 : Déterminer le statut écologique de communautés benthiques côtières : étude de cas en zone anthropisée subarctique

En se basant sur les travaux réalisés pour le chapitre 1, il est possible de se demander si les écosystèmes étudiés sont en bon état écologique, avec des perspectives de gestion et de conservation. Ainsi, les objectifs de ce chapitre sont :

— Comparer les spécificités de différents indicateurs environnementaux

— Valider les résultats obtenus en mettant en relation le statut environnemental obtenu avec les paramètres de l'habitat
— Identifier les avantages et inconvéniants de chaque indicateur

Les indicateurs ont été sélectionnés au moyen d'une revue de littérature puis classés selon leur méthodologie. Ils ont ensuite été appliqués sur les données du chapitre 1, après recherche d'informations complémentaires sur les espèces benthiques échantillonnées, notamment leur diversité et leurs réponses aux perturbations. Enfin, les corrélations entre indicateurs et paramètres de l'habitat ont été testées pour comprendre la pertinence de chaque indicateur.

Chapitre 3 : Description de l'exposition des écosystèmes benthiques côtiers aux activités humaines à l'échelle locale

Ce dernier chapitre entreprend de mettre en relation les communautés macrobenthiques avec les activités humaines présentes dans cette région en décrivant l'exposition anthropique des écosystèmes à l'échelle locale. Les objectifs spécifiques sont :

— Modéliser localement l'exposition des écosystèmes benthiques aux activités humaines
— Évaluer la structure des communautés en fonction de leur exposition anthropique

Pour ce chapitre, des modèles d'exposition basés sur la diffusion spatiale de particules et les évènements de pêche ont été développés pour déterminer les régions les plus exposées à différentes activités humaines locales. Le principal défi a été de modéliser adéquatement le comportement des particules dans un environnement où peu de données, notamment sur l'hydrodynamisme, étaient disponibles. Cet indice a caractérisé les influences individuelles et cumulées des activités humaines locales. Des modèles joints de distribution d'espèces – qui prennent en compte les interactions entre les taxons présents au sein d'une communauté – ont été utilisés pour comprendre les liens entre la structure des communautés benthiques d'une part et les variables de l'habitat et les indices d'exposition d'autre part.

ARTICLE 1

ÉVALUATION DE LA BIODIVERSITÉ ET DE L'HABITAT DES COMMUNAUTÉS BENTHIQUES CÔTIERS EN ZONE INDUSTRIALO-PORTUAIRE SUBARCTIQUE

1.1 Résumé

Les écosystèmes côtiers sont confrontés à des pressions anthropiques croissantes dans le monde entier et leur gestion nécessite une évaluation et une compréhension solides des impacts cumulatifs des activités humaines. Cette étude évalue la variation spatiale des communautés macrofauniques benthiques, des sédiments et des métaux lourds dans les écosystèmes côtiers subarctiques autour de Sept-Îles (Québec, Canada) – une zone portuaire importante dans le Golfe du Saint-Laurent. Les propriétés physiques des sédiments ont varié dans la zone étudiée, avec un profil général sablo-vaseux, sauf à des endroits spécifiques de la Baie des Sept Îles où des concentrations plus élevées de matière organique et de métaux lourds ont été détectées. Les assemblages macrofauniques ont été évalués pour deux classes de taille de taxons (organismes $>$ 0,5 mm et $>$ 1 mm) et ils ont été reliés aux paramètres de l'habitat à l'aide de modèles de régression. Des communautés d'organismes plus petits ont montré des signes de perturbation pour un assemblage proche des activités industrielles de la Baie des Sept Îles, avec un nombre accru d'espèces tolérantes et opportunistes, contrairement aux régions voisines dont la composition était similaire à celle d'autres écosystèmes dans le Golfe du Saint-Laurent. Cette étude augmente le connaissances sur les communautés benthiques subarctiques et contribue aux programmes de surveillance des écosystèmes en zone industrialo-portuaire.

L'article associé à ce chapitre, "*Biodiversity and habitat assessment of coastal benthic communities in a sub-Arctic industrial harbour area*", a été co-rédigé avec Christopher W. McKindsey, Cindy Grant, Lisa Tréau de Coeli, Richard St-Louis et Philippe Archambault. Il a été publié dans le journal *Water*, dans la section spéciale *Quantifying the Effects of Global Change on the Distribution and Quality of Aquatic Resources*, le 28 août 2020. J'ai établi les objectifs de ce chapitre avec Christopher W. McKindsey et Philippe Archambault, et j'ai effectué la collecte de données sur le terrain en 2016 et en 2017 avec le soutien de plusieurs stagiaires sous ma supervision. J'ai compilé les bases de données et effectué les analyses statistiques, tout en intégrant les données et résultats de la campagne 2014 effectuée par Cindy Grant et Lisa Tréau de Coeli. J'ai dirigé la rédaction de l'article, où l'ensemble des co-auteurs a contribué à l'interprétation des résultats en fonction de leur expertise et à la révision générale. Les données liées à cet article sont accessibles dans le dépôt en ligne hébergé par le site Scholars Portal Dataverse avec l'identifiant unique 10.5683/SP2/5LJYXO. Les résultats obtenus durant ces travaux ont été présentés lors de la Réunion Scientifique Annuelle de Québec-Océan à Rivière-du-Loup en novembre 2017, la 4^{e} *World Conference on Marine biodiversity* à Montréal en mai 2018 et le Colloque International sur la Recherche Scientifique Industrielle Portuaire à Sept-Îles en mai 2019.

Dreujou, E., McKindsey, CW,. Grant, C., Tréau de Coeli, L., St-Louis, R., Archambault, P. (2020). Biodiversity and habitat assessment of coastal benthic communities in a sub-arctic industrial harbor area. *Water* 12(9):2424. DOI:10.3390/w12092424.

Les sections suivantes correspondent à celles de l'article publié.

BIODIVERSITY AND HABITAT ASSESSMENT OF COASTAL BENTHIC COMMUNITIES IN A SUB-ARCTIC INDUSTRIAL HARBOUR AREA

1.2 Abstract

Coastal ecosystems face increasing anthropogenic pressures worldwide and their management requires a solid assessment and understanding of the cumulative impacts from human activities. This study evaluates the spatial variation of benthic macrofaunal communities, sediments, and heavy metals in the sub-Arctic coastal ecosystems around Sept-Îles (Québec, Canada) – a major port area in the Gulf of St. Lawrence. Physical sediment properties varied in the studied area, with a general sandy-silty profile except for specific locations in Baie des Sept Îles where higher organic matter and heavy metal concentrations were detected. Macrofaunal assemblages were evaluated for two taxa size classes (organisms $>$ 0.5 mm and $>$ 1 mm) and linked to habitat parameters using regression models. Communities of smaller organisms showed signs of perturbation for one assemblage close to industrial activities at Baie des Sept Îles, with an increased number of tolerant and opportunistic species, contrasting to neighboring regions whose compositions were similar to other ecosystems in the Gulf of St. Lawrence. This study enhances the understanding of sub-Arctic benthic communities and will contribute to monitoring programs for industrial harbor ecosystems.

Keywords: biodiversity, coastal benthos, macrofauna, sub-Arctic ecosystems, Gulf of St. Lawrence

1.3 Introduction

It is now widely recognized that marine ecosystems worldwide are susceptible to human-induced disturbances (Halpern et al., 2019), and benthic coastal habitats rank among the most vulnerable (Halpern et al., 2007; 2019). When human activities influence ecosystems, communities may be modified by the loss of sensitive species and the selection of tolerant or opportunistic species, for example (Pearson and Rosenberg, 1978; Sala and Knowlton, 2006). Such changes in species composition concomitantly impact ecosystem structure, affecting community characteristics (*e.g.*, species richness, evenness) or ecosystem functional diversity (Brander et al., 2010; Piot et al., 2014; Edie et al., 2018; Lacoste et al., 2018). Understanding the links between disturbances and community responses is therefore relevant to understanding ecosystem evolution and stability.

In the context of increasing anthropogenic influences on marine ecosystems, a better understanding of ecosystem effects is needed for efficient conservation and management measures, along with sustainable development. Many organizations have highlighted the importance of biologically diverse ecosystems for mankind and have set targets to preserve them and guide decision makers (*e.g.*, United Nations, 1992; Secretariat of the Convention on Biological Diversity, 2010; SDG, 2020). To achieve these goals, tools are needed to detect and manage human influences on the environment.

Many factors influence ecosystems, from large-scale (*e.g.*, climate, oceanic circulation) to small-scale phenomena (*e.g.*, sediment perturbation, species interactions), which are often location- and temporal-specific (Brown et al., 2014). This complexity is a major concern for environmental assessment studies, where the goal is to detect possible human influence in a context of natural variability. Ecosystem status is considered in relation to a reference condition that corresponds to pristine or low human-influenced conditions, which rely upon the best knowledge available (Borja et al., 2012). Ecological groundwork, such as biodiversity surveys, time series monitoring, or experimental studies, is then required to understand the ecosystem structure and stability and how to define accurate reference conditions for environmental assessments.

Macrofauna plays an important role in the structure and functioning of benthic marine ecosystems (Dauvin and Ruellet, 2007; Pratt et al., 2014). Examples of this include engineering species (*e.g.*, structural features for other species, bioturbation) or interactions with nutrient cycles (*e.g.*, nutrient burial in the sediment, remineralization, benthos-pelagos coupling) (Largaespada et al., 2012; Link et al., 2013; Belley et al., 2016; Bourque and Demopoulos, 2018). As many macrobenthic species have a sedentary lifestyle and a relatively long lifespan, they can serve as indicators of the ecological status to assess and predict human influences (*e.g.*, Dauer, 1993; Borja et al., 2000). Finally, the macrofaunal compartment includes a variety of individual body sizes, from larvae to fully grown adults, and distinctions from meiofauna can be difficult to establish (Eleftheriou and McIntyre, 2005). The definition of the body size range is an important consideration for ecosystem biodiversity studies, as it will be a trade-off between increased biodiversity data by including smaller organisms and increased fieldwork and identification time (Mckindsey and Bourget, 2001; Gage et al., 2002; Couto et al., 2010).

In Eastern Canada, the Gulf of St. Lawrence is subjected to a variety of human activities, including industrial centers, commercial shipping, harbors, fisheries, and aquaculture (Belley et al., 2010; Daigle et al., 2017; Schloss et al., 2017). Many ecological surveys on benthic invertebrates are available, in particular thanks to periodic stock assessment surveys, but few have specifically targeted coastal and shallow waters. One of the coastal areas likely most influenced by human activities is around Sept-Îles (Québec). In 2011, Sept-Îles harbor housed the sixth-largest Canadian port in terms of total exchanged goods, was second in terms of loaded tonnage shipped internationally (Statistics Canada, 2011), and port infrastructure and activity have been expanding since. Industrial activities at Sept-Îles are largely focused on international shipping of iron ore mined in northern Québec and Labrador and the production of aluminum, and there are many fisheries operating in Baie des Sept Îles (mainly snow crab, northern shrimp, halibut, and whelk) (Department of Fisheries and Oceans, 2019). Sept-Îles ecosystems are considered sub-Arctic, with sea ice formation in November/December and an important freshwater run-off due to snowmelt in April (Demers et al., 2018). To our knowledge, no studies have focused on benthic communities in this area. Thus, study of the marine ecosystems at Sept-Îles and neighboring regions will increase the understanding of sub-Arctic benthic ecosystems under anthropogenic influence and enhance local environmental assessments.

The objectives of this study are to: (i) describe coastal habitats and macrobenthic communities at Sept-Îles, in order to provide the first benthic ecological survey of the area; (ii) evaluate the links between communities and abiotic variables (including heavy metals); (iii) detect the presence of similar benthic assemblages, along with their relationships with habitats; (iv) evaluate how macrofaunal size range will influence community descriptions and relationships with habitat parameters. We expect that human activities in the region will influence the benthic community structure.

1.4 Methods

1.4.1 Study area

We targeted ecosystems in the Côte-Nord region of Québec, within four adjacent sectors distributed along a 200 km coastline: Baie des Sept Îles, the coast of Port-Cartier, and the entrances of the Pentecôte and Manitou rivers (Figure 5a). This area hosts several human activities, in particular an industrial harbor and related industrial operations at Sept-Îles, at the Pointe-Noire terminal (on the southern section of Baie des Sept

Îles) and the eastern side of Port-Cartier. Coastal ecosystems at the entrances of the Pentecôte and Manitou rivers are subjected to limited human influences. A village is located at Pointe-des-Anglais (south of the Pentecôte River), with a summer peak of use, particularly by tourists, and a decommissioned harbor. The Manitou River is relatively pristine, without hydroelectric plants or major human settlements.

Baie des Sept Îles is characterized by sandy beaches and tidal marshes, with a mean depth of 35 m before the entrance of the archipelago (Dutil et al., 2012). It is influenced by freshwater inputs from multiple streams and strong tidal currents resulting in a mixed water column and an estuarine circulation (Shaw, 2019). The other sectors present a mix of rocky and sandy coasts, with a steep bathymetry to as deep as 200 m.

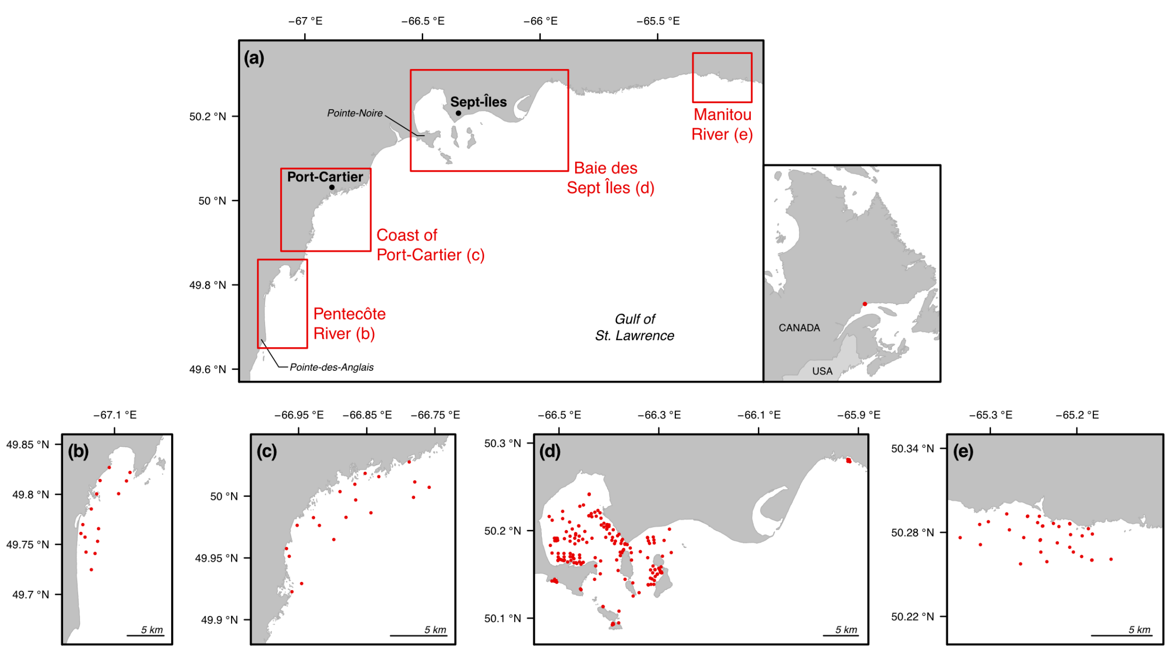

Figure 5 – Maps of the study area. (a) Location of the considered sectors; (b) Stations sampled in the Pentecôte River sector; (c) stations sampled in the coast of Port-Cartier sector; (d) stations sampled in Baie des Sept Îles sector; (e) stations sampled in the Manitou River sector.

1.4.2 Sample collection

We sampled coastal benthic ecosystems during three field campaigns: September 2014, June-July 2016, and July 2017. Sampling stations were positioned in each sector using a randomization algorithm, constrained between 0 m and 80 m deep. A total of 242 stations were sampled during these campaigns, with 175 in Baie des Sept Îles (Figure 5b), 19 in coast of Port-Cartier (Figure 5c), 15 at Pentecôte River (Figure 5d), and 33 at Manitou River (Figure 5e).

Benthic samples were collected using a Ponar grab (0.05 m^2) deployed from a boat with two independent casts. Station depth was obtained from the navigation sonar, then corrected with respect to tide height at time of sampling. The first cast collected three samples for the analyses of organic matter content, sediment grain-size, and heavy metal concentrations (habitat parameters). These samples were stored at -20 °C until processing in the laboratory. All the sediment obtained with the second cast was conserved for benthic macrofauna identification.

Two sieve mesh sizes were considered for macrofaunal samples to provide information on how the size range of the sampled individuals influences community descriptions and relationships with habitat parameters. Sediments were then sieved with either a 0.5 mm (2014 and 2017) or a 1 mm mesh size (2016 and 2017). This resulted in the inclusion of 166 stations for the 0.5 mm size class, located in Baie des Sept Îles and the Manitou River sectors, and 202 stations for the 1 mm size class, in all four sectors. Retained individuals were preserved in a solution of BORAX-buffered formalin (4%).

1.4.3 Laboratory work

1.4.3.1 Habitat parameters

All samples collected during the three field campaigns were processed for organic matter and grain-size analyses. The percentage of total organic matter (*i.e.*, sum of organic carbon and organic nitrogen) in the sediment was obtained by using the Loss-on-Ignition method (Davies, 1974). Grain-size analysis was done on a sieving column for the fraction with particles larger than 2 mm and with a Laser Diffraction Particle Size Analyzer for the smaller fractions. Results from both techniques were combined to yield a unified distribution range from 0.04 μm to 26.5 mm. From this, percentages of gravel, sand, silt, and clay were calculated as defined by Wentworth (1922) and Folk (1980).

Heavy metals were analyzed only for stations sampled in 2014 and 2016 in Baie des Sept Îles, due to practical and logistical constraints. Samples were processed at the analytical chemistry laboratory of Institut des Sciences de la Mer (Université du Québec à Rimouski, Rimouski), using Inductively Coupled Plasma Mass Spectrometry following a microwave mediated acid digestion of the sediment (Centre d'Expertise en Analyse Environnementale du Québec, 2014). We focused on metals for which toxicity criteria have been defined in the Biological Effects Database for Sediments by Environment Canada and Ministère du Développement Durable de l'environnement et des Parcs du Québec (2007): arsenic, cadmium, chromium, copper, mercury, lead, and zinc. We used this study to detect the toxicity of the sediment on benthic species: five levels from Rare to Frequent Effects (corresponding to mild to high toxicity levels) have been defined by aggregating experimental and field studies on the tolerance of species to concentrations of heavy metals (Centre d'Expertise en Analyse Environnementale du Québec, 2014). Iron and manganese were also added to this analysis to account for possible contamination from local ore industries. To have a significant number of stations for the statistical analyses and increase our spatial coverage, heavy metal concentrations for sediments from stations sampled in 2017 in Baie des Sept Îles were calculated based on 2014 and 2016 values with Inverse Distance Weighting interpolation (Dale and Fortin, 2014).

1.4.3.2 Biological samples

Samples for macrofauna identification were sorted using a stereomicroscope. Individuals were identified to the lowest taxonomic level possible with reference manuals and identification guides, and names were validated according to the World Register of Marine Species (WoRMS Editorial Board, 2020). Taxon density was recorded for each station by counting individuals collected per grab.

1.4.4 Statistical analysis

Statistical analyses were done using R v4.0 and PRIMER-e v6 with the PERMANOVA+ package (Clarke and Gorley, 2006; R Core Team, 2020). Habitat parameters were standardized by their mean and standard deviation prior to analysis and species densities were $\log(x + 1)$ transformed to avoid the highest values from dominating analyses. To address the fourth objective of this study, all following statistical analyses were done independently for communities of individuals retained by the 0.5 mm sieve (hereafter referred to as the 0.5 mm size class) and by the 1 mm sieve (the 1 mm size class) in order to compare their outcomes.

The Chao2 estimator was calculated to estimate the maximum expected number of taxa in the study area, and a rarefaction curve was computed to present how sampling effort was related to the observed biodiversity (Chao, 1987; Gotelli and Colwell, 2001). Taxa richness and total density of individuals were calculated for each station, along with Shannon diversity (base e logarithm), Pielou evenness, and taxonomic distinctness indices in order to provide integrative information on community structure, the relative prevalence of constituent taxa, and their taxonomic breadth (Warwick and Clarke, 1995; Clarke and Warwick, 1998; Legendre et al., 2005; Magurran and McGill, 2011).

To detect assemblages of similar taxa, we performed hierarchical agglomerative clustering based on Ward's method (Legendre and Legendre, 1998). The optimal number of clusters was determined using a K-means algorithm (Legendre and Legendre, 1998). Community variability within each cluster (*i.e.*, a measure of β diversity) was assessed by calculating Bray-Curtis dissimilarity. Characteristic taxa, *i.e.*, taxa that explained a significant proportion of within-cluster similarity, were identified using the similarity percentage routine (SIMPER; 9999 permutations) and the indicator value score (IndVal; 1000 randomization iterations) (Clarke, 1993; Dufrêne and Legendre, 1997). We calculated average values of abiotic variables for stations grouped within each cluster to detect possible ecological patterns and spatial relationships.

We examined relationships between the benthic community (independent variables) and habitat parameters (predictors) using regression models. Because heavy metal concentrations were only analyzed in Baie des Sept Îles, we considered two models with different predictors and spatial ranges: Model 1 with organic matter and grain-size classes (gravel, sand, silt, clay) at all sampled stations; and Model 2 with heavy metal concentrations (arsenic, cadmium, chromium, copper, iron, manganese, mercury, lead, zinc) at stations in Baie des Sept Îles. We studied potential links between community characteristics (taxa richness, total density of individuals, Shannon diversity, Pielou evenness) and both sets of predictors using multiple linear regressions. Variables were transformed (logarithm or square root) if the assumptions of normality and homoscedasticity were not respected, and multicollinearity was assessed by the Variance Inflation Factor (Quinn and Keough, 2002). Outlier stations were determined using Cook's Distance and removed from analyses (Cook, 1977; Legendre and Legendre, 1998), while correlations between predictors was assessed by the Spearman rank coefficient: when two variables were strongly correlated ($|\rho| > 0.8$), one was removed from subsequent analyses although both were considered in interpretations (Quinn and Keough, 2002). Significant predictors were identified with a best fit model procedure with package MASS (stepwise, with forward and backward selection of predictors), using the Akaike Information Criterion as the decision metric (Quinn and Keough, 2002; Venables and Ripley, 2002). Finally, we explored relationships between taxa assemblages and both sets of predictors through non-parametric multivariate regression with distance-based linear modelling (DistLM; 9999 permutations) (McArdle and Anderson, 2001). Outputs of the clustering were combined with the DistLM results in a distance-based redundancy analysis (dbRDA), a constrained ordination method, to further identify relationships between clusters and habitat parameters (McArdle and Anderson, 2001).

1.5 Results and discussion

1.5.1 Description of the ecosystems

1.5.1.1 Sediment parameters

Maps with the parameters values for each sampled station are presented in Figures 10 and 11. Organic matter concentration in the sediment ranged between 0.17% and 3.87%, except for two stations reaching 4.41% and 8.26%, and most sampled stations presented values inferior to 2%. The highest organic matter values were observed in Baie des Sept Îles, directly in front of the city of Sept-Îles and the industrial facilities of Pointe-Noire, and in the western section of the bay close to *Zostera marina* meadows.

Overall, sediment was mainly composed of a high sand content (average and standard error of 52.7% and 2.3%, respectively) mixed with silt (27.3% and 1.8%). Gravel content was very low at all stations (3.5% and 0.7%), as was clay content (16.5% and 2.2%) except for 35 shallow stations in the Baie des Sept Îles where a dominance by clay was evident (more than 80%).

Most of the heavy metal concentrations for stations in Baie des Sept Îles were below the lowest toxicity criterion (Rare Effect Level) established by Environment Canada and Ministère du Développement Durable de l'environnement et des Parcs du Québec (2007) and none reached levels corresponding to the highest toxicity criteria (Probable or Frequent Effect Levels). Arsenic, chromium, copper, mercury, and zinc concentrations reached low-to-moderate toxicity levels at 40, 167, 84, eight, and 49 stations out of 175, respectively. Interestingly, nearly all sampled stations present a moderate toxicity to chromium, but St-Louis et al. (2018) suggested that higher concentrations could be related to the presence of post-glacial clays as a source of heavy metals, thus contributing to the natural background sediment concentration of these metals. Heavy metal concentrations reported in our study area are lower than surrounding basal concentrations in the Gulf of St. Lawrence (Lee et al., 1999), except for certain sites within Baie des Sept Îles. A possible explanation may be a dilution effect due to tidal and hydrographic currents, but a circulation model is needed to confirm this hypothesis (Shaw, 2019). The highest values for arsenic, chromium, copper, iron, manganese, lead, and zinc (corresponding to moderate toxicity levels according to the classification), are located in shallow areas close to Sept-Îles industrial harbors and the Pointe-Noire sector, which may be a possible source of metal enrichment in the sediment as ore transformation industries operate in the area.

1.5.1.2 Benthic community

The complete list of sampled taxa can be found in Table 5. A total of 289 taxa were identified, with individuals from 14 phyla where annelids, arthropods, and mollusks were dominant (Table 5). We compared this dataset with available inventories of benthic invertebrates, and we obtained a 70% match with the Ocean Biodiversity Information System (OBIS) online database (OBIS, 2020) and a 96% match with the dedicated catalogue of the Gulf of St. Lawrence from Brunel et al. (1998). Most observed taxa were new mentions for the Sept-Îles region.

Ten taxa were not documented in the reference datasets of Brunel et al. (1998) and OBIS (2020): *Bathyporeia quoddyensis*, *Cyclaspis varians*, *Glycera alba*, *Microphthalmus sczelkowii*, *Kirkegaardia* sp., *Mya pseudoarenaria*, *Pholoe minuta tecta*, *Phylo ornatus*, *Thyasira gouldi*, *Tricellaria arctica*; though *C. varians*, *M. sczelkowii*, *M. pseudoarenaria*, *P. ornatus*, and *T. gouldi* are registered as being present in the region by the World Register of Marine Species (WoRMS Editorial Board, 2020).

Specific diversity studies for coastal and shallow waters in the Gulf of St. Lawrence are scarce, in particular for the Sept-Îles region. Because historical surveys have mainly focused on commercially important species birds or cetaceans, further taxonomical groundwork is needed before robust interpretations of these distributions can be made. Our study does not report the presence of known exotic species in the area (Simard et al., 2013), however sampling campaigns considering other macrofaunal components, such as rocky substrate communities or fouling invertebrates, would be a valuable addition to complement this portrait.

Comparisons of biodiversity inventories obtained for each size class found 137 taxa in common, while 114 were found only in the 0.5 mm size class (for a total of 251 taxa) and 38 only in the 1 mm size class (total of 175 taxa). This difference highlights the advantage of using smaller mesh sizes to survey macrofaunal diversity, as small individuals, larvae, and some juvenile stages cannot be properly sampled with larger sieves and will result in a smaller biodiversity inventory (Gage et al., 2002). When we estimated the total taxa richness with the Chao2 index, we sampled 63% of the estimated taxa pool (Chao2 = 397) for the 0.5 mm size class and 75% (Chao2 = 232) for the 1 mm size class. Thus, the sampling campaigns were not able to survey the entire biodiversity of the region, where between 58 and 119 taxa may theoretically still be discovered or reported. The rarefaction curve for each size class did not reach an asymptote, further reinforcing the need to increase sampling effort to accurately describe the regional benthic biodiversity (Figure 6).

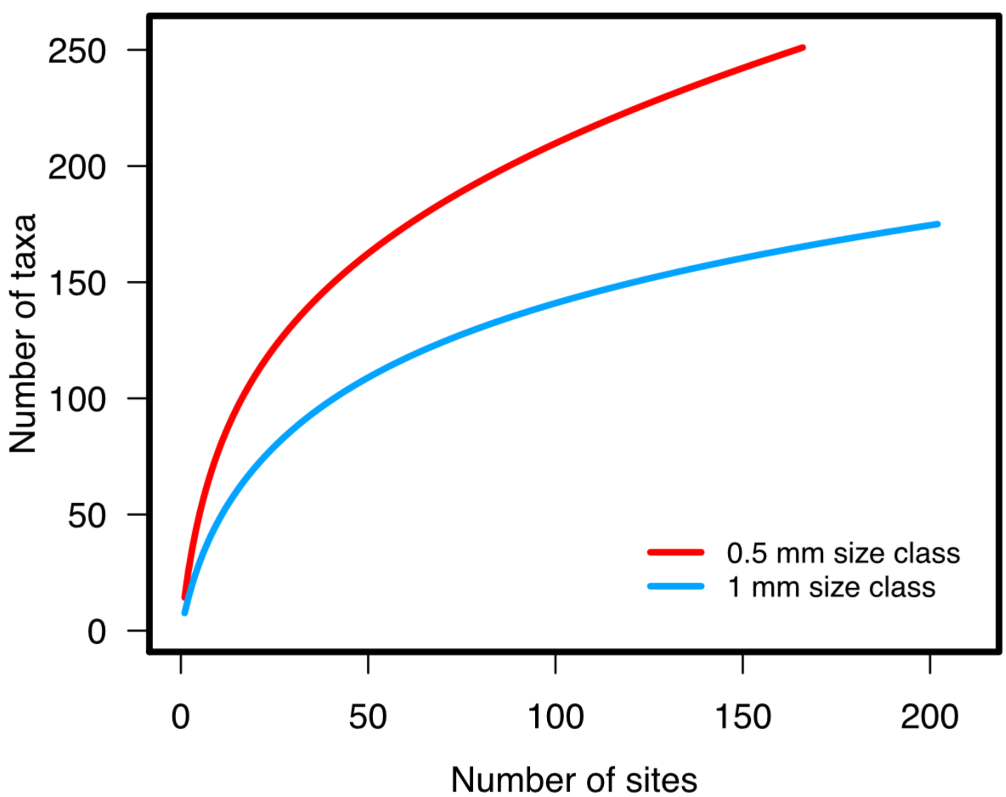

Figure 6 – Rarefaction curves for taxa assemblages of the 0.5 mm and the 1 mm size classes.

The highest values of taxa richness and total density were found in the Baie des Sept Îles sector, in particular in front of the city of Sept-Îles and in the southern section of the bay close to Pointe-Noire (Figure 12). Densities as high as 2000 individuals per grab were found at these stations for the 0.5 mm size class, consisting of mostly the polychaete *Micronephthys neotena*. There was a decrease in Shannon diversity, Pielou evenness, and taxonomic distinctness relative to other stations for both size classes. These patterns may indicate that these communities experience local structuring factors, such as ecosystem perturbation (Pearson and Rosenberg, 1978), which is further reinforced by the differences in habitat parameters reported in the previous section.

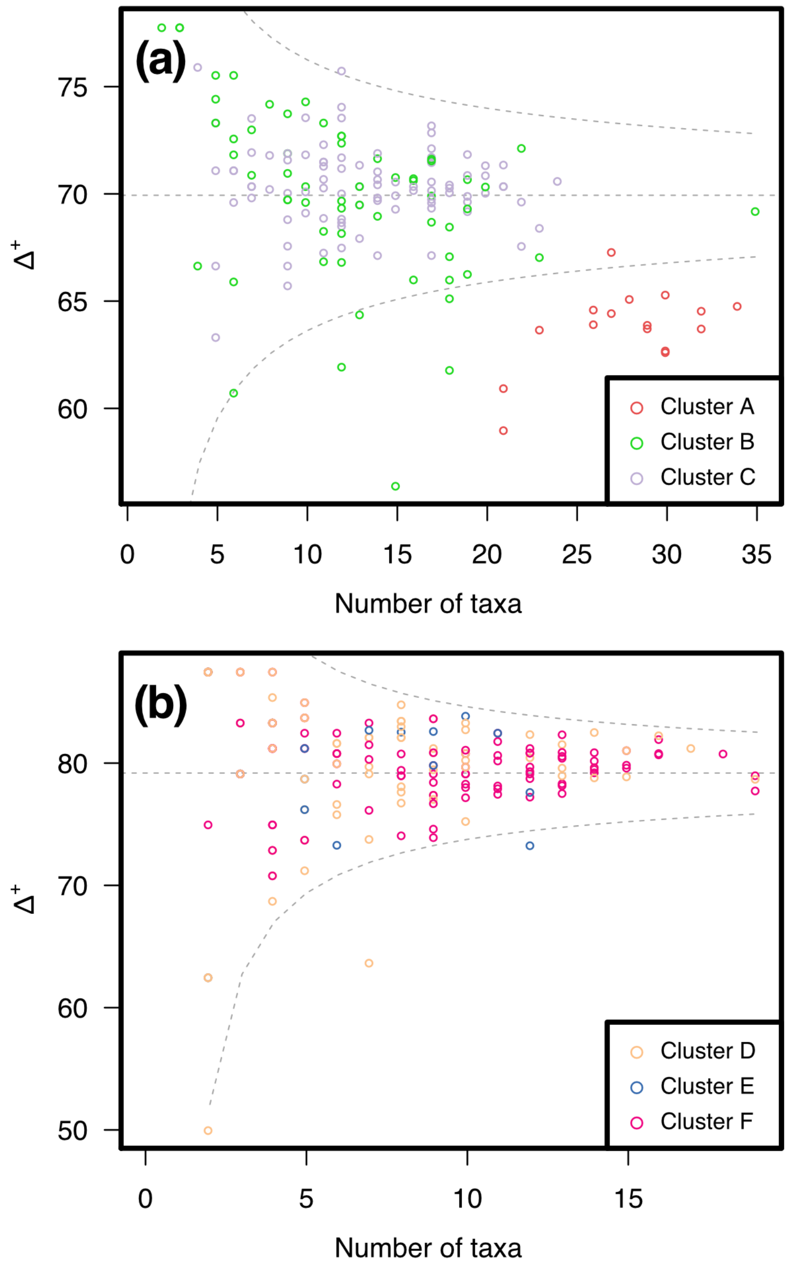

Figure 7 – Scatterplot of the taxonomic distinctness (Δ+) as a number of taxa sampled at a station. The dashed grey line corresponds to the expected value of Δ+ and the dashed grey curves are the upper and lower limits of the standard deviation interval calculated with the value of Δ+ at each station. Point colour represent similarity groups defined by Hierarchical Agglomerative Clustering. (a) Scatterplot for the 0.5 mm size class; (b) Scatterplot for the 1 mm size class.

A comparison between mean values of community characteristics obtained for each size class is presented in Table 1. The major differences were observed for taxa richness (nearly twice as high for the 0.5 mm size class) and for total density (nearly six times higher). As expected, more individuals were retained by the 0.5 mm sieving mesh. The average Shannon index was slightly higher for the 0.5 mm size class, while average Pielou evenness was around 0.7 for both classes. Concerning taxonomic distinctness, the majority of stations of both size classes were within the confidence interval around the expected average (calculated by the standard deviation of the index in relation to taxa richness), except for some 0.5 mm size class stations which were characterized by a low taxonomic distinctness relative to the rest of the sampled stations (Figure 7).

Table 1 – Mean values (and standard error) of community characteristics calculated with stations included in the 0.5 mm and the 1 mm size classes.

Variable	**Unit**	**0.5 mm size class**	**1 mm size class**
Taxa richness	taxa	14.27 (0.53)	7.58 (0.31)
Density of individuals	ind.grab^{-1}	225.91 (32.94)	39.62 (4.3)
Shannon diversity	NA	1.79 (0.04)	1.42 (0.05)
Pielou evenness	NA	0.71 (0.01)	0.75 (0.02)
Taxonomic distinctness	NA	69.14 (0.5)	76.49 (1.22)

Relationships between community characteristics and habitat parameters are described in Table 2. For both size classes, predictive power of the regressions varied between 0.02 and 0.5 and models considering organic matter and grain-size classes as predictors presented higher R^2_{adj} than did those considering heavy metal concentrations. Depth had a positive influence on nearly all community characteristics except total density, which is coherent with general patterns of coastal marine biodiversity (Gray and Elliott, 2009; Levinton, 2013; Piacenza et al., 2015). Three groups of variables were strongly correlated: organic matter/silt, chromium/iron/manganese, and copper/lead/zinc (thus considered together in interpretations).

For the 0.5 mm size class, most of the predictors selected by the best fit model procedure for Models 1 and 2 had a positive influence on community characteristics, except for arsenic, cadmium, and mercury in Model 2 where coefficients were negative (Table 2). Regressions for the 1 mm size class resulted in fewer predictors selected, with a positive influence of copper/lead/zinc in Model 2 and a negative influence of sand and clay in Model 1 and cadmium in Model 2 (Table 2). When comparing the outcomes of Model 1 for each size class, gravel, sand, and clay contents had opposite effects. For Model 2, few heavy metals influenced the 1 mm size class compared to the 0.5 mm size class, where only taxa richness and Shannon diversity were significantly related to predictors (with a very low predictive power). These results may indicate an increased influence of metals on smaller organisms, with an increased vulnerability except for copper/lead/zinc where the influence was positive. Concentration of the latter metals in surface sediments may be closely correlated to sediment texture (Burton et al., 2005), suggesting that this positive influence may also arise from a sediment composition favorable to both copper/lead/zinc concentrations and benthic communities. Ellis et al. (2017) reported significant relationships between heavy metal loading and traits such as body size, but further research is needed to understand responses of different macrofaunal components. Concerning the whole benthic community, the DistLM procedures selected all available predictors in Models 1 and 2 for the 0.5 mm size class, with $R^2 = 0.27$ and 0.18, respectively, while for the 1 mm size class depth, organic matter/silt, sand, clay were selected in Model 1 ($R^2 = 0.14$) and cadmium, chromium/iron/manganese, copper/lead/zinc in Model 2 ($R^2 = 0.07$).

Regressions for the 1 mm size class had less predictive power than did those for the 0.5 mm size class. Organism size may affect these regressions, as sieve size employed has been shown to influence the calculation of community characteristics (see previous sections and *e.g.*, Mckindsey and Bourget, 2001; Gage et al., 2002). Another possible influence could be the sampling strategy, as the 0.5 mm size class considered stations in two sectors while stations sampled for the 1 mm size class were located in four sectors. These results offer valuable insights for the understanding of benthic ecosystems, especially in order to predict their evolution when considering forcing factors, such as climate change or anthropogenic development, on habitat parameters. Furthermore, comparison of models for the 0.5 mm and 1 mm size classes allow to show how sampling strategies (*e.g.*, sieving mesh size, spatial range considered) influence community descriptions, which provides methodological recommendations for future environmental assessments in this region.

Table 2 – Predictor coefficients (and standard error) from the multiple linear regression models of community characteristics obtained for the 0.5 mm and the 1 mm size classes. Model 1 corresponds to organic matter and grain size classes as predictors for stations in every sector, and Model 2 corresponds to heavy metal concentrations as predictors for stations in Baie des Sept Îles only. OM = organic matter, As = arsenic, Cd = cadmium, Cr = chromium, Cu = copper, Fe = iron, Mn = manganese, Hg = mercury, Pb = lead, Zn = zinc, n = number of stations considered, "—" = predictors excluded by the best fit model selection. Significant p-values of marginal tests on predictors are highlighted in bold.

	Model 1						
	Intercept	**Depth**	**OM/silt**	**Gravel**	**Sand**	**Clay**	$\mathbf{R_{adj}^2}$
0.5 mm size class (n = 159)							
Specific richness	0.02 (0.06)	0.24 (0.07)	0.29 (0.08)	0.2 (0.09)	0.25 (0.12)	0.75 (0.11)	0.33
p-value =	0.7095	**0.0011**	**0.0005**	**0.0267**	**0.0305**	**<0.0001**	
Density of individuals	0.02 (0.06)	- 0.09 (0.07)	0.54 (0.07)	0.12 (0.08)	0.52 (0.11)	0.89 (0.1)	0.5
p-value =	0.7992	0.1524	**<0.0001**	0.137	**<0.0001**	**<0.0001**	
Shannon diversity	0.06 (0.06)	0.54 (0.07)	—	0.16 (0.09)	—	0.29 (0.07)	0.27
p-value =	0.3786	**<0.0001**		0.0671		<0.0001	
Pielou evenness	0.05 (0.07)	0.31 (0.08)	- 0.14 (0.08)	—	- 0.2 (0.12)	- 0.22 (0.11)	0.16
p-value =	0.4181	**<0.0001**	0.0713		0.0825	0.0523	
1 mm size class (n = 195)							
Specific richness	- 0.03 (0.06)	0.27 (0.07)	—	—	- 0.39 (0.08)	- 0.46 (0.11)	0.25
p-value =	*0.5874*	***<0.0001***			***<0.0001***	***<0.0001***	
Density of individuals	- 0.02 (0.07)	- 0.17 (0.08)	—	—	- 0.2 (0.09)	- 0.31 (0.13)	0.03
p-value =	*0.7931*	***0.0234***			***0.0272***	***0.0217***	
Shannon diversity	- 0.03 (0.06)	0.43 (0.06)	—	- 0.12 (0.08)	- 0.31 (0.07)	- 0.32 (0.11)	0.34
p-value =	*0.6265*	***<0.0001***		*0.1292*	***<0.0001***	***0.0029***	
Pielou evenness	0.02 (0.07)	0.31 (0.07)	—	—	—	—	0.1
p-value =	*0.8276*	***<0.0001***					

(Table 2 continued)

	Model 2						
	Intercept	**As**	**Cd**	**Cr/Fr/Mn**	**Hg**	**Cu/Pb/Zn**	$\mathbf{R^2_{adj}}$
0.5 mm size class (n = 142)							
Specific richness	- 0.06 (0.08)	- 0.41 (0.13)	- 0.69 (0.15)	—	- 0.46 (0.16)	1.03 (0.19)	0.18
p-value =	0.4276	**0.003**	**<0.0001**		**0.0051**	**<0.0001**	
Density of individuals	- 0.11 (0.06)	- 0.51 (0.1)	0.74 (0.13)	0.27 (0.11)	- 0.73 (0.13)	1.43 (0.16)	0.49
p-value =	0.0641	**<0.0001**	**<0.0001**	**0.013**	**<0.0001**	**<0.0001**	
Shannon diversity	0.04 (0.08)	—	—	- 0.28 (0.08)	—	—	0.07
p-value =	0.6124			**0.0007**			
Pielou evenness	0.06 (0.08)	—	—	- 0.17 (0.08)	—	—	0.02
p-value =	0.4713			**0.0345**			
1 mm size class (n = 126)							
Specific richness	0.06 (0.08)	—	- 0.51 (0.15)	—	—	0.31 (0.15)	0.08
p-value =	0.4529		**0.0009**			**0.0375**	
Density of individuals	0.02 (0.09)	—	—	—	—	—	0
p-value =	0.8373						
Shannon diversity	0.09 (0.08)	—	- 0.43 (0.15)	—	—	0.29 (0.14)	0.05
p-value =	0.2492		**0.0038**			**0.0437**	
Pielou evenness	0.06 (0.08)	—	—	—	—	—	0
p-value =	0.4249						

1.5.2 Similarity between taxa assemblages

1.5.2.1 0.5 mm size class

Hierarchical agglomerative clustering for the 0.5 mm size class identified three clusters of similar stations (Figure 8a). Mean Bray-Curtis dissimilarity was 0.38, 0.85, and 0.69 for clusters A, B, and C, respectively, indicating a higher variability for communities of stations within clusters B and C compared to cluster A (Table 3). Mean within-cluster taxonomic distinctness varies between cluster A (63.8%) and clusters B and C (69.9% and 70.4%, respectively). As shown in Figure 7a, stations from cluster A were well discriminated from those of clusters B and C, because of a lower taxonomic distinctness outside the confidence interval.

Table 3 – Bray-Curtis dissimilarity of the clusters obtained for the 0.5 mm and the 1 mm size classes. The diagonal of the triangular matrix corresponds to within-cluster dissimilarity, and other cells are across-cluster dissimilarity.

0.5 mm size class			
	A	*B*	*C*
A	37.68		
B	92.53	84.85	
C	90.37	91.67	69.35
1 mm size class			
	D	*E*	*F*
D	92.57		
E	92.16	69.15	
F	94.82	97.87	75.43

Stations of cluster A were located exclusively in Baie des Sept Îles, in front of the city of Sept-Îles and the industrial operations of Pointe-Noire (Figure 8a). The benthic community of stations in cluster A was dominated by annelids, representing 89.5% of the sampled individuals, with some phoronids (5%). *Micronephthys neotena*, *Nephtys* sp., *Prionospio steenstrupi*, *Scoloplos armiger*, and *Phoronida* accounted for 51% of the total contribution to cluster similarity, while 50 taxa were selected by the IndVal index as characteristic of the community based on their contribution to the cluster similarity. These stations were shallow, at 4 m deep on average, with a high organic matter content and a sediment mostly composed of clay, and average heavy metal concentrations were the highest observed in our sampling (Table 4). Several characteristic taxa can be linked to human perturbation: *M. neotena* has been found in high density at decommissioned dump sites within Baie-des-Chaleurs (eastern Canada), suggesting an opportunistic behavior toward perturbation (Noyes, 1980; Pocklington, 1989), while *S. armiger* and *P. steenstrupi* are ranked 'tolerant to disturbance' and 'second-order opportunistic', respectively, in the organic matter enrichment classification of Borja et al. (2000).

Cluster B regrouped some stations in Baie des Sept Îles (close to the coast and in the archipelago) and nearly all of those at Manitou River (Figure 8a). Taxa assemblage was characterized by a combination of arthropods (32.7% of the sampled individuals), annelids (22.3%), and mollusks (20.9%), with a presence of nematodes (14.6%) and echinoderms (8%). Of the total contribution to community similarity, 67% was explained by *Echinarachnius parma*, Nematoda, *Phoxocephalus holbolli*, Harpacticoida, and *Spisula solidissima*, with 12 significant taxa for this cluster according to IndVal scores. Mean station depth was 14 m, with low organic matter, a high content of sand and clay; stations of this cluster located in Baie des Sept Îles had low concentrations of heavy metal in the sediment (Table 4). This cluster regroups taxa that are sensitive to perturbation, such as *P. holbolli* and *S. solidissima*, both of which have been classified as being 'very sensitive to disturbance' (Borja et al., 2000). Furthermore, Nelson et al. (1988) described physiological impacts of heavy metal loading for *S. solidissima*, especially for increased copper concentrations.

For cluster C, stations were widely distributed in Baie des Sept Îles (except for one station at Manitou River; Figure 8a). Arthropods (42.3%) and annelids (37.8%) were the main phyla of this cluster, with a further large proportion (14.2%) represented by mollusks. Fifteen taxa were selected by the IndVal scores, with characteristic taxa being *M. neotena*, *Macoma calcarea*, *Eudorellopsis integra*, *Protomedeia grandimana*, and *Leucon (Leucon) nasicoides* (64% of the total similarity). Station depth was high, 33 m deep on average, and the sediment had a sandy-silty profile with moderate concentrations of organic matter and heavy metal compared to the other clusters (Table 4). Here, taxa do not present a particular relationship to perturbation, with many representative taxa being 'indifferent to disturbance', including *M. calcarea*, *P. grandimana*, and *L. (Leucon) nasicoides*, with the exception of *M. neotena* as described above (Borja et al., 2000).

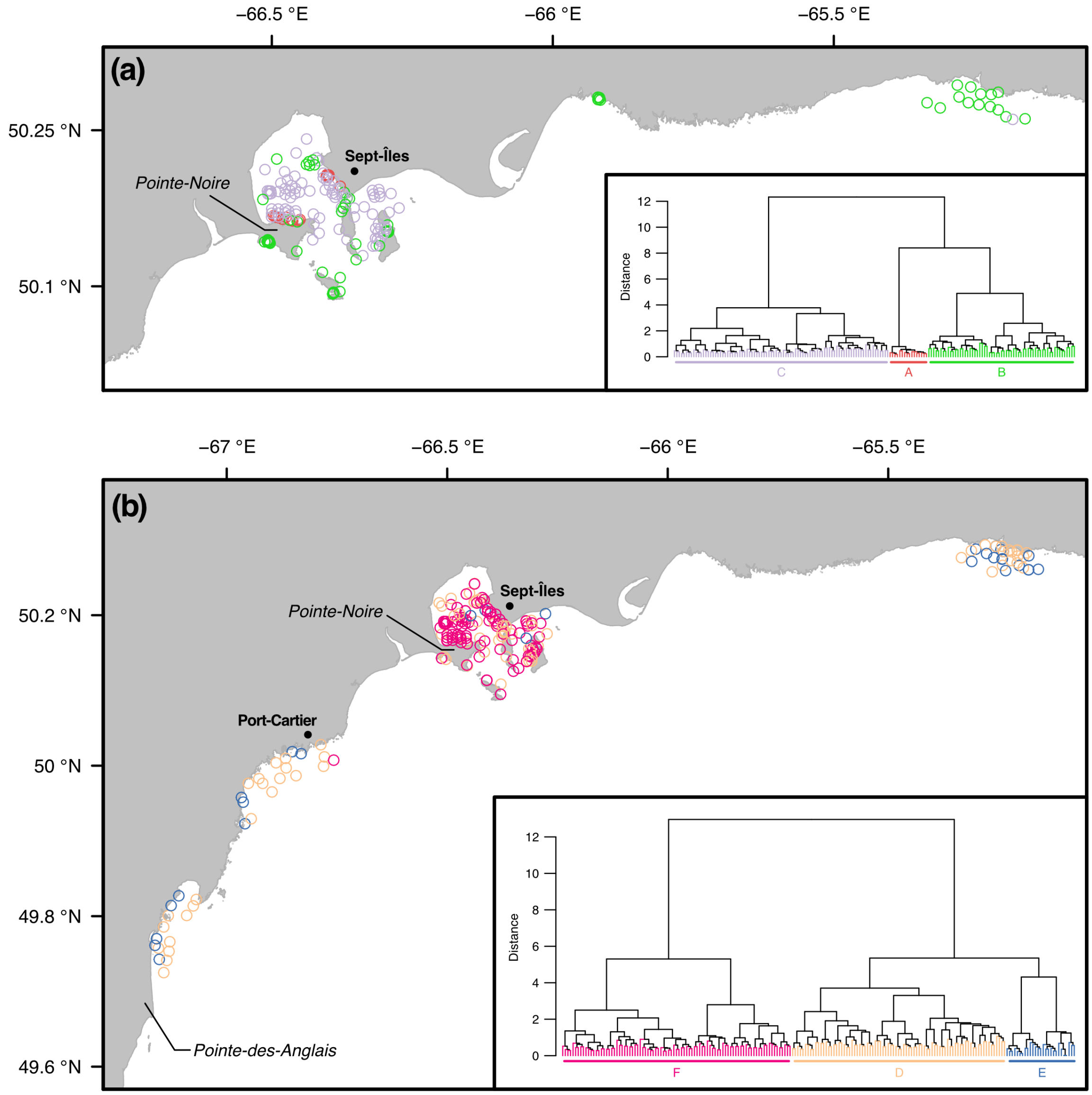

Figure 8 – Hierarchical agglomerative clustering of taxa assemblages. The dendrogram presents relationships between stations, displayed on the map with one color for each group. (a) Clusters for the 0.5 mm size class; (b) Clusters for the 1 mm size class.

Table 4 – Mean values (and standard error) of habitat parameters calculated with stations belonging in the groups defined by the hierarchical agglomerative clustering for the 0.5 mm and the 1 mm size classes. Only Baie des Sept Îles stations (when available in the cluster) were considered for the calculation of heavy metal concentration averages. n = number of stations considered for the calculation.

		0.5 mm size class			**1 mm size class**		
Variable	**Unit**	**Cluster A**	**Cluster B**	**Cluster C**	**Cluster D**	**Cluster E**	**Cluster F**
All four sectors		(n = 16)	(n = 61)	(n = 89)	(n = 83)	(n = 27)	(n = 89)
Depth	m	6.99 (0.35)	13.72 (1.64)	32.76 (2.02)	21.5 (1.88)	16.1 (2.53)	32.52 (2.04)
Organic matter	%	2.58 (0.42)	0.52 (0.04)	1.76 (0.1)	0.72 (0.08)	0.35 (0.03)	1.77 (0.1)
Gravel	%	0 (0)	5.9 (1.8)	1.7 (1)	6.8 (1.6)	2.2 (0.1)	1.7 (1)
Sand	%	0 (0)	53.3 (5.3)	46.3 (2.3)	63.8 (3.7)	92.4 (1.8)	47 (2.5)
Silt	%	0.1 (0)	11.3 (2.4)	49.1 (2.2)	25 (3)	4.7 (1.2)	48.4 (2.5)
Clay	%	99.9 (0)	29.5 (5.6)	3 (1.6)	4.4 (1.2)	0.6 (0.01)	2.9 (1.5)
Baie des Sept Îles only		(n = 16)	(n = 47)	(n = 88)	(n = 42)	(n = 3)	(n = 89)
Arsenic	$mg.kg^{-1}$	3.74 (0.32)	2.51 (0.19)	3.89 (0.24)	3.56 (0.55)	1.83 (0.52)	3.96 (0.25)
Cadmium	$mg.kg^{-1}$	0.15 (0.01)	0.11 (0.01)	0.14 (0.01)	0.13 (0.01)	0.1 (0.01)	0.14 (0.01)
Chromium	$mg.kg^{-1}$	80.29 (5.34)	54.5 (3.68)	58.07 (1.71)	51.77 (3.5)	32.73 (5.27)	58.19 (1.66)
Copper	$mg.kg^{-1}$	19.89 (1.33)	6.86 (0.73)	12.33 (0.51)	9.3 (1.05)	6.37 (1.39)	12.26 (0.5)
Iron	$g.kg^{-1}$	64.73 (3.73)	57.56 (4.42)	55.27 (1.68)	49.2 (2.47)	33.56 (4.57)	54.96 (1.69)
Manganese	$g.kg^{-1}$	2.17 (0.32)	0.93 (0.06)	1.18 (0.06)	0.86 (0.06)	0.57 (0.09)	1.17 (62.25)
Mercury	$mg.kg^{-1}$	0.04 (0.02)	0.01 (0.01)	0.02 (0.01)	0.02 (0.01)	0.01 (0.01)	0.03 (0.01)
Lead	$mg.kg^{-1}$	7.32 (0.58)	3.26 (0.27)	5.46 (0.2)	4.65 (0.4)	2.63 (0.37)	5.59 (0.2)
Zinc	$mg.kg^{-1}$	77.21 (4.22)	43.54 (2.25)	61.09 (1.77)	54.28 (4.09)	39.77 (3.11)	61.61 (1.68)

The comparison of characteristic taxa in clusters shows cluster A to be quite different from the other clusters. This difference is visible on the two dbRDAs, where there is an evident discrimination of cluster A's assemblages relative to those of clusters B and C, mainly explained by depth and clay for Model 1 and chromium/iron/manganese concentrations for Model 2 (Figure 9a,b). Bray-Curtis dissimilarities indicate a higher similarity within this cluster, which may be explained by uniformization due to some perturbation (Clarke and Warwick, 2001; Séguin et al., 2014). Furthermore, cluster A's stations are close to human activities, a possible source for higher contents of organic matter, heavy metal, and clay contents relative to other stations in Baie des Sept Îles. As suggested by Pearson and Rosenberg (1978), those specific sites may present an ecosystem disturbance due to an increased organic matter. Local hydrodynamics are likely one of the main factors influencing sediment composition (Shaw, 2019). We may also postulate that dredging activities, where a dumping site is operated close to cluster A's stations, may impact on the composition of benthic sediments by favoring the accumulation of fine particles (Cooper et al., 2011). Contaminants in sediments, such as heavy metals, are known to impact marine species at the individual level, for example by affecting their metabolism and their reproductive success, which extends their impacts to the distribution of community traits (Johnston and Roberts, 2009; Chiarelli and Roccheri, 2014; Desrosiers et al., 2018). The link between higher contaminant concentrations and human perturbation is well established (*e.g.*, Schafer et al., 1990; Bolton et al., 2004). However, the presence of post-glacial clay can temper perceived effects of heavy metal inputs from anthropogenic sources on benthic communities (St-Louis et al., 2018), even though specific sites close to harbor installations seem to present overall higher concentrations than those in the rest of Baie des Sept Îles.

These results strongly suggest that cluster A regroups stations with a higher perturbation status than those of cluster B, while cluster C possesses an intermediate profile, with a possible relationship with organic matter content and heavy metal concentrations. Consequently, this size class allowed to detect a certain perturbation signal on benthic communities, located in specific areas.

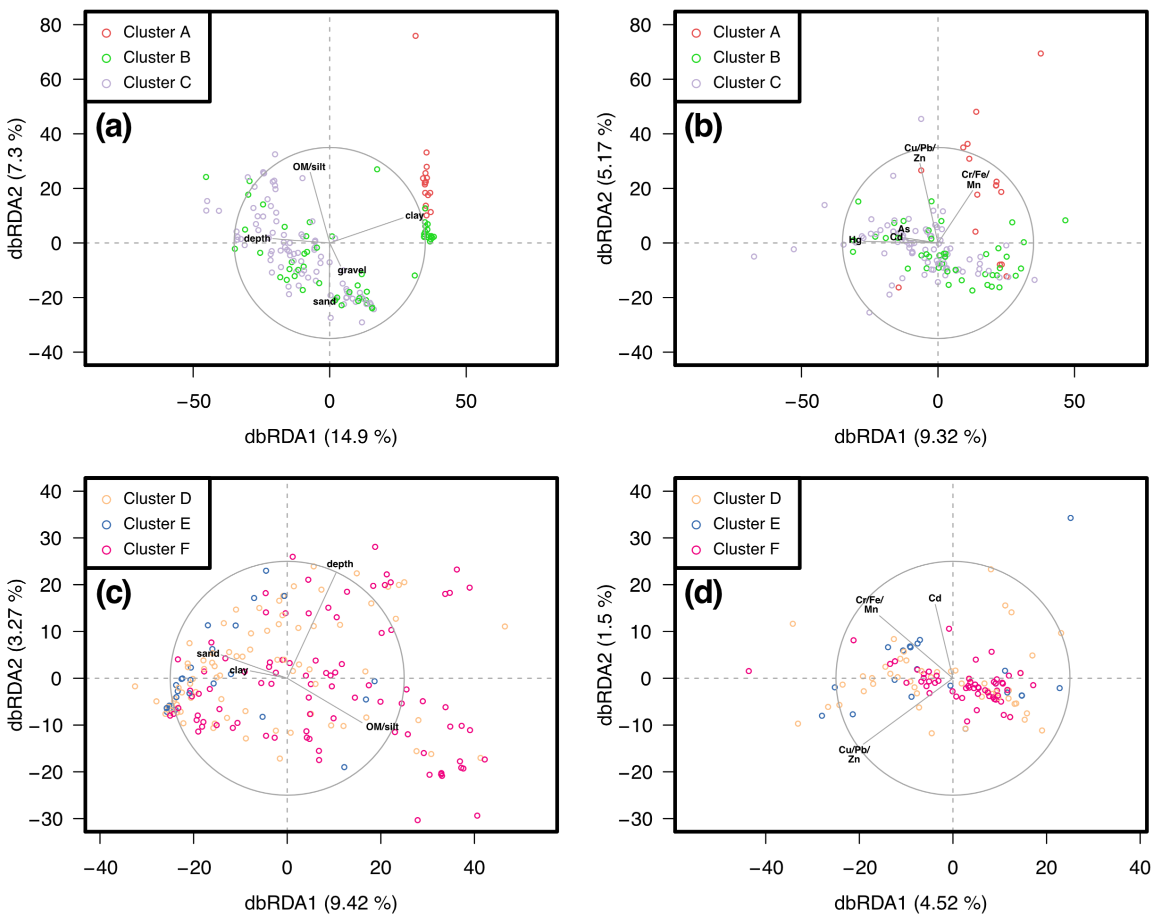

Figure 9 – Constrained ordination with a distance-based redundancy analysis for taxa assemblages obtained for the 0.5 mm and 1 mm size classes. Only predictors selected by the distance-based linear modelling (DistLM) are displayed. Point color represents similarity groups defined by the hierarchical agglomerative clustering. (a) Ordination for the 0.5 mm size class with organic matter and grain-size classes as predictors (Model 1); (b) Ordination for the 0.5 mm size class with heavy metal concentrations as predictors (Model 2); (c) Ordination for the 1 mm size class with organic matter and grain-size classes as predictors (Model 1); (d) Ordination for the 1 mm size class with heavy metal concentrations as predictors (Model 2). OM = organic matter, As = arsenic, Cd = cadmium, Cr = chromium, Cu = copper, Fe = iron, Mn = manganese, Hg = mercury, Pb = lead, Zn = zinc.

1.5.2.2 1 mm size class

Three groups of stations were also identified by hierarchical agglomerative clustering for the 1 mm size class (Figure 8b). Mean Bray-Curtis dissimilarity was highest for cluster D (0.93), intermediate for cluster F (0.81), and lowest for cluster E (0.78; Table 3). No particular relationship could be identified based on mean taxonomic distinctness, even though the average for cluster E's (71.1%) was slightly less than those for clusters D and F (77.6% and 79.7%, respectively; Figure 7b).

Cluster D regrouped stations from all four sampled sectors, in particular most of the stations on the coast of Port-Cartier and the Pentecôte and Manitou rivers were included here (71%, 67%, and 58% of the stations, respectively; Figure 8b). Arthropods (39.4% of the sampled individuals), mollusks (20.2%), annelids (18.2%), and echinoderms (17%) were the phyla dominating the community. Characteristic taxa were *Cistenides granulata*, *E. parma*, *P. holbolli*, *Nephtys caeca*, and *Strongylocentrotus* sp. (62% of the total similarity), and 13 taxa had significant IndVal scores. Stations were quite shallow (21.5 m deep) on average, with a low organic matter content and a sediment composed of sand with some silt (Table 4). Interestingly, while many taxa are classified as 'indifferent to disturbance', this is not the case for *P. holbolli* and *Strongylocentrotus* sp. which are 'very sensitive', indicating mixed responses of taxa in this cluster's taxa (Borja et al., 2000).

Concerning cluster E, nearly all stations were located outside of Baie des Sept Îles (Figure 8b). Its assemblage was dominated by echinoderms (48%) and mollusks (40.6%), with *E. parma* and *Mesodesma arctatum* accounting for 93% of the total similarity and were two of the three taxa selected by the IndVal calculation. Compared to the other clusters, stations were shallower (mean depth of 16.1 m), had less organic matter, and the sediment was mostly sandy (Table 4). Three stations in Baie des Sept Îles had low heavy metal concentrations, but these values are not informative compared to the other clusters because of the very low number of stations available to calculate averages. *E. parma* and *M. arctatum* are not referenced in the classification of Borja et al., but many echinoids and some species of the genus *Mesodesma* are known to be 'very sensitive to disturbance' (Borja et al., 2000), which is an evidence this cluster may be an evidence of a low perturbation profile.

Finally, cluster F regrouped only stations in Baie des Sept Îles (Figure 8b). Of the sampled individuals, 41.2% were arthropods, 35.5% annelids, 21.4% mollusks, and 26 taxa were selected by their IndVal scores. Characteristic taxa were *M. calcarea*, *E. integra*, *Ennucula tenuis*, *M. neotena*, and *Protomedeia grandimana* (66% of the total similarity). Stations were at 32.5 m deep on average, with an equal proportion of sand and silt, a high organic matter content, and high overall heavy metal concentrations (Table 4). This cluster is very similar to cluster C for the 0.5 mm size class, both in terms of habitat parameters values and characteristic taxa.

Stations of clusters D and F presented a wider dispersion on the dbRDAs than those of cluster E, where the latter were grouped and influenced by a high sand content and low organic matter/silt for Model 1, and heavy metal concentrations for Model 2 (Figure 9c-d). Depth, organic matter/silt, sand, clay (Model 1), and all heavy metals except arsenic and mercury (Model 2) were selected to explain the structure of the 1 mm size class assemblages, even though R^2 values were quite low (0.14 and 0.07; Figure 9c-d). Clusters D and E do not seem to be particularly perturbed, having a profile quite characteristic of other coastal ecosystems in the Gulf of St. Lawrence and presenting taxa that are sensitive to disturbance. On the other hand, cluster F seems to exhibit a somewhat different profile, where there is a higher sediment organic matter content. Apart from a possible human perturbation, another possible driver to explain the structure of the benthic communities is the sedimentary profile. Sediments in clusters D and E's stations are mainly sandy whereas those for cluster F are mostly sandy silty. The dominance of sand-dwelling species such as *M. arctatum* and *E. parma* in clusters D and E supports this hypothesis (Bourget, 1997).

These results show that clusters obtained for the 1 mm size class seem to be more variable than those observed for the 0.5 mm size class, as shown by higher Bray-Curtis dissimilarity values for the former. By sampling only individuals larger than 1 mm only, the assessment of benthic ecosystems seems to be insufficient to differentiate ecosystem perturbation from natural variability (Bachelet, 1990; Gage et al., 2002; Couto et al., 2010). This is of even greater importance when considering that the 0.5 mm size class can detect some ecosystem perturbation, as shown above. Further field campaigns, including factorial sampling with possibly impacted and reference areas, along with local experimental and manipulative work, would provide more robust conclusions on the 1 mm size class and link taxa responses to environmental perturbation.

1.6 Conclusions

Our work provides the first description of benthic habitats and communities in the sub-Arctic ecosystems of Sept-Îles, which is an important contribution to local and regional biodiversity surveys and a necessary step towards establishing baseline conditions for environmental assessments. Regression models between community characteristics (*e.g.*, taxa richness or diversity), taxon densities, and habitat parameters (*e.g.*, organic matter, heavy metals) highlight how abiotic parameters may impact benthic communities and provide predictive methods to assess the evolution of these communities in future environmental conditions. Finally, we detected taxa assemblages with similar distributions in the environment, which we related to the values of habitat parameters to highlight perturbation patterns.

By studying multiple size classes of macrofauna, we were able to compare how they will influence the description of benthic communities. When including smaller organisms (0.5 mm size class), conclusions were more robust and we were able to document local perturbation responses. The evaluation of only larger organisms (1 mm size class) was able to decrease the time needed to sample and identify taxa, but at the cost of reducing the strength of similarity analyses and of detecting relationships to habitat parameters variation and perturbation.

Our results provide valuable guidelines for the environmental monitoring of benthic ecosystems in industrial harbor areas, along with ecological data to better understand how sub-Arctic ecosystems react to environmental perturbation.

1.7 Acknowledgements

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1.8 Supplementary material

Table 5 – List of the taxa along with the number of stations where this taxon was present and the number of collected individuals, for the 0.5 mm and the 1 mm size classes.

			0.5 mm size class		1 mm size class	
Aphia ID	**Phylum**	**Accepted name**	**Presence**	**Density**	**Presence**	**Density**
102866	Arthropoda	*Aceroides (Aceroides) latipes*	22	57	14	27
136340	Arthropoda	*Akanthophoreus gracilis*	31	163	9	13
878476	Mollusca	*Ameritella agilis*	16	39	14	47
547000	Arthropoda	*Ameroculodes edwardsi*	8	10	8	10
101908	Arthropoda	*Ampelisca macrocephala*	—	—	1	2
158022	Arthropoda	*Ampelisca vadorum*	2	2	2	2
101364	Arthropoda	Ampeliscidae spp.	—	—	5	7
762338	Annelida	*Ampharete oculata*	—	—	1	8
129155	Annelida	*Ampharete* sp.	1	2	—	—
981	Annelida	Ampharetidae spp.	—	—	5	9
111186	Bryozoa	*Amphiblestrum auritum*	1	1	—	—
125064	Echinodermata	*Amphipholis squamata*	—	—	9	133
1135	Arthropoda	Amphipoda	44	127	6	20
123613	Echinodermata	*Amphiura* sp.	—	—	1	1
102002	Arthropoda	*Ampithoe rubricata*	—	—	2	3
882	Annelida	Annelida	13	89	—	—
102513	Arthropoda	*Anonyx lilljeborgi*	9	16	9	16
102516	Arthropoda	*Anonyx sarsi*	—	—	2	2
1292	Cnidaria	Anthozoa	3	4	4	5
101368	Arthropoda	Aoridae spp.	1	4	—	—
157214	Annelida	*Arcteobia anticostiensis*	1	1	1	1
333034	Annelida	*Aricidea acmira catherinae*	3	13	—	—
129430	Annelida	*Aricidea* sp.	7	57	—	—
531617	Mollusca	*Arrhoges occidentalis*	1	1	3	4
137683	Mollusca	*Astarte* sp.	8	12	7	10
156746	Mollusca	*Astarte subaequilatera*	—	—	3	12
156747	Mollusca	*Astarte undata*	—	—	3	12
123219	Echinodermata	*Asterias* sp.	1	1	1	1
141652	Mollusca	*Axinopsida orbiculata*	27	134	18	85
130275	Annelida	*Axiothella catenata*	2	5	2	5
106122	Arthropoda	*Balanus* sp.	1	2	—	—
102873	Arthropoda	*Bathymedon longimanus*	4	11	3	8
102875	Arthropoda	*Bathymedon obtusifrons*	2	15	2	15
158034	Arthropoda	*Bathyporeia quoddyensis*	—	—	2	2
105	Mollusca	Bivalvia	114	10486	51	483
386411	Mollusca	*Boreochiton ruber*	8	24	—	—
110397	Arthropoda	*Brachydiastylis* sp.	4	24	6	28
106673	Arthropoda	Brachyura	2	2	2	2
146142	Bryozoa	Bryozoa	3	9	—	—
416573	Arthropoda	*Byblis gaimardii*	1	1	—	—
110851	Bryozoa	*Callopora* sp.	5	9	5	9
1606	Cnidaria	Campanulariidae spp.	1	1	—	—
1607	Cnidaria	Campanulinidae spp.	35	35	—	—

Aphia ID	Phylum	Accepted name	0.5 mm size class		1 mm size class	
			Presence	Density	Presence	Density
158057	Arthropoda	*Cancer irroratus*	2	2	—	—
110734	Bryozoa	Candidae spp.	9	14	4	4
129211	Annelida	*Capitella* sp.	1	1	—	—
921	Annelida	Capitellidae spp.	20	529	—	—
101851	Arthropoda	*Caprella septentrionalis*	3	13	—	—
110873	Bryozoa	*Cellepora* sp.	9	370	6	275
111397	Bryozoa	*Celleporella hyalina*	1	1	—	—
2088	Mollusca	Chaetodermatida	1	1	—	—
2081	Chaetognatha	Chaetognatha	18	25	13	17
129242	Annelida	*Chaetozone* sp.	1	1	—	—
107315	Arthropoda	*Chionoecetes opilio*	1	4	—	—
140692	Mollusca	*Chlamys islandica*	2	2	2	2
129525	Annelida	*Chone* sp.	1	1	1	1
139000	Mollusca	*Ciliatocardium ciliatum*	1	12	1	12
919	Annelida	Cirratulidae spp.	1	1	6	17
1082	Arthropoda	Cirripedia	16	246	—	—
238377	Annelida	*Cistenides granulata*	1	3	1	3
157317	Annelida	*Cistenides hyperborea*	24	56	43	142
157316	Annelida	*Cistenides* sp.	2	3	—	—
157320	Annelida	*Clymenella zonalis*	1	1	—	—
129984	Annelida	*Cossura longocirrata*	—	—	1	2
237004	Arthropoda	*Crassicorophium bonellii*	1	6	1	6
140440	Mollusca	*Crenella decussata*	5	11	—	—
1059487	Bryozoa	*Cribrilina cryptooecium*	17	77	17	61
1066	Arthropoda	Crustacea	1	1	—	—
1137	Arthropoda	Cumacea	2	2	—	—
157810	Arthropoda	*Cyclaspis varians*	4	5	—	—
156832	Mollusca	*Cyclocardia borealis*	—	—	1	1
139474	Mollusca	*Cylichna alba*	1	5	3	7
140102	Mollusca	*Cyrtodaria siliqua*	3	16	—	—
111174	Bryozoa	*Dendrobeania murrayana*	3	3	2	2
157815	Arthropoda	*Diastylis polita*	1	1	—	—
110487	Arthropoda	*Diastylis rathkei*	—	—	1	1
157817	Arthropoda	*Diastylis sculpta*	10	46	7	41
110398	Arthropoda	*Diastylis* sp.	16	95	11	47
131121	Annelida	*Dipolydora quadrilobata*	1	1	1	1
129611	Annelida	*Dipolydora* sp.	1	1	—	—
971	Annelida	Dorvilleidae spp.	1	1	—	—
117888	Cnidaria	*Dynamena pumila*	1	14	—	—
158062	Echinodermata	*Echinarachnius parma*	—	—	1	1
123082	Echinodermata	Echinoidea	46	317	57	378
574096	Mollusca	*Ecrobia truncata*	3	7	—	—
157884	Arthropoda	*Edotia montosa*	5	56	2	19
157885	Arthropoda	*Edotia triloba*	3	5	4	6
111355	Bryozoa	*Electra pilosa*	6	13	—	—
506605	Mollusca	*Ennucula delphinodonta*	3	3	—	—
140584	Mollusca	*Ennucula tenuis*	1	1	—	—
1820	Hemichordata	*Enteropneusta*	53	229	58	211
130613	Annelida	*Eteone flava*	1	2	—	—
130616	Annelida	*Eteone longa*	—	—	1	7
129443	Annelida	*Eteone* sp.	2	12	—	—
157374	Annelida	*Eteone trilineata*	20	122	6	8
130903	Annelida	*Euchone analis*	—	—	1	1

Aphia ID	Phylum	Accepted name	0.5 mm size class Presence	0.5 mm size class Density	1 mm size class Presence	1 mm size class Density
129528	Annelida	*Euchone* sp.	9	243	—	—
111361	Bryozoa	*Eucratea loricata*	1	32	1	32
1600	Cnidaria	Eudendriidae spp.	12	12	—	—
110524	Arthropoda	*Eudorella emarginata*	4	4	—	—
157820	Arthropoda	*Eudorellopsis integra*	6	9	6	9
140536	Mollusca	*Euspira pallida*	66	1163	63	669
117690	Cnidaria	*Filellum serpens*	1	1	1	1
1207	Arthropoda	Gammaridea	1	1	—	—
101537	Arthropoda	*Gammarus* sp.	4	6	—	—
101	Mollusca	Gastropoda	—	—	1	1
130116	Annelida	*Glycera alba*	2	14	—	—
130118	Annelida	*Glycera capitata*	—	—	4	17
157392	Annelida	*Glycera dibranchiata*	2	2	3	3
129296	Annelida	*Glycera* sp.	8	25	5	7
130140	Annelida	*Goniada maculata*	11	40	7	13
953	Annelida	Goniadidae spp.	41	102	48	110
158095	Arthropoda	*Guernea (Prinassus) nordenskioldi*	—	—	2	4
1484	Arthropoda	Halacaridae spp.	2	19	—	—
100667	Cnidaria	Halcampidae spp.	6	44	1	1
1342053	Mollusca	*Haminoea solitaria*	1	1	—	—
158099	Arthropoda	*Hardametopa carinata*	1	1	1	1
130769	Annelida	*Harmothoe imbricata*	1	1	—	—
129491	Annelida	*Harmothoe* sp.	3	9	8	14
1102	Arthropoda	Harpacticoida	3	3	3	3
102974	Arthropoda	*Harpinia propinqua*	87	651	2	5
157436	Annelida	*Hartmania moorei*	—	—	2	12
152302	Annelida	*Hediste diversicolor*	2	3	—	—
128143	Arthropoda	*Hemicythere villosa*	26	96	16	36
104054	Brachiopoda	*Hemithiris psittacea*	6	45	—	—
946	Annelida	Hesionidae spp.	—	—	1	2
138749	Mollusca	*Heteranomia squamula*	1	2	—	—
140103	Mollusca	*Hiatella arctica*	1	4	1	4
123083	Echinodermata	Holothuroidea	7	23	9	25
107323	Arthropoda	*Hyas coarctatus*	4	6	4	6
1337	Cnidaria	Hydrozoa	—	—	1	1
157891	Arthropoda	*Idotea phosphorea*	1	1	—	—
101389	Arthropoda	Ischyroceridae spp.	3	13	3	11
102412	Arthropoda	*Ischyrocerus anguipes*	1	1	1	1
1131	Arthropoda	Isopoda	7	148	5	76
884676	Annelida	*Kirkegaardia* sp.	1	1	—	—
140170	Mollusca	*Lacuna vincta*	4	12	4	11
117136	Cnidaria	*Lafoea* sp.	1	1	—	—
110516	Arthropoda	*Lamprops fasciatus*	1	2	1	1
110517	Arthropoda	*Lamprops fuscatus*	14	26	14	26
1307579	Arthropoda	*Lamprops quadriplicata*	2	2	—	—
140187	Mollusca	*Lepeta caeca*	7	12	6	9
158112	Arthropoda	*Leptocheirus pinguis*	1	2	—	—
110501	Arthropoda	*Leptostylis ampullacea*	1	12	1	3
110618	Arthropoda	*Leucon (Leucon) nasica*	1	4	—	—
110619	Arthropoda	*Leucon (Leucon) nasicoides*	43	408	33	185
880017	Mollusca	*Limecola balthica*	16	212	13	38
140262	Mollusca	*Littorina littorea*	5	11	4	10
140263	Mollusca	*Littorina obtusata*	1	1	—	—

Aphia ID	Phylum	Accepted name	0.5 mm size class		1 mm size class	
			Presence	Density	Presence	Density
967	Annelida	Lumbrineridae spp.	5	18	3	14
101395	Arthropoda	Lysianassidae spp.	3	5	—	—
141580	Mollusca	*Macoma calcarea*	88	584	84	420
138531	Mollusca	*Macoma* sp.	2	5	—	—
158037	Arthropoda	*Maera danae*	1	1	1	1
130305	Annelida	*Maldane sarsi*	2	3	2	3
923	Annelida	Maldanidae spp.	31	203	29	197
141819	Mollusca	*Margarites costalis*	—	—	1	3
138592	Mollusca	*Margarites* sp.	1	1	—	—
111411	Bryozoa	*Membranipora membranacea*	2	2	—	—
156805	Mollusca	*Mesodesma arctatum*	4	89	16	217
130349	Annelida	*Micronephthys neotena*	2	7	—	—
130168	Annelida	*Microphthalmus aberrans*	6	63	—	—
130174	Annelida	*Microphthalmus sczelkowii*	4	35	—	—
129313	Annelida	*Microphthalmus* sp.	1	1	—	—
101694	Arthropoda	*Monoculodes* sp.	6	20	—	—
102901	Arthropoda	*Monoculopsis longicornis*	2	5	—	—
140472	Mollusca	*Muculus (Musculus) discors*	1	1	1	1
140430	Mollusca	*Mya arenaria*	1	4	11	17
156249	Mollusca	*Mya pseudoarenaria*	1	1	—	—
110949	Bryozoa	*Myriapora* sp.	1	1	—	—
876479	Mollusca	*Mysella planulata*	3	14	3	5
138228	Mollusca	*Mytilus* sp.	23	462	10	133
799	Nematoda	Nematoda	61	1377	17	165
152391	Nemertea	Nemertea	3	21	3	17
131069	Annelida	*Neoleanira tetragona*	1	1	1	1
956	Annelida	Nephtyidae spp.	11	17	14	21
157499	Annelida	*Nephtys bucera*	—	—	5	5
130355	Annelida	*Nephtys caeca*	22	31	32	66
130356	Annelida	*Nephtys ciliata*	—	—	2	6
130362	Annelida	*Nephtys incisa*	35	73	37	75
130364	Annelida	*Nephtys longosetosa*	1	1	—	—
129370	Annelida	*Nephtys* sp.	23	6901	—	—
22496	Annelida	Nereididae spp.	1	1	—	—
130404	Annelida	*Nereis pelagica*	—	—	1	1
156916	Mollusca	*Nucula proxima*	—	—	4	11
140577	Mollusca	*Nuculana minuta*	11	13	11	13
1566	Arthropoda	*Nymphonidae spp*	1	1	—	—
117388	Cnidaria	*Obelia geniculata*	1	1	—	—
117389	Cnidaria	*Obelia longissima*	1	1	—	—
117034	Cnidaria	*Obelia* sp.	5	5	—	—
137826	Mollusca	*Oenopota* sp.	9	10	9	10
2036	Annelida	Oligochaeta	20	261	16	144
130494	Annelida	*Ophelia limacina*	11	18	13	16
924	Annelida	Opheliidae spp.	2	3	2	3
125125	Echinodermata	*Ophiopholis aculeata*	2	2	2	2
124933	Echinodermata	*Ophiura robusta*	6	220	6	220
102690	Arthropoda	*Orchomenella minuta*	9	14	10	16
1078	Arthropoda	Ostracoda	37	341	—	—
107240	Arthropoda	*Pagurus pubescens*	3	4	7	9
106854	Arthropoda	*Pagurus* sp.	1	1	1	1
107651	Arthropoda	*Pandalus montagui*	1	1	1	1
903	Annelida	Paraonidae spp.	1	1	—	—

Aphia ID	Phylum	Accepted name	0.5 mm size class		1 mm size class	
			Presence	Density	Presence	Density
111547	Bryozoa	*Parasmittina trispinosa*	1	1	—	—
954693	Mollusca	*Parathyasira equalis*	1	2	1	2
181343	Mollusca	*Parvicardium pinnulatum*	8	13	8	13
156940	Mollusca	*Periploma leanum*	3	5	3	5
110593	Arthropoda	*Petalosarsia declivis*	—	—	1	1
129293	Annelida	*Pherusa* sp.	2	5	—	—
423717	Mollusca	*Philine lima*	1	1	1	1
196322	Mollusca	Philinoidea	1	2	—	—
127524	Arthropoda	*Philomedes* sp.	6	7	4	4
130602	Annelida	*Pholoe longa*	15	117	2	15
335827	Annelida	*Pholoe minuta tecta*	12	154	—	—
129439	Annelida	*Pholoe* sp.	69	304	41	92
1789	Phoronida	Phoronida	18	1140	—	—
101403	Arthropoda	Phoxocephalidae spp.	—	—	1	1
102989	Arthropoda	*Phoxocephalus holbolli*	54	553	29	113
334506	Annelida	*Phyllodoce groenlandica*	22	520	—	—
334510	Annelida	*Phyllodoce maculata*	1	1	—	—
334512	Annelida	*Phyllodoce mucosa*	—	—	3	15
129455	Annelida	*Phyllodoce* sp.	6	16	—	—
931	Annelida	Phyllodocidae spp.	1	1	4	4
334519	Annelida	*Phylo ornatus*	—	—	1	1
793	Platyhelminthes	Platyhelminthes	—	—	3	3
101404	Arthropoda	Pleustidae spp.	1	1	—	—
1091	Arthropoda	Podocopida	8	104	—	—
155879	Arthropoda	Podoplea	1	1	—	—
883	Annelida	Polychaeta	24	24	—	—
129472	Annelida	*Polygordius* sp.	19	729	—	—
939	Annelida	Polynoidae spp.	31	88	33	87
102223	Arthropoda	*Pontogeneia inermis*	1	6	—	—
101527	Arthropoda	*Pontogeneia* sp.	—	—	1	1
103079	Arthropoda	*Pontoporeia femorata*	36	361	30	295
130954	Annelida	*Potamilla neglecta*	—	—	1	10
130326	Annelida	*Praxillella praetermissa*	16	76	10	42
129360	Annelida	*Praxillella* sp.	1	2	—	—
131164	Annelida	*Prionospio steenstrupi*	22	1165	—	—
160446	Mollusca	*Propebela turricula*	6	6	6	6
102443	Arthropoda	*Protomedeia fasciata*	11	44	11	37
102444	Arthropoda	*Protomedeia grandimana*	45	1092	51	925
157836	Arthropoda	*Pseudoleptocuma minus*	4	11	4	17
136246	Arthropoda	*Pseudotanais* sp.	9	31	—	—
139975	Mollusca	*Puncturella noachina*	1	2	—	—
1302	Arthropoda	Pycnogonida	2	4	2	4
131170	Annelida	*Pygospio elegans*	1	1	—	—
423709	Arthropoda	*Quasimelita formosa*	1	2	2	92
423710	Arthropoda	*Quasimelita quadrispinosa*	25	102	28	109
141134	Mollusca	*Retusa obtusa*	1	2	1	2
985	Annelida	Sabellidae spp.	8	14	7	9
127599	Arthropoda	*Sarsicytheridea* sp.	18	284	6	169
127951	Arthropoda	*Sclerochilus contortus*	15	271	—	—
130261	Annelida	*Scoletoma fragilis*	1	2	—	—
129340	Annelida	*Scoletoma* sp.	11	39	10	35
130265	Annelida	*Scoletoma tetraura*	3	3	1	1
130537	Annelida	*Scoloplos armiger*	2	10	—	—

Aphia ID	Phylum	Accepted name	0.5 mm size class Presence	0.5 mm size class Density	1 mm size class Presence	1 mm size class Density
129425	Annelida	*Scoloplos* sp.	25	1299	9	85
110866	Bryozoa	*Scrupocellaria* sp.	2	2	1	1
582749	Mollusca	*Serripes groenlandicus*	1	1	—	—
1614	Cnidaria	Sertulariidae spp.	3	4	3	3
1268	Sipuncula	Sipuncula	29	29	—	—
506189	Mollusca	*Solamen glandula*	18	32	15	27
138597	Mollusca	*Solariella sp.*	1	1	1	1
14635	Mollusca	Solenoidea	11	34	11	23
131183	Annelida	*Spio filicornis*	8	31	—	—
129625	Annelida	*Spio sp.*	9	20	—	—
913	Annelida	Spionidae spp.	1	1	—	—
131187	Annelida	*Spiophanes bombyx*	5	7	1	1
156996	Mollusca	*Spisula solidissima*	5	16	—	—
101409	Arthropoda	Stenothoidae spp.	18	412	—	—
131077	Annelida	*Sthenelais limicola*	1	1	1	1
129595	Annelida	*Sthenelais sp.*	1	1	—	—
129678	Annelida	*Streptosyllis sp.*	2	2	—	—
123390	Echinodermata	*Strongylocentrotus sp.*	1	8	—	—
196391	Mollusca	*Tachyrhynchus erosus*	5	26	21	87
234208	Mollusca	*Testudinalia testudinalis*	1	1	1	1
129249	Annelida	*Tharyx sp.*	—	—	2	7
156454	Mollusca	*Thracia septentrionalis*	6	34	—	—
141663	Mollusca	*Thyasira gouldi*	23	83	20	57
138552	Mollusca	*Thyasira sp.*	30	142	29	118
760340	Bryozoa	*Tricellaria arctica*	1	11	8	44
160421	Mollusca	*Trichotropis bicarinata*	1	1	—	—
117258	Cnidaria	*Tubularia sp.*	2	2	2	2
160488	Bryozoa	*Tubuliporina sp.*	1	1	—	—
140348	Mollusca	*Turritellopsis stimpsoni*	1	1	—	—
158156	Arthropoda	*Unciola irrorata*	4	4	4	4
243	Mollusca	Veneridae spp.	—	—	1	2
1255501	Arthropoda	*Wecomedon nobilis*	1	1	1	1
157005	Mollusca	*Yoldia limatula*	—	—	3	3
157006	Mollusca	*Yoldia myalis*	3	3	8	9

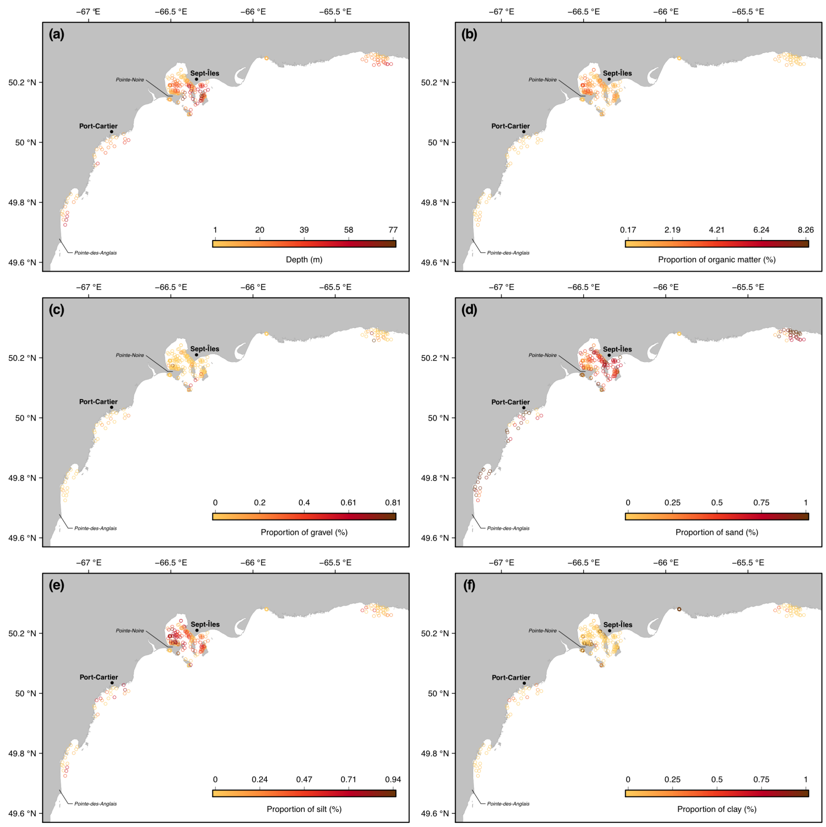


Figure 10 – Values of habitat parameters at each station. (a) Map for station depth; (b) Map for organic matter content; (c) Map for gravel content; (d) Map for sand content; (e) Map for silt content; (f) Map for clay content.

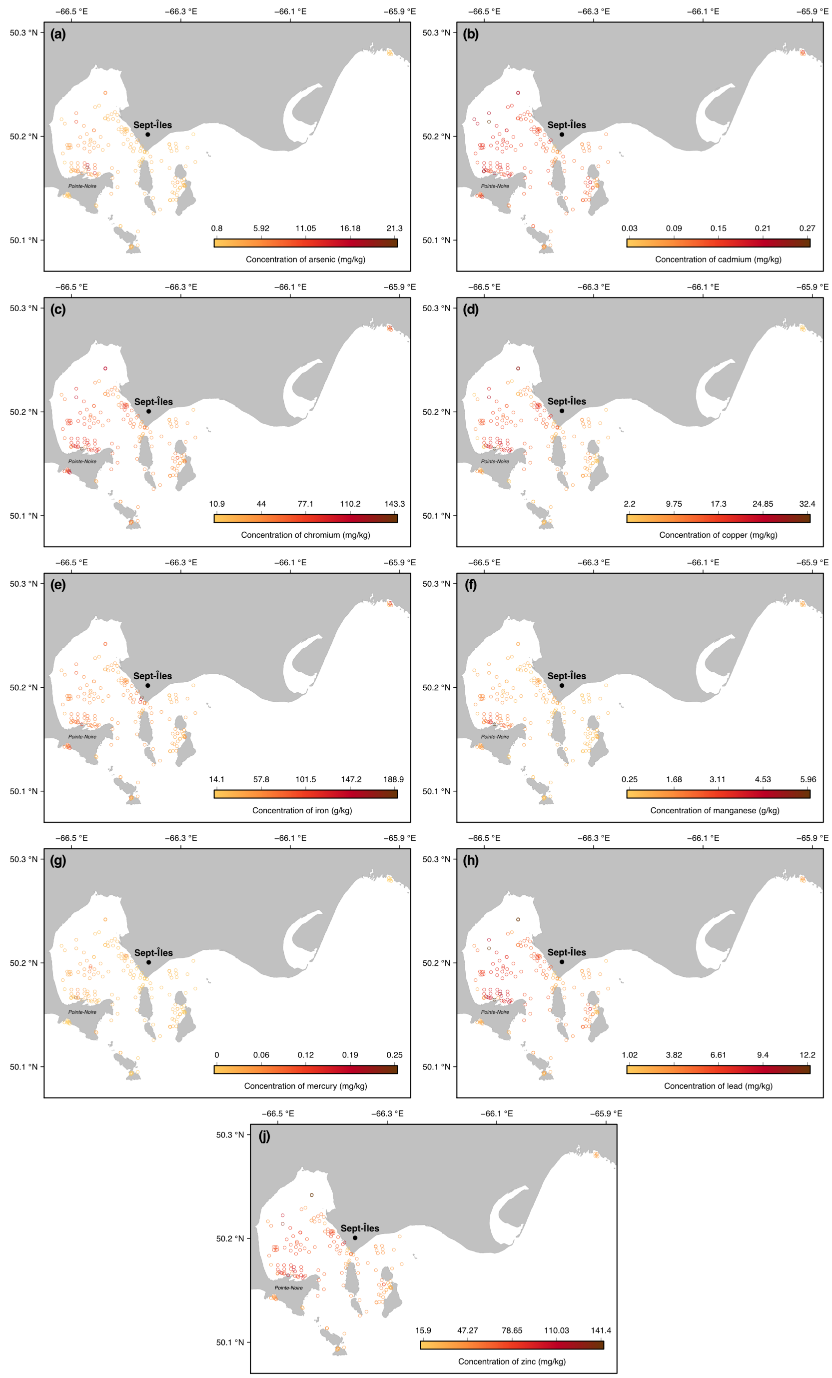


Figure 11 – Values of heavy metal concentrations at each station. (a) Map for arsenic concentration; (b) Map for cadmium concentration; (c) Map for chromium concentration; (d) Map for copper concentration; (e) Map for iron concentration; (f) Map for manganese concentration; (g) Map for mercury concentration; (h) Map for lead concentration; (j) Map for zinc concentration.

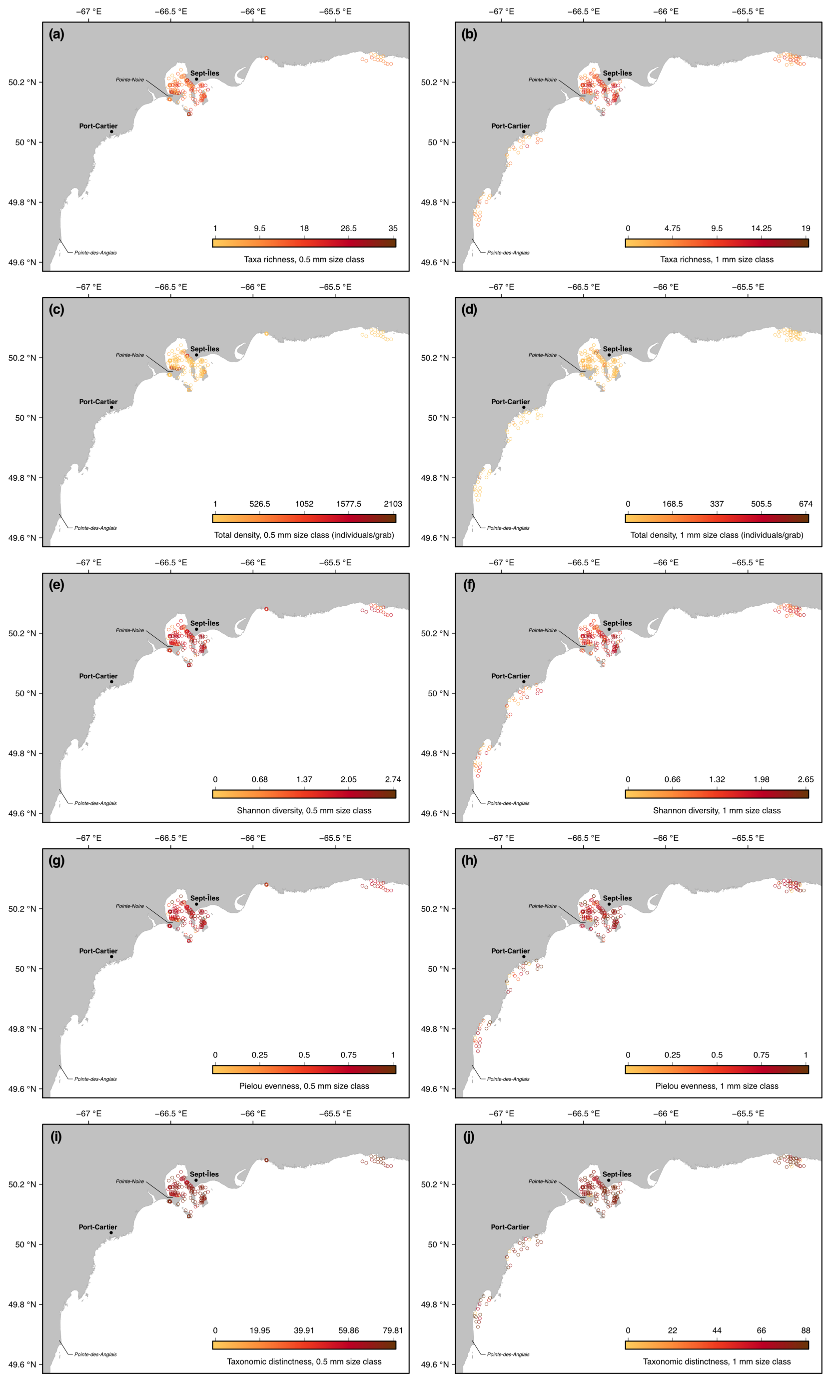


Figure 12 – Values of community characteristics at each station. (a) Map for taxa richness for the 0.5 mm size class; (b) Map for taxa richness for the 1 mm size class; (c) Map for total density for the 0.5 mm size class; (d) Map for total density for the 1 mm size class; (e) Map for Shannon diversity for the 0.5 mm size class; (f) Map for Shannon diversity for the 1 mm size class; (g) Map for Pielou evenness for the 0.5 mm size class; (h) Map for Pielou evenness for the 1 mm size class; (i) Map for taxonomical distinctness for the 0.5 mm size class; (j) Map for taxonomical distinctness for the 1 mm size class.

ARTICLE 2

DÉTERMINER LE STATUT ÉCOLOGIQUE DE COMMUNAUTÉS BENTHIQUES CÔTIÈRES : ÉTUDE DE CAS EN ZONE ANTHROPISÉE SUBARCTIQUE

2.1 Résumé

Compte tenu de l'influence généralisée des activités humaines sur les écosystèmes marins, l'évaluation de l'état écologique fournit des informations précieuses pour les initiatives de conservation et le développement durable. Ainsi, de nombreux indicateurs environnementaux ont été développés dans le monde et il est nécessaire d'évaluer leur performance en calculant l'état écologique dans une variété d'écosystèmes et à de multiples échelles spatio-temporelles. Cette étude a calculé et comparé seize indicateurs de l'état écologique, classés dans trois catégories méthodologiques : mesures d'abondance, paramètres de diversité et espèces caractéristiques. Cette sélection a été appliquée aux écosystèmes benthiques côtiers de Sept-Îles (Québec, Canada), une zone industrialo-portuaire majeure dans le golfe du Saint-Laurent, et mise en relation avec les paramètres de l'habitat (matière organique, fractions granulométriques et concentrations de métaux lourds). Presque tous les indicateurs ont mis en évidence un état écologique généralement bon dans la zone d'étude, où les communautés présentaient un profil non-perturbé, avec une diversité élevée de taxons et de fonctions écosystémiques, sans la dominance des taxons opportunistes. Plusieurs corrélations significatives avec les

paramètres de l'habitat ont été détectées, en particulier avec les métaux lourds, et les analyses de rééchantillonnage ont détecté des résultats relativement solides. Cette étude fournit des renseignements précieux sur l'application d'indicateurs dans les écosystèmes côtiers canadiens, ainsi que sur leur utilisation à des fins d'évaluation environnementale.

L'article associé à ce chapitre, "*Determining the ecological status of benthic coastal communities: a case study in a Canadian industrial harbour area*", a été co-rédigé avec Nicolas Desroy, Lisa Tréau de Coeli, Julie Carrière, Christopher W. McKindsey et Philippe Archambault. Il a été accepté dans la revue *Frontiers in Marine Science*, dans la section spéciale *Biodiversity and Distribution of Benthic Invertebrates - From Taxonomy to Ecological Patterns and Global Processes*, le 5 mars 2021. J'ai établi les objectifs de ce chapitre avec Nicolas Desroy, Christopher W. McKindsey et Philippe Archambault. Je me suis basé sur les données obtenues lors de la campagne d'échantillonnage en 2017 effectuée pour le premier chapitre, en collaboration avec Julie Carrière, auxquelles j'ai ajouté des données sur les traits biologiques collectées depuis différentes bases de données en ligne, validées par Lisa Tréau de Coeli. J'ai calculé les indicateurs environnementaux au cours d'un stage à la station biologique de Dinard (Institut Français de Recherche pour l'Exploitation de la mer) avec Nicolas Desroy, et j'ai ensuite effectué les analyses statistiques pour évaluer et comparer les résultats des différents indicateurs. J'ai dirigé la rédaction de l'article, où l'ensemble des co-auteurs a contribué à l'interprétation des résultats en fonction de leur expertise et à la révision générale. Les données liées à cet article sont accessibles dans le dépôt en ligne hébergé par le site Scholars Portal Dataverse avec l'identifiant unique 10.5683/SP2/WDDDMI. Une partie des résultats de ces analyses a été présentée lors de la Réunion Scientifique du *Canadian Healthy Oceans Network* II à Ottawa en novembre 2018.

Dreujou, E., Desroy, N., Carrière, J., Tréau de Coeli, L., McKindsey, CW., Archambault, P. (2021). Determining the ecological status of benthic coastal communities: a case study in an anthropized sub-Arctic area. *Frontiers in Marine Science* 8:637546. DOI:10.3389/fmars.2021.637546.

Les sections suivantes correspondent à celles de l'article accepté.

DETERMINING THE ECOLOGICAL STATUS OF BENTHIC COASTAL COMMUNITIES: A CASE STUDY IN AN ANTHROPIZED SUB-ARCTIC AREA

2.2 Abstract

With the widespread influence of human activities on marine ecosystems, evaluation of ecological status provides valuable information for conservation initiatives and sustainable development. To this end, many environmental indicators have been developed worldwide and there is a growing need to evaluate their performance by calculating ecological status in a wide range of ecosystems at multiple spatial and temporal scales. This study calculated and contrasted sixteen indicators of ecological status from three methodological categories: abundance measures, diversity parameters and characteristic species. This selection was applied to coastal benthic ecosystems at Sept-Îles (Québec, Canada), an important industrial harbor area in the Gulf of St. Lawrence, and related to habitat parameters (organic matter, grain size fractions, and heavy metal concentrations). Nearly all indicators highlighted a generally good ecological status in the study area, where communities presented an unperturbed profile with high taxa and functional diversities and without the dominance of opportunistic taxa. Some correlations with habitat parameters were detected, especially with heavy metals, and bootstrap analyses indicated quite robust results. This study provides valuable information on the application of environmental indicators in Canadian coastal ecosystems, along with insights on their use for environmental assessments.

Keywords: environmental indicators, ecological status, coastal benthos, macrofauna, Gulf of St. Lawrence

2.3 Introduction

Anthropogenic influences on marine ecosystems occur globally, with possible perturbation of habitats and communities (Halpern et al., 2007, 2019). Many international organizations have recognized the importance of biologically diverse ecosystems for humanity and have established objectives and targets for their protection and sustainable use (United Nations, 1992; Secretariat of the Convention on Biological Diversity, 2010; SDG, 2020). The management of ecosystems requires an understanding of how habitats and communities respond to drivers of change, *i.e.*, forces that affect environmental processes and modify ecosystem state from equilibrium (Boonstra et al., 2015; Beauchesne et al., 2020b; Orr et al., 2020). In addition to natural drivers (*e.g.*, temperature anomalies, freshwater inputs, hypoxic events), influences from human activities (*e.g.*, fisheries, chemical pollution, species introductions) are also considered as ecosystem drivers. As natural and anthropogenic drivers may affect ecosystems concomitantly, it is important to understand how both relate to observed effects (Brown et al., 2014). To tackle these questions, environmental assessments rely on the best available knowledge, acquired through ecological groundwork in ecosystems of interest (such as biodiversity surveys, time series monitoring or experimental studies), and on the communication of results to a wide range of stakeholders (Borja et al., 2012; Borja, 2014; Chapman, 2016; Teixeira et al., 2016). Because such assessments are important foundations for decision makers, it is essential to properly account for the inherent complexity and variability of ecological data.

The use of integrative methods, such as indicators, is particularly relevant in this context. An indicator of ecological status is defined as a quantitative measure that synthesizes ecosystem information to infer ecosystem status (Rice, 2003; Rees et al., 2008). Many holistic frameworks, such as ecosystem-based management, marine spatial planning and DPSIR (Driver Pressure State Impact Responses) models, have included indicators in their methodology (Smeets and Weterings, 1999; Niemi and McDonald, 2004; Rees et al., 2008; Levin et al., 2009; Atkins et al., 2011; Borja et al., 2016; Santos et al., 2019). For example, the Marine Strategy Framework Directive identified indicators and descriptors to monitor the ecological status of European marine waters (European Commission, 2008; Borja et al., 2013, 2015, 2016). However, environmental indicators evaluate specific ecosystem components, perturbations and/or spatiotemporal scales, potentially limiting their applicability in other systems, thus leading to the development of many indicators worldwide (Niemi and McDonald, 2004; Pinto et al., 2009; Teixeira et al., 2016).

One of the ecosystem components most frequently selected for environmental indicators are macrobenthic invertebrates, as they play an important role in the structure and functioning of benthic marine ecosystems (Dauvin and Ruellet, 2007; Pratt et al., 2014). Examples of this include engineering species (*e.g.*, structural features for other species, bioturbation) and interactions with nutrient cycles (*e.g.*, nutrient sequestration in sediments, remineralization, benthic-pelagic coupling) (Largaespada et al., 2012; Link et al., 2013; Belley et al., 2016; Bourque and Demopoulos, 2018). Many macrobenthic species are characterized by a sedentary lifestyle and a relatively long life span, which is particularly interesting when studying human influence as communities will reflect medium-term conditions, resulting in adaptation or local extinction (*e.g.* Dauer, 1993; Borja et al., 2000; Wei et al., 2020).

As pointed out by Rice (2003) and Salas et al. (2006), environmental indicators may be classed into categories according to their methodological basis, including three main categories used in environmental assessments. Category 1 regroups indicators based on measures of abundance – such as density and biomass of individuals – to infer community status. Relationships between abundance and a community status have frequently been discussed, as species do not have the same tolerance to disturbance (Pearson and Rosenberg, 1978). As such, the use of abundance-biomass curves has been proposed to detect if communities are in a balanced state, where K-selected taxa are dominant, compared to a disturbed state, with a dominance of r-selected taxa (Pearson and Rosenberg, 1978; Gray, 1979; Warwick and Clarke, 1994). Category 2 indicators are biodiversity parameters, *i.e.*, community characteristics such as taxa identity and prevalence, which allow complex information to be aggregated into a unique metric. Finally, indicators in Category 3 are computed based on variations of responses of taxa to disturbance. Pioneer works by Pearson and Rosenberg (1978) proposed a model of benthic community evolution along a gradient of organic enrichment, laying the path toward a set of indicators that relate community structure and ecological status.

Environmental indicators, such as the AZTI Marine Biotic Index or the Infaunal Trophic index, have been applied in a number of North American ecosystems, including Chesapeake Bay, Willapa Bay and the Southern California coast (United States), but efficiency to detect perturbation has been mixed (Word, 1978; Maurer et al., 1999; Ferraro and Cole, 2004; Borja et al., 2008b; Pelletier et al., 2018). Less commonly, studies on the Pacific and Atlantic coasts of Canada have also evaluated the utility of existing indicators, although these studies have most often found poor performance (Sutherland et al., 2007; Burd et al., 2008; Callier et al., 2008; Robert et al., 2013). There is thus a need to test and validate indicators for Canadian 201 ecosystems, in particular by comparing outcomes and efficiency of existing methods, which will greatly benefit to tackle ecosystem management objectives within Canada's Ocean Act and the Oceans Strategy (Government of Canada, 1996; Department of Fisheries and Oceans, 2002).

To this end, we evaluated various indicators of ecological status in a coastal industrial harbor area, where human activities may significantly impact local benthic ecosystems. Industrial harbor areas are regions regrouping significant industrial activities coupled with harbor platforms linking production with commercial shipping routes worldwide. We selected the region of Sept-Îles (Québec, Canada) for this study. Located in the Gulf of St. Lawrence, one of the management areas designated by Fisheries and Oceans Canada and a major strategic region for Québec (Department of Fisheries and Oceans, 2009; Daigle et al., 2017; Schloss et al., 2017; Ferrario and Archambault, in press), Sept-Îles is the fourth largest Canadian port in 2019 in terms of total exchanged goods and the second largest in Québec (Statistics Canada, 2011; Binkley, 2020). Industrial activities at Sept-Îles are largely focused on international shipping of iron ore mined in northern Québec and Labrador, the production of aluminum and various fisheries operate in the bay (Department of Fisheries and Oceans, 2019).

The objectives of this study are to (i) compare outcomes of various environmental indicators on benthic ecosystems of the Sept-Îles region and (ii) understand how these indicators relate to habitat parameters for validation and to select appropriate applications.

2.4 Materials and methods

2.4.1 Study area

We targeted ecosystems with a sandy-silty sediment in the industrial harbor area of Sept-Îles (Côte-Nord region of Québec, Canada), which considers ecosystems in the Baie des Sept Îles and the archipelago at its entrance (Figure 13A) (Dreujou et al., 2018, 2020b). Coasts are characterized by sandy beaches, tidal marshes and anthropogenic structures. Mean depth is 35 m in the bay and can reach up to 150 m in the archipelago (Dutil et al., 2012). It is influenced by freshwater inputs from multiple streams and strong tidal currents resulting in a mixed water column and an estuarine circulation (Shaw, 2019). Ecosystems in the Sept-Îles region are considered sub-Arctic due to the formation of ice on the shore in November/December and in the bay in January/February, along with an important freshwater run-off due to snowmelt in April (Demers et al., 2018).

This region hosts several human activities, including industrial, commercial and dredging operations located at the City of Sept-Îles and the Pointe-Noire sector (on the southern section of Baie des Sept Îles), along with an aquaculture site and various fisheries throughout the bay (Figure 13B). Many projects have been done in this region to characterize pelagic and benthic communities and habitats in relation to coastal stressors (Canadian Healthy Oceans Network, 2016; Carrière, 2018; Dreujou et al., 2020b).

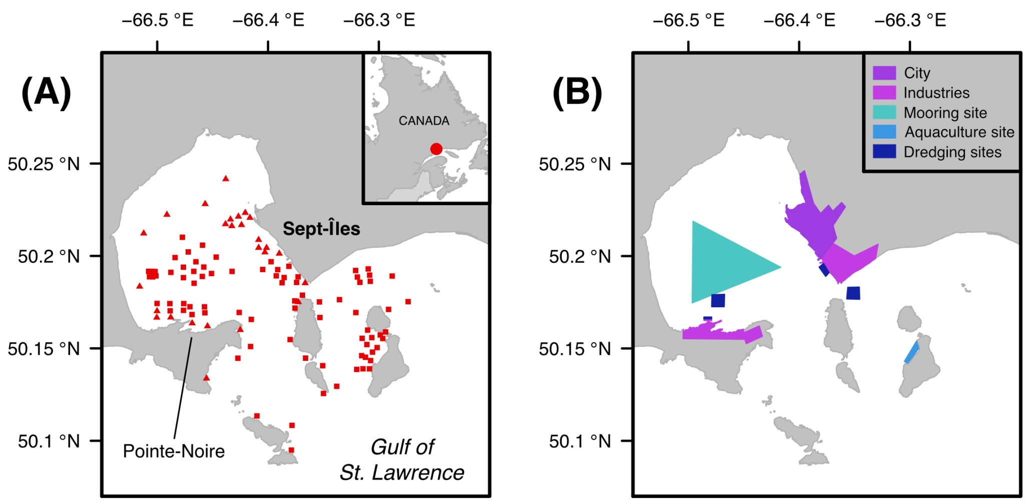


Figure 13 – Map of the study area. (A) Location of the sampled stations, with light blue triangles and dark blue squares representing shallow (< 15 m) and deep (> 15 m) stations, respectively, (B) Location and identity of human activities present in the area.

2.4.2 Benthic ecosystems sampling

The sampling design and methods used to collect and analyze ecological samples were similar to those presented in Dreujou et al. (2020b), with the exception that only one region (Baie des Sept Îles) and one type of community (individuals higher than 0.5 mm) were considered.

A total of 108 stations were selected in the study area, using a randomization algorithm to cover the full extent of the sector, constrained between 0 and 80 m deep, and with increased sampling effort in areas with human activities (Figure 13A). Himmelman (1991) showed that benthic communities in the Northern Gulf of St. Lawrence above and below 15–20 m deep differ. Likewise, preliminary fieldwork in the study region detected a thermocline in the water column at ca. 15 m deep. Consequently, we discriminated two groups of stations in order to ensure habitat homogeneity within depth classes: shallow (<15 m, 26 stations) and deep habitats (>15 m, 82 stations). We sampled the benthic ecosystem in July 2017, using a Ponar grab (0.05 m^2) deployed from a boat, with two independent casts at each station.

The first cast collected two subsamples – one for the analyses of organic matter content and another for sediment grain size – stored at -20°C until processing in the laboratory. The percentage of total organic matter (*i.e.*, sum of organic carbon and organic nitrogen) in the sediment was determined using the Loss-on-Ignition method (Davies, 1974). Grain-size analysis was done on a sieving column for the fraction with particles larger than 2 mm and with a Laser Diffraction Particle Size Analyzer for the smaller fractions. Results from both techniques were combined to yield a unified size distribution range from 0.04 μm to 26.5 mm. From this, percentages of gravel, sand, silt and clay were calculated as defined by Wentworth (1922) and Folk (1980).

All sediment obtained from the second cast was sieved on a 0.5 mm mesh size and preserved in a solution of BORAX-buffered formalin (4%) solution for subsequent benthic macrofauna identification (Dreujou et al., 2020b). The resulting samples were sorted using a stereomicroscope and taxa identified to the lowest taxonomic level possible with reference manuals and identification guides; names were validated according to the World Register of Marine Species (WoRMS Editorial Board, 2020). Taxon density and biomass per grab were recorded by counting and weighting (blotted wet mass) individuals in each sample, respectively.

In addition to these parameters, we considered estimates of heavy metal concentrations in the sediment. Concentrations at the sampled stations were calculated based on values obtained in the same area in 2014 and 2016, retrieved from a database hosted by Carrière (2018), using Inverse Distance Weighting interpolation (Dale and Fortin, 2014). We focused on metals for which toxicity criteria have been defined in the Biological Effects Database for Sediments (Environment Canada and Ministère du Développement Durable de l'environnement et des Parcs du Québec, 2007; Centre d'Expertise en Analyse Environnementale du Québec, 2014): arsenic, cadmium, chromium, copper, mercury, lead and zinc; we also included iron and manganese to account for possible contamination from local ore industries.

2.4.3 Environmental indicator calculation

Indicators of ecological status were selected from Pinto et al. (2009), Teixeira et al. (2016) and DEVOTES (2012), and grouped into three Categories according to their methodology (Table 6). We targeted indicators related to descriptors D1 (biological diversity), D6 (seafloor integrity), and D8 (contaminants) of Good Environmental Status (European Commission, 2008; Borja et al., 2013), choosing those that applied to benthic invertebrates in soft-bottom habitats. We considered each station separately, allowing an assessment of the spatial variability and mean for each indicator, and when possible we pooled all stations together to obtain an estimate for the bay-scale system. We used R v4.0 to perform data manipulations and calculations (R Core Team, 2020).

Table 6 – Summary of the evaluated indicators.

Indicator	Unit	Range	References used
Category 1: Abundance measures			
Total density	ind.grab^{-1}	[0 ; +∞[	—
Total biomass	gWM.grab^{-1}	[0 ; +∞[	—
W-Statistic index	NA	[-1 ; 1]	*Warwick & Clarke (1994)*
Category 2: Diversity measures			
Specific richness	taxa	[0 ; +∞[	—
Shannon index	NA	[0 ; 5]	*Magurran & McGill (2011)*
Margalef index	NA	[0 ; +∞[	*Magurran & McGill (2011)*
Simpson index	NA	[0 ; 1]	*Magurran & McGill (2011)*
Pielou evenness	NA	[0 ; 1]	*Magurran & McGill (2011)*
Taxonomic diversity	NA	[0 ; +∞[	*Warwick & Clarke (1995), Clarke (1998)*
Functional richness	NA	[0 ; +∞[	*Mason et al. (2005), Villéger et al. (2008)*
Functional evenness	NA	[0 ; 1]	*Mason et al. (2005), Villéger et al. (2008)*
Functional divergence	NA	[0 ; 1]	*Mason et al. (2005), Villéger et al. (2008)*
Category 3: Characteristic species			
AZTI Marine Biotic Index	NA	[0 ; 7]	*Borja et al. (2000)*
Multivariate Marine Biotic Index	NA	[0 ; 1]	*Muxika et al. (2007)*
BENTIX	NA	[0 ; 6]	*Simboura & Zenetos (2002)*
Benthic Opportunistic Polychaete Amphipod index	NA	[0 ; log(2)]	*Dauvin & Ruellet (2007)*

We included in Category 1 the total density (number of individuals collected per grab), total biomass (wet mass of individuals collected per grab), and the W-Statistic Index, calculated based on abundance-biomass curves for the community (Warwick and Clarke, 1994). Those indicators were computed using benthic taxa abundance sampled at each station.

For Category 2, we considered taxa richness (number of collected taxa) and related metrics to describe the community's structure and the relative prevalence of taxa within it, such as the Shannon index, Margalef index, Simpson index, and Pielou evenness (Legendre and Legendre, 1998; Magurran and McGill, 2011). We also considered taxonomic and functional diversities, based on taxonomic relationships between taxa and information about biological traits, respectively (Warwick and Clarke, 1995; Clarke and Warwick, 1998; Mason et al., 2005; Villéger et al., 2008). Taxa richness, Shannon index, Margalef index, Simpson index, and Pielou evenness were calculated using the benthic community at each station. For taxonomic diversity, we gathered relatedness data for taxa using the WoRMS online database (WoRMS Editorial Board, 2020). To estimate functional diversity, we computed functional richness, functional evenness and functional divergence (Mason et al., 2005; Villéger et al., 2008) by considering five biological traits – body composition, body size, feeding type, mobility and lifestyle – with a total of 26 modalities (Table 7). Because taxa can present several modalities for a trait, we assigned a continuous value between 0 (absence of the modality) and 1 (presence of the modality) for each taxon and each trait (the sum of values for every modality within a trait equals 1). Biological trait data was extracted from WoRMS, SealifeBase, the Encyclopedia of Life, and Arctic Traits databases as well as dedicated articles (Degen and Faulwetter, 2019; EoL, 2020; Palomares and Pauly, 2020; WoRMS Editorial Board, 2020). R Packages *vegan* and *FD* were used to calculate indicators in this category (Laliberté and Legendre, 2010; Laliberté et al., 2014; Oksanen et al., 2019).

Table 7 – Summary of the functional traits and modalities.

Biological trait	**Modality**
Body composition	Non-calcified tissue
	Calcareous (not specified)
	Calcareous —calcium carbonate
	Calcareous —amorphous calcium carbonate
	Calcareous —aragonite
	Calcareous —calcite
	Calcareous —High magnesium calcite
	Chitinous
Body length	Small (<3 mm)
	Medium (between 3 and 10 mm)
	Large (>10 mm)
Feeding type	Surface deposit feeder
	Subsurface deposit feeder
	Filter/suspension feeder
	Grazer
	Predator
	Scavenger
	Parasite
Mobility	Sessile
	Limited
	Mobile
Lifestyle	Fixed
	Tubicolous
	Burrower
	Crawler
	Swimmer

Finally, indicators in Category 3 included the AZTI Marine Biotic Index (AMBI) and its multivariate version (M-AMBI), which are based on the relative proportion of taxa classified into five ecological groups depending on their tolerance to perturbation (Grall and Glémarec, 1997; Borja et al., 2000; Muxika et al., 2007), BENTIX, where only two ecological groups are considered (Simboura and Zenetos, 2002), and the Benthic Opportunistic Polychaetes Amphipods Index (BOPA), which compares proportions of opportunistic polychaetes and amphipods (Dauvin and Ruellet, 2007). Sampled taxa were assigned to ecological groups, from group I to V, based on the list of Borja et al., version of May 2019 (AZTI, 2019) (Table 11). M-AMBI scores were based on references conditions described in Table 8. Because this list was developed for European taxa, we assigned groups to unregistered taxa based on species physiology studies and taxonomic relationships (Pelletier et al., 2018). We used this list to further regroup taxa to a 'sensitive' (groups I and II) and a 'tolerant' (groups III to V) metagroup to compute BENTIX (Simboura and Zenetos, 2002), and to obtain the proportion of opportunistic polychaetes (groups III to V) and sensitive amphipods (group I) to calculate BOPA (Dauvin and Ruellet, 2007) (Table 11). M-AMBI was calculated using the dedicated software AMBI v5.0 (AZTI, 2019), where 'bad' and 'high' status conditions are required for taxa richness, Shannon index and AMBI (Muxika et al., 2007). Because historical data on benthic invertebrates is scarce in our study area, we used the outcomes of our sampling to establish these values by selecting the 5 and 95 percentiles of the variable distribution (for 'bad' and 'high' status, respectively, Table 12) (Buchet, 2010).

2.4.4 Integration and statistical analysis

Results for each indicator were reviewed qualitatively and compared to benthic ecosystem data in the Gulf of St. Lawrence, when available. Robustness for indicators in Categories 1 and 2 was calculated as the 95% confidence interval using a resampling routine (bootstrap, 1000 replicates), and the difference between averages of each indicator and the resampling averages (*i.e.*, bootstrap bias).

We computed the Ecological Quality Ratios for Category 3 indicators. This ratio compares the value of an indicator to a reference, such as a targeted state or unperturbed/pristine ecosystem, so that an Ecological Quality Status can be assigned (five categories: 'bad', 'low', 'moderate', 'good', and 'high' status). The formula to compute the Ecological Quality Ratio is the following (Bund and Solimini, 2007):

$$EQR = \frac{V_{ind} - R_{bad}}{R_{high} - R_{bad}}$$

V_{ind} is the value of an indicator, R_{bad} is the reference value for a 'bad' status and R_{high} is the reference value for a 'high' status. Limits between each Ecological Quality Status class are specific to the indicator used (Borja et al., 2000; Simboura and Zenetos, 2002; Muxika et al., 2005; Dauvin and Ruellet, 2007; Muxika et al., 2007).

Finally, we explored covariation between indicators and habitat parameters (organic matter content, grain size distribution and heavy metal concentrations), using scatterplots for each pair of variables. Correlation was assessed with Spearman's rank coefficients to understand the relevance of each indicator to the computation of ecological status (Quinn and Keough, 2002).

2.5 Results

Sediment was mostly composed of sand and silt fractions, with concentrations of organic matter rarely surpassing 3%. Heavy metal concentrations did not reach high toxicity levels as defined by Environment Canada (Table 13) (Environment Canada and Ministère du Développement Durable de l'environnement et des Parcs du Québec, 2007; Centre d'Expertise en Analyse Environnementale du Québec, 2014; Dreujou et al., 2020b). A total of 132 taxa were identified, belonging to eight phyla, with a dominance of arthropods, mollusks, and annelids (Table 11). The most abundant taxa were the polychaete *Micronephthys neotena*, the cumacean *Eudorellopsis integra*, the amphipod *Protomedeia grandimana*, Nematoda (adults), and the bivalve *Macoma calcarea* (Table 11). From this list, no species which can be considered as exotic to this region have been reported (Simard et al., 2013).

2.5.1 Indicator outcomes

2.5.1.1 Category 1 indicators

Indicators in this category presented greater mean values in deep than shallow stations, with the exception of total density (Table 8). Shallow stations showed a higher total density than deep stations, but this may be an outlier effect due to a single station close to the City of Sept-Îles (Figure 16A-C), where density was 899 individuals.grab^{-1} with a dominance of *P. grandimana.* Overall, shallow and deep stations presented low total biomass, except for a couple of stations due to the presence of the echinoderms *Echinarachnius parma* and *Strongylocentrotus* sp. (Figure 16A-C). The W-Statistic Index was positive and close to zero at nearly all shallow and deep stations (Figure 16A-C) and the abundance-biomass curve presented higher abundance than biomass values when species were ranked (Figure 14).

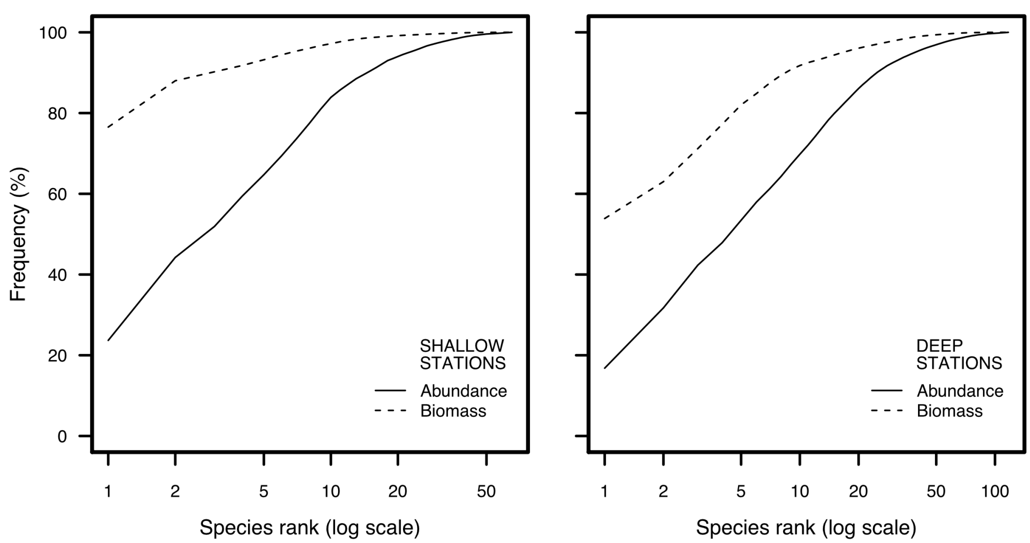


Figure 14 – Values of abundance and biomass for ranked species (logarithm scale), for shallow stations and deep stations.

Table 8 – Values of the mean and standard error (SE) for each indicator, the difference between bootstrapped mean and the true mean (bias) and the 95 % confidence interval (CI), for shallow and deep stations. AMBI = AZTI Marine Biotic Index, M-AMBI = Multivariate AZTI Marine Biotic Index, BOPA = Benthic Opportunistic Polychaetes Amphipods Index.

	Shallow stations (n = 26)				**Deep stations (n = 82)**			
Indicator	**Bay-scale**	**Mean (SE)**	**Bias**	**95 % CI**	**Bay-scale**	**Mean (SE)**	**Bias**	**95 % CI**
Category 1								
Total density	3606	138.7 (36.3)	0.38	[136.9 ; 141.3]	7309	89.13 (7.6)	0.16	[88.81 ; 89.77]
Total biomass	191.16	7.35 (4.2)	0.057	[7.14 ; 7.68]	715.06	8.72 (2.3)	0.024	[8.55 ; 8.84]
W-Statistic index	0.13	0.011 (0.003)	0.016	[0.027 ; 0.028]	0.11	0.025 (0.002)	0.008	[0.033 ; 0.033]
Category 2								
Specific richness	65	9.19 (0.9)	0.036	[9.17 ; 9.29]	117	13.99 (0.5)	0.009	[13.96 ; 14.03]
Shannon index	2.67	1.353 (0.1)	0.007	[1.354 ; 1.366]	3.18	1.952 (0.05)	0.0002	[1.949 ; 1.955]
Margalef index	7.81	1.92 (0.1)	0.014	[1.93 ; 1.95]	13.04	3.05 (0.1)	0.0001	[3.04 ; 3.05]
Simpson index	0.88	0.62 (0.04)	0.003	[0.62 ; 0.63]	0.92	0.77 (0.02)	0.0002	[0.77 ; 0.77]
Pielou evenness	0.64	0.65 (0.05)	0.004	[0.657 ; 0.663]	0.67	0.76 (0.02)	0.0003	[0.76 ; 0.76]
Taxonomic diversity	68.48	51.66 (3.8)	0.357	[51.79 ; 52.25]	74.8	63.48 (1.3)	0.014	[63.39 ; 63.55]
Functional richness	—	23.35 (4.6)	3.171	[26.11 ; 26.93]	—	31.76 (2.5)	7.59	[38.83 ; 39.88]
Functional evenness	—	0.554 (0.04)	0.002	[0.55 ; 0.554]	—	0.632 (0.01)	0.002	[0.633 ; 0.635]
Functional divergence	—	0.77 (0.05)	0.007	[0.77 ; 0.78]	—	0.83 (0.01)	0.011	[0.82 ; 0.82]
Category 3								
AMBI	1.57	1.5 (0.1)	—	—	1.53	1.45 (0.05)	—	—
M-AMBI	—	0.68 (0.05)	—	—	—	0.7 (0.03)	—	—
BENTIX	5.15	4.95 (0.2)	—	—	5.25	5.31 (0.09)	—	—
BOPA	0.002	0.003 (0.001)	—	—	0.004	0.007 (0.003)	—	—

2.5.1.2 Category 2 indicators

Category 2 indicators showed similar trends for shallow and deep stations, while being generally higher for the latter (Table 8). In particular, there is a close similarity between the spatial distributions of taxa richness, Shannon and Margalef indices and taxonomic diversity (Figure 16D-L). Variability for shallow stations is quite low, except for a station in front of Pointe-Noire where only one taxon was present, while deep stations tend to display the highest values in the archipelago compared to the center of the bay (Figure 16D-L). Mean values for the Simpson index and Pielou evenness reached 0.62 (standard error of 0.04) and 0.77 (0.02), respectively, for shallow stations and 0.66 (0.05) and 0.76 (0.02) for deep stations (Table 8). The same relationship between shallow and deep stations is observed for these metrics, even though the distribution for both is skewed with some stations closer to coasts presenting very low values (Figure 16D-L). Concerning functional diversity, deep stations presented higher mean functional richness, functional evenness and functional divergence relative to those at shallow stations (Table 8). The most abundant modality for each biological trait was non-calcified tissue for body composition, small individuals for body size, surface deposit-feeders for feeding type, mobile organisms for mobility and burrowers for lifestyle, at both shallow and deep stations.

2.5.1.3 Category 3 indicators

Classification of taxa into ecological groups to compute Category 3 indicators yielded 51 taxa in group I (sensitive to disturbance, 38.6% of the taxa), 63 in group II (indifferent to disturbance, 47.7%), 11 in group III (tolerant to disturbance, 8.3%), 1 in each of groups IV and V (second- and first-order opportunists, respectively, 0.8%) and 5 were not assigned due to a too broad taxonomic resolution (Table 11). This classified 114 taxa in the 'sensitive' group and 13 in the 'tolerant' group (Table 11). Concerning polychaetes and amphipods, we observed four opportunistic polychaetes (*Cossura longocirrata*, *Eteone* sp., *Hediste diversicolor*, *Praxillella praetermissa*) and nine sensitive amphipods (*Ameroculodes edwardsi*, *Ampelisca vadorum*, *Byblis gaimardii*, Lysianassidae, *Maera danae*, *Phoxocephalus holbolli*, *Pontoporeia femorata*, *Quasimelita formosa*, *Quasimelita quadrispinosa*).

An AMBI score of 1.57 and 1.53 was obtained for the bay-scale estimate at shallow and deep stations, respectively, which corresponds to a 'slight imbalance' site classification (Borja et al., 2000). Overall, low AMBI values were obtained at each station, being 1.5 on average (standard error of 0.13) for shallow stations and 1.45 (0.05) for deep stations, and never exceeding 3, and no particular spatial trend can be observed (Table 8, Figure 16M-P). The bay-scale M-AMBI could not be computed with the percentile method, and at the station level, generally high mean values of 0.68 (0.05) and 0.7 (0.03) were observed for shallow and deep stations, respectively (Table 8). Stations outside of the bay tended to be characterized by higher values than those inside it, especially close to the coast and in the northern section of the bay, but this may be related to the spatial distribution of taxa richness and the Shannon index (Figure 16M-P). The BENTIX bay-scale estimate was 5.15 for shallow stations and 5.25 for deep stations, while at the station-level mean values were 4.95 (0.23) and 5.31 (0.09), respectively (Table 8). These values correspond to a 'normal/pristine' pollution classification for the majority of the area sampled, except for some stations close to coasts (Simboura and Zenetos, 2002). Finally, BOPA produced low scores of 0.002 and 0.004 for shallow and deep bay-scale estimates, respectively, similar to means of 0.0028 (0.0012) for shallow and 0.0067 (0.003) for deep stations, respectively (Table 8), denoting 'high status' classifications. Only two stations had a score higher than 0.05, a trend that is not shared with neighboring stations, which may indicate localized low-intensity perturbations (Figure 16M-P).

Calculation of Ecological Quality Ratios using Category 3 indicators produced similar results for AMBI, BENTIX and BOPA (Figure 15). The majority of stations (shallow and deep) presented a ‘high’ or ‘good’ ecological status except for a few stations with a ‘poor’ status (Figure 15). In contrast, results for M-AMBI were less uniform, with a high variation among both shallow and deep stations, such that no general trends may be highlighted (Figure 15).

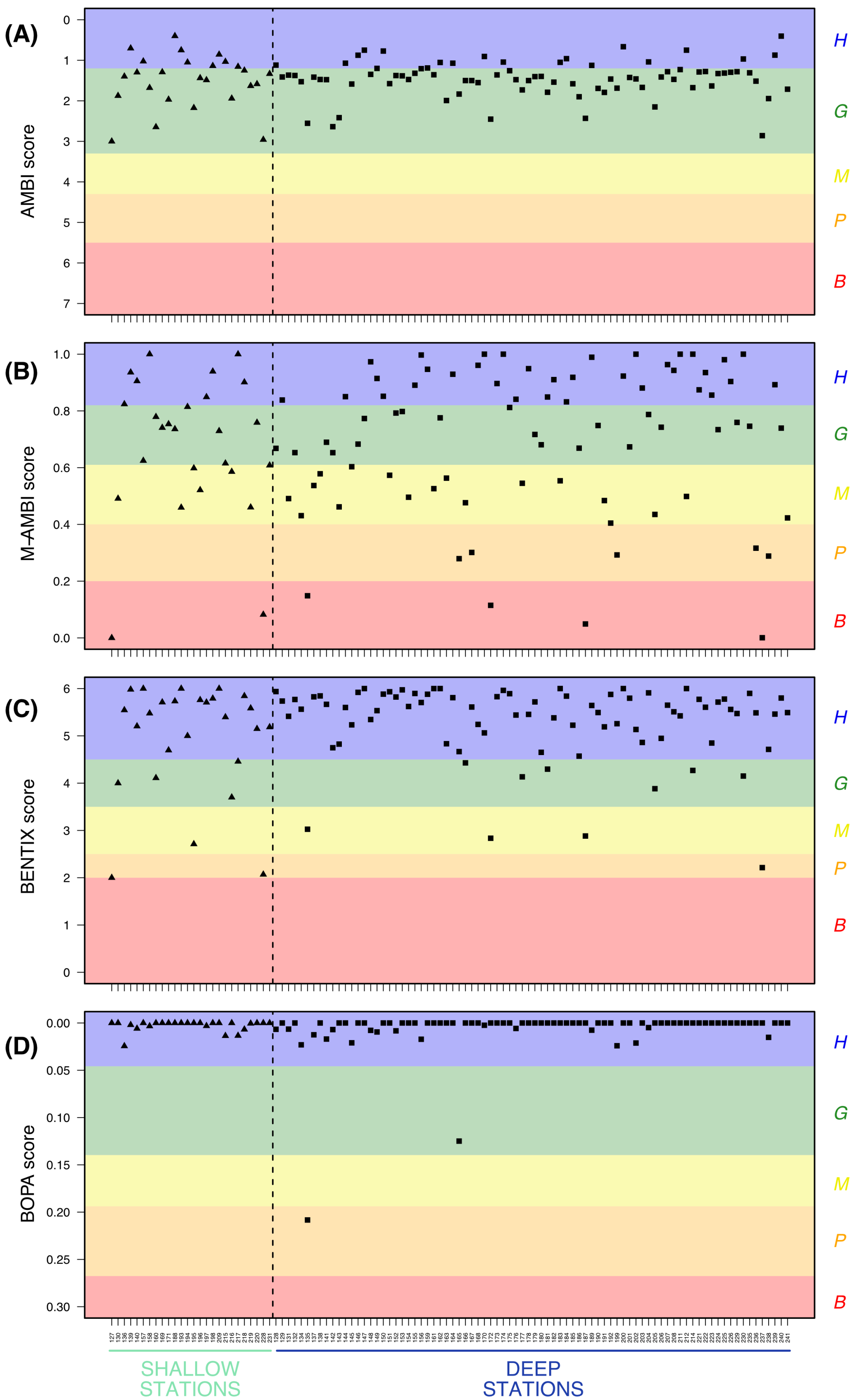

(A)
AMBI score
(B)
M-AMBI score
(C)
BENTIX score
(D)
BOPA score
H
G
M
P
B
SHALLOW STATIONS
DEEP STATIONS

Figure 15 – Values of Category 3 indicators ranked according to Ecological Quality Ratios, calculated for shallow and deep stations. (A) Calculated with the AZTI Marine Biotic Index (AMBI), (B) Calculated with the Multivariate AZTI Marine Biotic Index (M-AMBI), (C) Calculated with the BENTIX, (D) Calculated with the Benthic Opportunistic Polychaetes Amphipods Index (BOPA). B = 'bad' status (red), P = 'poor' status (orange), M = 'moderate' status (yellow), G = 'good' status (green), H = 'high' status (blue).

2.5.2 Robustness and covariation

For Category 1 and 2 indicators, bootstrap bias was low at both shallow and deep stations (less than 0.4), except for functional richness where it reached 3.17 and 7.59, respectively (Table 8), demonstrating a relatively high robustness of the indicators. The true mean was included in the 95% confidence interval for five indicators at shallow stations (taxa richness, total density, total biomass, functional evenness, functional divergence) and eight at deep stations (taxa richness, total density, total biomass, Shannon index, Margalef index, Simpson index, Pielou evenness, taxonomic diversity) (Table 8).

The analysis of covariation between indicators reported moderate to very high Spearman coefficients ($0.22 < |\rho| < 0.96$) (Table 9). Category 2 indicators presented the highest proportion of within-Category significant correlations at both shallow and deep stations (Table 9). The vast majority of these correlations were positive, with the strongest correlations between Shannon and Margalef indices, and were represented by linear proportionality between indicators on the scatterplots. Category 2 indicators were also frequently correlated to indicators from Categories 1 and 3, especially for the W-Statistic Index and the M-AMBI (Table 9). The latter Categories did not present high within-Category correlations, except between AMBI/BENTIX and M-AMBI/BOPA at shallow stations, and the W-Statistic Index and AMBI at deep stations.

Table 9 – Spearman rank correlation coefficients between indicators, for shallow and deep stations. Only significant relationships (on the lower section of the triangular matrix) are presented. TD = total density, TB = total biomass, W = W-Statistic index, S = taxa richness, H = Shannon index, M = Margalef index, λ = Simpson index, J = Pielou evenness, Δ = taxonomic diversity, FR = functional richness, FE = functional evenness, FD = functional divergence, AMBI = AZTI Marine Biotic Index, M-AMBI = Multivariate AZTI Marine Biotic Index, BOPA = Benthic Opportunistic Polychaetes Amphipods Index.

	Category 1			**Category 2**									**Category 3**			
Indicator	**TD**	**TB**	**W**	**S**	**H**	**M**	λ	**J**	Δ	**FR**	**FE**	**FD**	**AMBI**	**M-AMBI**	**BENTIX**	**BOPA**
Shallow stations																
Category 1																
TD																
TB	—															
W	—	—														
Category 2																
S	0.77	0.43	—													
H	—	—	0.62	0.58												
M	—	—	0.53	0.76	0.81											
λ	—	—	0.68	—	0.89	0.61										
J	-0.66	—	0.59	—	0.46	—	0.7									
Δ	-0.44	—	0.71	—	0.59	0.48	0.75	0.86								
FR	0.8	0.5	—	0.87	—	0.58	—	-0.41	—							
FE	—	—	0.67	—	0.58	0.41	0.65	0.54	0.51	—						
FD	—	—	—	0.41	—	—	—	—	—	—	—					
Category 3																
AMBI	—	-0.42	—	—	—	—	—	—	—	—	—	—				
M-AMBI	—	0.48	—	0.8	0.78	0.86	0.5	—	—	0.64	0.4	0.43	—			
BENTIX	—	—	—	—	—	—	—	—	—	—	—	—	-0.78	—		
BOPA	—	—	—	0.45	—	0.41	—	—	—	—	—	—	—	0.53	—	

(Table 9 continued)

	Category 1			Category 2									Category 3			
Indicator	**TD**	**TB**	**W**	**S**	**H**	**M**	**λ**	**J**	**Δ**	**FR**	**FE**	**FD**	**AMBI**	**M-AMBI**	**BENTIX**	**BOPA**
Deep stations																
Category 1																
TD																
TB	—															
W	-0.31	0.35														
Category 2																
S	0.58	—	0.37													
H	—	—	0.75	0.67												
M	—	—	0.61	0.9	0.86											
λ	-0.23	—	0.75	0.47	0.96	0.7										
J	-0.67	—	0.63	—	0.64	0.29	0.79									
Δ	-0.39	—	0.69	0.28	0.81	0.57	0.89	0.88								
FR	0.35	—	0.32	0.71	0.46	0.67	0.33	—	—							
FE	-0.55	—	0.42	—	—	—	0.31	0.59	0.43	—						
FD	—	—	-0.32	-0.27	-0.39	-0.37	-0.41	-0.39	-0.5	-0.28	—					
Category 3																
AMBI	—	—	-0.29	—	-0.25	-0.23	-0.28	-0.31	-0.3	—	—	0.32				
M-AMBI	—	—	0.64	0.79	0.87	0.89	0.76	0.39	0.6	0.58	—	-0.4	-0.52			
BENTIX	—	—	—	—	—	—	—	—	—	—	—	-0.24	-0.7	—		
BOPA	—	—	—	—	—	—	—	—	-0.22	—	—	—	—	—	—	

2.5.3 Relationships with habitat parameters

Correlations between Category 1 indicators and abiotic parameters detected non-significant relationships with sediment parameters (except between the W-Statistic Index and gravel and sand contents at deep stations), while they were significant and negative between most heavy metals and total density and total biomass at shallow stations, and the W-Statistic Index at deep stations (Table 10). The absolute value of Spearman's rank coefficients was high for total density and total biomass at shallow stations (between -0.4 and -0.61), highlighting relatively strong relationships, while they were less for the W-Statistic Index at deep stations (between -0.22 and -0.29).

For Category 2 indicators, correlations with sediment parameters were significant only for some cases involving taxa richness, the Margalef index, taxonomic diversity and functional richness (Table 10). Relationships with heavy metals were detected mainly at deep stations, in particular for cadmium, copper, lead and zinc; at shallow stations, functional richness showed significant correlations with all heavy metals except cadmium, while functional divergence and taxa richness presented marginal correlations. The vast majority of these relationships were moderate to high (between -0.22 and -0.45), except at deep stations for gravel and sand contents and between functional divergence and some heavy metals.

Finally, several significant relationships were observed between Category 3 indicators and sediment parameters (organic matter, sand and silt contents), including at shallow stations for AMBI and BENTIX and at deep stations for BENTIX and BOPA (Table 10). Organic matter was negatively correlated with AMBI values (coefficient of -0.43) at shallow stations and positively with BENTIX values at shallow and deep stations (0.45 and 0.27, respectively); sand and silt contents had the opposite effect at shallow stations for AMBI (0.47 and -0.47, respectively) and at deep stations for BENTIX (-0.26 and

0.23, respectively) and BOPA (-0.31 and 0.34, respectively) values. Many relationships with heavy metals were detected at deep stations for all indicators except AMBI (Table 10). In particular, M-AMBI presented negative correlations with heavy metals (between -0.24 and -0.38) whereas correlations with BENTIX and BOPA were positive (between 0.23 and 0.36).

Table 10 – Spearman rank correlation coefficients between environmental indicators and habitat parameters, for shallow and deep stations. Only significant relationships are presented. TD = total density, TB = total biomass, W = W-Statistic index, S = taxa richness, H = Shannon index, M = Margalef index, λ = Simpson index, J = Pielou evenness, Δ = taxonomic diversity, FR = functional richness, FE = functional evenness, FD = functional divergence, AMBI = AZTI Marine Biotic Index, M-AMBI = Multivariate AZTI Marine Biotic Index, BOPA = Benthic Opportunistic Polychaetes Amphipods Index, OM = organic matter, As = arsenic, Cd = cadmium, Cr = chromium, Cu = copper, Fe = iron, Mn = manganese, Hg = mercury, Pb = lead, Zn = zinc.

	Sediment parameters					**Heavy metal concentrations**								
Indicator	**OM**	**Gravel**	**Sand**	**Silt**	**Clay**	**As**	**Cd**	**Cr**	**Cu**	**Fe**	**Mn**	**Hg**	**Pb**	**Zn**
Category 1														
TD	—	—	—	—	—	-0.46	—	-0.52	-0.55	-0.49	—	-0.52	-0.55	-0.52
TB	—	—	—	—	—	-0.42	-0.42	-0.59	-0.51	-0.39	-0.53	—	-0.5	-0.61
W	—	—	—	—	—	—	-0.4	—	—	—	—	—	—	—
Category 2														
S	—	—	—	—	—	-0.47	—	—	—	—	—	-0.39	—	—
H	—	—	—	—	—	—	—	—	—	—	—	—	—	—
M	—	—	—	—	—	—	—	—	—	—	—	—	—	—
λ	—	—	—	—	—	—	—	—	—	—	—	—	—	—
J	—	—	—	—	—	—	—	—	—	—	—	0.42	—	—
Δ	—	—	—	—	—	—	—	—	—	—	—	—	—	—
FR	—	—	—	—	—	-0.43	—	-0.5	-0.43	-0.47	-0.46	-0.5	-0.42	-0.47
FE	—	—	—	—	—	—	—	—	—	—	—	—	—	—
FD	—	—	—	—	—	-0.6	—	—	—	—	—	—	-0.4	-0.4
Category 3														
AMBI	-0.43	—	0.47	-0.47	—	—	—	—	—	—	—	—	—	—
M-AMBI	—	—	—	—	—	—	—	—	—	—	—	—	—	—
BENTIX	0.45	—	—	—	—	—	—	—	—	—	—	—	—	—
BOPA	—	—	—	—	—	—	—	—	—	—	—	—	—	—

(Table 10 continued)

	Sediment parameters					Heavy metal concentrations								
Indicator	**OM**	**Gravel**	**Sand**	**Silt**	**Clay**	**As**	**Cd**	**Cr**	**Cu**	**Fe**	**Mn**	**Hg**	**Pb**	**Zn**
Deep stations														
Category 1														
TD	—	—	—	—	—	—	—	—	—	-0.23	—	—	—	—
TB	—	—	—	—	—	—	—	—	—	—	—	—	—	—
W	—	0.24	-0.22	—	—	-0.29	-0.29	-0.27	-0.24	-0.27	-0.26	—	-0.28	-0.29
Category 2														
S	-0.25	—	—	—	—	-0.27	-0.32	-0.31	-0.32	-0.45	-0.31	—	-0.3	-0.34
H	—	—	—	—	—	-0.29	-0.29	-0.33	-0.29	-0.36	-0.31	-0.25	-0.31	-0.31
M	-0.26	—	—	—	—	-0.32	-0.33	-0.36	-0.37	-0.45	-0.36	-0.28	-0.35	-0.38
λ	—	—	—	—	—	-0.22	-0.23	-0.27	-0.22	-0.28	-0.25	-0.22	-0.26	-0.24
J	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Δ	—	—	0.23	-0.29	—	-0.29	-0.32	-0.35	-0.35	-0.35	-0.34	-0.33	-0.34	-0.35
FR	—	0.25	—	—	—	-0.25	-0.32	-0.29	-0.32	-0.36	-0.29	-0.28	-0.27	-0.33
FE	—	—	—	—	—	—	—	-0.22	-0.25	—	-0.27	-0.27	-0.24	-0.22
FD	—	—	—	—	—	—	0.28	—	0.29	—	—	0.29	0.28	0.34
Category 3														
AMBI	—	—	—	—	—	—	—	—	—	—	—	—	—	—
M-AMBI	—	—	—	—	—	-0.24	-0.3	-0.3	-0.27	-0.38	-0.28	—	-0.28	-0.31
BENTIX	0.27	—	-0.26	0.23	—	0.23	0.23	0.24	0.25	—	—	0.23	—	—
BOPA	—	—	-0.31	0.34	—	0.33	0.28	0.36	0.31	0.33	0.38	0.3	0.33	0.3

2.6 Discussion

2.6.1 Strengths, limitations, and ecological considerations of indicators

The analysis of benthic communities using Category 1 indicators relies on abundance relationships (either density or biomass of individuals) without consideration of taxonomic identity. Their calculation requires the least laboratory and analytical time relative to the other calculated indicators. Deep stations present a higher density of benthic organisms than shallow stations, as predicted by patterns of coastal marine biodiversity (Gray and Elliott, 2009; Levinton, 2013; Piacenza et al., 2015). The abundance-biomass curve for shallow and deep stations is characteristic of an unstressed profile (Pearson and Rosenberg, 1978; Warwick and Clarke, 1994), which is further supported by the W-Statistic Index being positive and close to 0 at nearly all stations (Clarke, 1990). Studying communities through abundance relationships thus provides interesting results concerning the status of the ecosystem, but the main assumption behind indicators in this Category is that all species are equivalent and have an identical role in the ecosystem structure and functioning. This, however is not necessarily true, as some species can be considered 'key species' in ecosystems due to unique engineering or trophic roles (*e.g.* Bond, 1994; Lawton and Jones, 1995). Thus, Category 1 indicators should be coupled with ancillary methods focusing on biological characteristics of the species, such as life-history traits and physiological characteristics.

Category 2 indicators focus on community biodiversity, granting additional detail than that provided by Category 1 indicators. The notion of biodiversity can be interpreted along multiple points of view in an ecosystem, such as the diversity of species, genes, habitats or functions (United Nations, 1992; Wilson, 1992; Hooper et al., 2005; Stachowicz et al., 2007). While each targeted component has specific implications for the ecosystem, high richness and high diversity values have generally been interpreted as signs of good ecological status (Covich et al., 2004; Borja et al., 2013). This statement needs to be considered carefully, as it is necessary to discuss results with comparable ecosystems and historical data so that diversity trends are interpreted according to local background patterns (Covich et al., 2004).

Taxa richness indicated a quite diverse community, being nearly twice as great at deep stations, as expected given general trends (Gray and Elliott, 2009; Levinton, 2013; Piacenza et al., 2015). These numbers (132 taxa observed) are comparable to results from available benthic invertebrate surveys done in the study area, such as 27 taxa reported by OBIS (2020) in the Gulf of St. Lawrence. Results from diversity indicators (Shannon index, Margalef index, Simpson index, Pielou evenness and taxonomic diversity) showed moderate to high benthic diversity, with no dominance by any taxa (even distribution) and great taxonomic breadth. Few stations differ from this general trend, and those that do are mostly close to coasts where diversity is low and there is no clear evidence of perturbation. Diversity indicators are frequently used in ecological studies to characterize communities and to detect disturbance, which allows discussing their results building on a vast corpus of studies worldwide (Magurran and McGill, 2011). These univariate estimates, however, may mask individual responses arising for example from adaptation or changes in biotic relationships, and they should be coupled with multivariate methods, such as ordination or similarity analysis, so that community-level effects are accurately described (Legendre and Legendre, 1998; Quinn and Keough, 2002; Magurran and McGill, 2011).

Concerning functional diversity, functional richness is generally lower at shallow stations and the two other indicators are in the same range, albeit being slightly greater at deep stations. These results suggest that taxa at shallow stations have more specialized niches, *i.e.*, less diverse functional strategies (Villéger et al., 2008), indicating some redundancy of biological traits. This property is linked to an increased ecosystem stability and resilience, where possible extinctions due to perturbation will not modify the ecosystem structure even if some taxa disappear (Rosenfeld, 2002; Mouillot et al., 2013). However, bootstrap bias was very high for this indicator, making conclusions less robust. Moderate to high functional evenness and divergence denote that values for given biological traits are not evenly distributed and are skewed toward extremes (0 or 1). The consideration of biological traits in addition to species identity allows to study functions of the ecosystem, such as productivity, chemical elements cycling or energy transfers (?Bellwood et al., 2019). This is an important addition to environmental assessments, as biological traits and adaptative responses to disturbance are highly related (Mouillot et al., 2013; Miatta et al., 2021). Indicators of functional diversity thus offer valuable information to characterize communities in complement to other Category 2 indicators, at the expense of increased analytical time to assemble a traits database, for which information may be lacking or difficult to obtain.

For Category 3 indicators, community ecological status is assessed by considering the tolerance of taxa to perturbation. Values for these indicators highlight an overall high status in the study area, where taxa sensitive to perturbation are present without a dominance of opportunists, as illustrated by the number of stations with a 'high' Ecological Quality Status. M-AMBI detected greater variability between stations relative to the other indicators, particularly within the bay. A possible interpretation for this result may be the influence of the percentile method used to compute 'bad' and 'high' status conditions for this indicator, advocating for a careful description of comparison conditions and a fortiori references values (Borja et al., 2012). Furthermore, classification into ecological groups may introduce bias, as the list of Borja et al. (2000) was primarily designed for European coastal ecosystems. Inclusion of taxa found on Canadian coasts has been made based on taxonomic similarity with species already included in the list, by reviewing studies on perturbation tolerance and by expert opinion, but these choices need to be ground-truthed by dedicated ecological works. A wide spectrum of indicators may be included within Category 3, as compiled by Pinto et al. (2009) and Teixeira et al. (2016), and their use in environmental assessments is linked to scientific and management objectives. Such indicators have been used with success in a variety of ecosystems (*e.g.*, Borja et al., 2008a; Gillett et al., 2015), but they require a high volume of data to accurately relate taxa and perturbation status, such as field observations, modeling of species distributions (native and non-indigenous), physiological studies and experimental work.

With these results, we can compare strengths and limitations of the calculated indicators. Category 1 indicators gather relevant baseline information on the ecosystem and requires the least time to be computed. The downside is that it is difficult to discriminate between anthropogenic perturbation and natural variability as other community characteristics may be impacted and, most importantly, they cannot be compared to reference conditions of ecological status. Category 2 indicators, such as the commonly used Shannon index or Pielou evenness, are easy to compute from well-built taxa lists, although taxonomic and functional diversity demand more time to gather complementary information about phylogenetic relationships and ecosystem functions. However, the latter indicators provide more information on the community structure and are backed by ecological literature to infer a certain ecological status (Magurran and McGill, 2011). Finally, Category 3 indicators demand the most time to be calculated, in particular with the classification of taxa relative to their response to disturbance, but they have been specifically designed to determine Ecological Quality Ratios and to consider reference conditions. Bias or uncertainty may be introduced during the classification process as extensive experimental groundwork is needed to properly assign taxa to groups, which is not always available. Many of these indicators are region-specific, with possible poor performance in other ecosystems (*e.g.* Callier et al., 2008; Robert et al., 2013), so that further research is needed to properly assess ecological status in sub-Arctic regions.

2.6.2 Implications for Sept-Îles and Canadian ecosystems

Sept-Îles is an important industrial harbor area for Québec, with a variety of economic activities taking place in the bay and archipelago. All calculated indicators except M-AMBI pointed toward diverse benthic communities of generally good ecological status and no particular perturbation patterns have been detected in the study area, which is coherent with previous descriptions of benthic ecosystems in this region (Carrière, 2018; Dreujou et al., 2020b). When applying Category 3 indicators on the data from these studies, we obtained similar conclusions with ecological status indicated as 'good' to 'high' for AMBI, BENTIX and BOPA.

No particular trend has been observed for stations identified as potentially impacted by Dreujou et al. (2020b) in coastal regions close to the City of Sept-Îles and Pointe-Noire, further suggesting an overall limited effect of perturbations. Compared to the regions where many of the assayed indicators were developed, *e.g.*, in Atlantic and Mediterranean European ecosystems, the magnitude of human activity is considerably lower at Sept-Îles. As such, it is possible the range of variation induced by anthropogenic perturbation is not sufficient to severely impact benthic ecosystems. This is even more relevant when we compare these results to other industrial harbor areas worldwide, where human influence is more pronounced (*e.g.*, Hewitt et al., 2005; Borja et al., 2006; Chan et al., 2016; Birch et al., 2020). Other hypotheses may explain this, such as (i) high community resilience and resistance, (ii) limitation of effective impacts of activities by the dynamic of the ecosystem (*e.g.*, flushing from tidal currents), and (iii) perturbation effects may be more pronounced on other components (such as phytoplankton or pelagic species).

Ecological indicators represent a valuable method to set conservation targets and to guide sustainable environmental projects. In light of initiatives such as the Marine Strategy Framework Directive, where indicators and descriptors have been identified to monitor the ecological status of European marine waters (European Commission, 2008; Borja et al., 2013, 2015, 2016), local stakeholders have the possibility to build on these works to establish ecosystem-based management adapted to Canadian ecosystems. Further research in other industrial coastal areas, including long-term monitoring, is needed to obtained coherent and robust environmental assessments.

2.6.3 Validation and limitations

Assessing relationships between indicators highlighted correlations, especially among Category 2 indicators. While this does not necessarily imply causality in the interpretations, covariation indicates that information gathered by some indicators is similar. This was expected for indicators relying on specific ecosystem components to be computed, such as M-AMBI and the Shannon index, both of which being a function of taxa richness. With an environmental assessment perspective, these results show that calculation of some indicators will not provide additional information when the objective is to detect trends in a targeted area. Understanding these links will allow refining methodological protocols and to produce more efficient and accurate assessments.

The use of ecological status indicators requires a validation procedure to ensure that outcomes are relevant (Dauvin et al., 2010; Heink et al., 2016; Burgass et al., 2017; Moriarty et al., 2018). Because the region of Sept-Îles is not frequently represented in the scientific literature, it is then difficult to have baseline data to validate ecosystem assessments. The studies of Dreujou et al. (2018) and Dreujou et al. (2020b) represent the only campaigns describing benthic habitats and communities in the Baie des Sept Îles, and increased sampling would greatly improve indicator validation, in particular for Category 1 and 2 indicators where references conditions are currently not available.

Inference of ecological status based on Category 3 indicators is relying on explicit reference conditions for 'bad' and 'high' status. Defining values for these conditions based on contemporary ecosystems will most certainly introduce bias, as most are likely to show some level of degradation (which cannot be assessed), and alternatives, such as historical datasets, are rare (Muxika et al., 2007; Borja et al., 2012).

Overall, Category 1 and 2 indicators were relatively robust, with little difference between mean values calculated from the real and bootstrapped datasets (except for functional richness), indicating quite homogeneous results. The vast majority of significant correlations between indicators and environmental factors were found for heavy metal concentrations and most such correlations were negative. This implies that indicators would successfully detect perturbation due to heavy metal content, thus resulting in reduced ecological status, but fail to detect perturbations affecting other habitat parameters. AMBI and BENTIX were correlated to organic matter content, which was expected as original works of Borja et al. (2000) and Simboura and Zenetos (2002) were based on models predicting community changes in response to an organic enrichment (Pearson and Rosenberg, 1978; Grall and Glémarec, 1997). Concerning grain size variables, only sand and silt contents showed any significant correlations with indicators, mainly at deep stations, which may be due to very low amounts of gravel and clay in the sediments.

It is important to note that indicators summarize complex ecological data into unique values (univariate in a statistical point of view), which may be insufficient to correctly assess perturbation. Category 3 indicators were developed for specific types of disturbance, such as organic matter loading or oil-spill detection (Pearson and Rosenberg, 1978; Borja et al., 2000; Dauvin and Ruellet, 2007), and for specific ecosystems (*e.g.*, European Commission, 2008). Even though we detected significant relationships with heavy metal concentrations, dedicated methods to monitor these types of perturbation would greatly benefit this portrait. Finally, the consideration of cumulative impacts from various sources of disturbance may be a good perspective for environmental assessments (Crain et al., 2008), in order to develop more holistic indicators.

2.7 Conclusion

This study provides insight on the use of environmental indicators in Canadian coastal ecosystems, in particular by applying and comparing indicators within an important industrial harbor area. An overall good status for benthic ecosystems was detected in our study area, which is a valuable addition to guide stakeholders and ecosystem management at the local scale. We were able to present strengths and caveats for each Categories of indicators, relating them to ecological implications and possible improvements. Finally, we were able to study the robustness of indicators and we highlighted improvements for ground-throughing and validation.
Further environmental assessments in sub-Arctic coastal areas in the Gulf of St. Lawrence and the Canadian Eastern Atlantic coast are needed to obtain a broader portrait of the region with a more diverse range of environmental conditions. Long-term monitoring will also produce reliable time series data to better understand variability of sub-Arctic benthic ecosystems.

2.8 Acknowledgements

The authors would like to acknowledge and thank all the people that helped during the field campaigns, lab work and data analysis: Claudy Dechêsne, Serge Gallienne, Dany Lévesque, Aurélie Foveau, Cindy Grant, Laure de Montety, Philippe-Olivier Dumais and Raphaël Bouchard. This study is a contribution to the research program of FRQNT Strategic Network Québec-Océan, which also provided scientific support.

2.9 Supplementary material

Table 11 – Classification of the sampled taxa into ecological groups defined by Grall and Glémarec (1997); Borja et al. (2000); Simboura and Zenetos (2002); Dauvin and Ruellet (2007). Density corresponds to the number of individuals sampled per grab at shallow (< 15 m) and deep (> 15 m) stations. The confidence score goes from 3 (highest confidence) to 0 (lowest confidence) depending on the level of certainty for the taxon classification. AMBI = AZTI Marine Biotic Index, BOPA = Benthic Opportunistic Polychaetes Amphipods Index, CS = confidence score, S = sensitive, T = tolerant, SA = sensitive amphipod, OP = opportunistic polychaete, NA = not assigned.

(Table on next pages)

	Density		Ecological groups				
Taxon name	**Shallow**	**Deep**	**AMBI**	**BENTIX**	**BOPA**	**References**	**CS**
Aceroides (Aceroides) latipes	3	54	II	S	–	Borja et al. (2000)	3
Akanthophoreus gracilis	0	154	I	S	–	Borja et al. (2000)	3
Alamprops quadriplicatus	2	0	I	S	–	Borja et al. (2000)	3
Ameritella agilis	5	7	II	S	–	Borja et al. (2000)	3
Ameroculodes edwardsi	0	9	I	S	SA	Borja et al. (2000)	3
Ampelisca vadorum	0	1	I	S	SA	Borja et al. (2000)	3
Amphipoda	30	86	NA	NA	–	–	–
Anonyx lilljeborgi	2	14	II	S	–	Borja et al. (2000)	3
Anthozoa	0	1	II	S	–	Borja et al. (2000)	1
Arcteobia anticostiensis	0	1	II	S	–	Borja et al. (2000) (Polynoidae)	2
Arrhoges occidentalis	0	1	I	S	–	Borja et al. (2000) (Aporrhais sp)	2
Astarte sp	0	9	I	S	–	Borja et al. (2000)	2
Axinopsida orbiculata	18	80	III	T	–	Borja et al. (2000)	3
Axiothella catenata	0	5	I	S	–	Borja et al. (2000) (Axiothella sp)	2
Bathymedon longimanus	2	9	II	S	–	Borja et al. (2000)	3
Bathymedon obtusifrons	1	14	II	S	–	Borja et al. (2000)	3
Boreochiton ruber	0	24	I	S	–	Borja et al. (2000)	3
Brachydiastylis sp	0	2	II	S	–	Borja et al. (2000)	2
Byblis gaimardii	0	8	I	S	SA	Borja et al. (2000)	3
Cancer irroratus	1	0	II	S	–	Gittenberg & Van Loon (2013) (C. pagurus)	1
Caprella septentrionalis	277	0	II	S	–	Borja et al. (2000)	3
Chaetodermatida	1	23	NA	NA	–	–	–
Chionoecetes opilio	0	2	I	S	–	Borja et al. (2000)	3
Chlamys islandica	0	1	I	S	–	Borja et al. (2000)	3
Chone sp	12	0	II	S	–	Borja et al. (2000)	2
Ciliatocardium ciliatum	0	1	I	S	–	Borja et al. (2000)	3
Cirripedia	3	0	II	S	–	Borja et al. (2000)	3
Cistenides granulata	0	50	II	S	–	Borja et al. (2000)	3
Cossura longocirrata	0	6	IV	T	OP	Borja et al. (2000)	3
Crassicorophium bonellii	1	2	III	T	–	Borja et al. (2000)	3
Crenella decussata	0	10	I	S	–	Borja et al. (2000)	3
Cumacea	0	3	I	S	–	Borja et al. (2000)	3
Cyclocardia borealis	0	5	I	S	–	Borja et al. (2000) (C. thouarsii)	2
Cyrtodaria siliqua	0	1	I	S	–	Gilkinson et al. (2005)	2
Diastylis rathkei	32	12	III	T	–	Borja et al. (2000)	3
Diastylis sculpta	13	28	II	S	–	Borja et al. (2000)	3

	Density		Ecological groups				
Taxon name	**Shallow**	**Deep**	**AMBI**	**BENTIX**	**BOPA**	**References**	**CS**
Diastylis sp	0	1	I	S	–	Borja et al. (2000)	1
Echinarachnius parma	13	61	I	S	–	Borja et al. (2000) (Echinoidea)	2
Edotia montosa	5	0	II	S	–	Borja et al. (2000)	3
Ennucula tenuis	0	222	II	S	–	Borja et al. (2000)	3
Eteone sp	0	6	III	T	OP	Borja et al. (2000)	2
Euchone sp	32	0	II	S	–	Borja et al. (2000)	2
Eudorella emarginata	0	9	II	S	–	Borja et al. (2000)	3
Eudorellopsis integra	66	1092	II	S	–	Tillin & Tyler-Walters (2014) (group of Bathyporeia elegans and E. deformis)	2
Euspira pallida	0	1	II	S	–	Borja et al. (2000)	3
Glycera capitata	1	1	II	S	–	Borja et al. (2000)	3
Glycera sp	8	4	II	S	–	Borja et al. (2000)	2
Goniada maculata	1	101	II	S	–	Borja et al. (2000)	3
Guernea (Prinassus) nordenskioldi	0	19	III	T	–	de la Ossa Carretero et al. (2011) (Dexamene spinosa)	1
Halacaridae	0	1	I	S	–	Borja et al. (2000)	2
Haminella solitaria	0	1	II	S	–	Borja et al. (2000)	3
Hardametopa carinata	1	0	II	S	–	Borja et al. (2000) (Stenothoidae)	1
Harmothoe sp	0	3	II	S	–	Borja et al. (2000)	2
Harpacticoida	105	117	NA	NA	–	–	–
Hediste diversicolor	18	19	III	T	OP	Borja et al. (2000)	3
Heteranomia squamula	0	4	I	S	–	Borja et al. (2000)	3
Hiatella arctica	1	5	I	S	–	Borja et al. (2000)	3
Holothuroidea	1	5	I	S	–	Borja et al. (2000)	3
Idotea phosphorea	13	0	II	S	–	Borja et al. (2000) (Idotea sp)	2
Ischyroceridae	0	1	II	S	–	Borja et al. (2000) (Ischyrocerus anguipes)	2
Ischyrocerus anguipes	138	0	II	S	–	Borja et al. (2000)	3
Isopoda	0	1	NA	NA	–	–	–
Lacuna vincta	8	0	II	S	–	Borja et al. (2000)	3
Lamprops fuscatus	7	16	I	S	–	Borja et al. (2000)	3
Lepeta caeca	1	10	I	S	–	Borja et al. (2000)	3
Leucon (Leucon) nasicoides	2	406	II	S	–	Borja et al. (2000)	3
Littorina littorea	9	1	II	S	–	Borja et al. (2000)	3
Lumbrineridae	7	7	II	S	–	Borja et al. (2000)	2
Lysianassidae	3	1	I	S	SA	Borja et al. (2000)	2
Macoma calcarea	168	407	II	S	–	Borja et al. (2000)	3
Maera danae	1	0	I	S	SA	Borja et al. (2000) (Maera sp)	2

Taxon name	Density		Ecological groups				
	Shallow	Deep	AMBI	BENTIX	BOPA	References	CS
Maldane sarsi	0	3	II	S	–	Borja et al. (2000)	3
Maldanidae	4	185	I	S	–	Borja et al. (2000)	2
Micronephthys neotena	741	1228	II	S	–	Borja et al. (2000)	3
Monoculopsis longicornis	0	17	II	S	–	Borja et al. (2000)	3
Musculus discors	0	1	I	S	–	Borja et al. (2000)	3
Mytilus sp	128	10	III	T	–	Borja et al. (2000)	2
Nematoda	271	773	III	T	–	Borja et al. (2000)	1
Nemertea	0	16	III	T	–	Borja et al. (2000)	1
Neoleanira tetragona	1	0	II	S	–	Borja et al. (2000)	3
Nephtyidae	5	11	II	S	–	Borja et al. (2000)	2
Nephtys caeca	4	7	II	S	–	Borja et al. (2000)	3
Nephtys incisa	0	70	II	S	–	Borja et al. (2000)	3
Nephtys sp	0	2	II	S	–	Borja et al. (2000)	2
Nuculana minuta	0	13	I	S	–	Borja et al. (2000)	3
Nymphonidae	0	1	NA	NA	–	–	–
Oenopota sp	2	8	I	S	–	Borja et al. (2000)	2
Oligochaeta	53	89	V	T	–	Borja et al. (2000)	1
Ophelia limacina	0	6	I	S	–	Borja et al. (2000)	3
Opheliidae	0	3	I	S	–	Borja et al. (2000) (Ophelialimacina)	2
Ophiopholis aculeata	0	2	II	S	–	Borja et al. (2000)	3
Ophiura robusta	0	219	II	S	–	Borja et al. (2000)	3
Orchomenella minuta	1	2	II	S	–	Borja et al. (2000)	3
Ostracoda	2	331	I	S	–	Bodegart et al. (1997), Ruiz et al. (2005), Gooday et al. (2009)	1
Pagurus pubescens	0	1	II	S	–	Borja et al. (2000)	2
Pagurus sp	0	4	II	S	–	Borja et al. (2000)	3
Pandalus montagui	0	1	II	S	–	Borja et al. (2000)	3
Parathyasira equalis	0	2	III	T	–	Borja et al. (2000)	3
Parvicardium pinnulatum	1	5	I	S	–	Borja et al. (2000)	3
Periploma leanum	0	5	II	S	–	Borja et al. (2000) (P. discus)	2
Retusophiline lima	0	1	II	S	–	Borja et al. (2000)	3
Philomedes sp	0	5	II	S	–	Borja et al. (2000)	3
Pholoe longa	13	0	II	S	–	Borja et al. (2000) (Pholoe sp)	2
Pholoe sp	33	161	II	S	–	Borja et al. (2000)	2
Phoxocephalus holbolli	48	25	I	S	SA	Borja et al. (2000)	3
Polynoidae	17	61	II	S	–	Borja et al. (2000) (Polynoidae)	2
Pontogeneia inermis	6	0	II	S	–	Borja et al. (2000) (P. rostrata)	2
Pontoporeia femorata	152	164	I	S	SA	Borja et al. (2000)	3

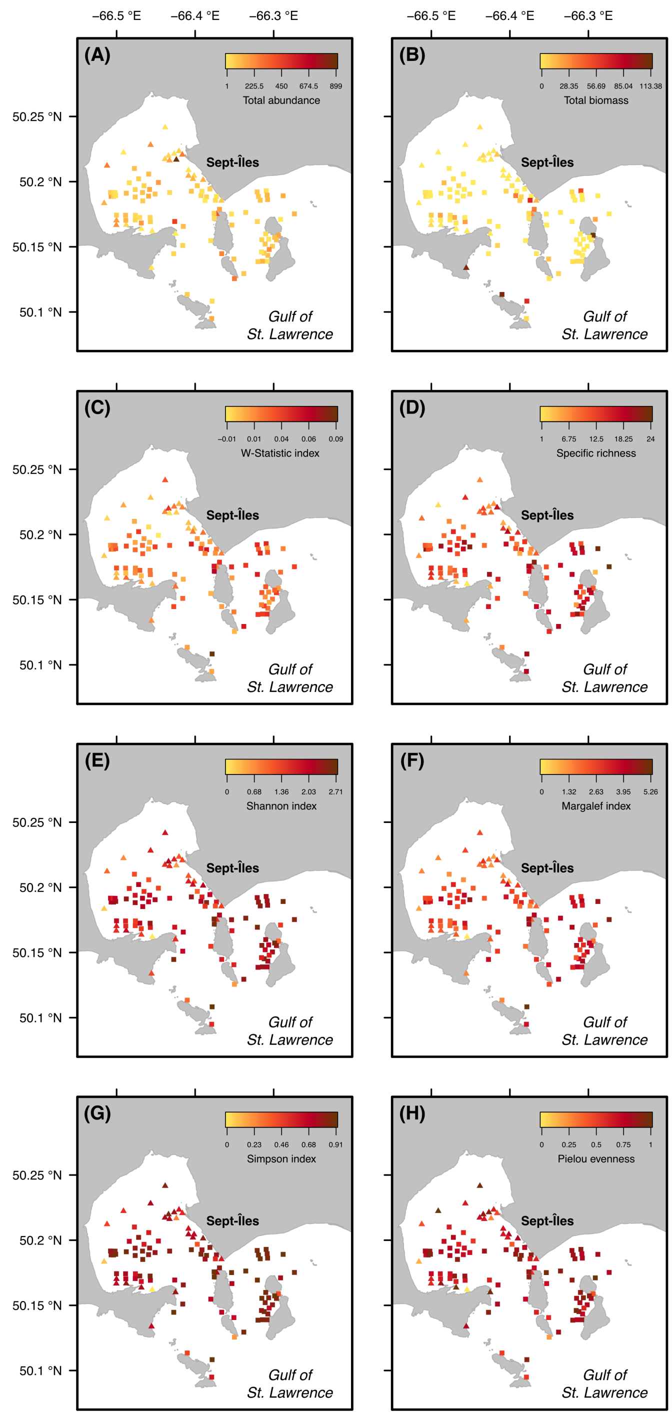
	Density		Ecological groups				
Taxon name	**Shallow**	**Deep**	**AMBI**	**BENTIX**	**BOPA**	**References**	**CS**
Praxillella praetermissa	1	41	III	T	OP	Borja et al. (2000)	3
Propebela turricula	0	5	I	S	–	Borja et al. (2000)	3
Protomedeia fasciata	6	38	II	S	–	Borja et al. (2000)	3
Protomedeia grandimana	854	238	II	S	–	Borja et al. (2000)	3
Puncturella noachina	0	4	I	S	–	Borja et al. (2000)	3
Quasimelita formosa	5	97	I	S	SA	Borja et al. (2000) (Melitidae)	2
Quasimelita quadrispinosa	0	2	I	S	SA	Borja et al. (2000)	3
Retusa obtusa	6	3	II	S	–	Borja et al. (2000)	3
Sabellidae	191	10	I	S	–	Borja et al. (2000)	2
Scoletoma fragilis	14	17	II	S	–	Borja et al. (2000)	3
Scoletoma sp	0	2	II	S	–	Borja et al. (2000)	2
Scoloplos	0	1	I	S	–	Borja et al. (2000)	2
Serripes groenlandicus	0	2	I	S	–	Borja et al. (2000)	3
Sipuncula	0	29	I	S	–	Borja et al. (2000)	1
Solamen glandula	0	1	II	S	–	Borja et al. (2000) (S. columbianum)	2
Solariella sp	0	19	I	S	–	Borja et al. (2000)	2
Strongylocentrotus sp	0	24	I	S	–	Borja et al. (2000) (S. droebachiensis)	3
Tachyrhynchus erosus	0	1	I	S	–	Borja et al. (2000) (Turritella sp)	2
Thracia septentrionalis	35	37	I	S	–	Borja et al. (2000)	3
Thyasira gouldi	0	142	I	S	–	Borja et al. (2000)	3
Thyasira sp	0	11	II	S	–	Borja et al. (2000)	1
Trichotropis bicarinata	0	1	II	S	–	Borja et al. (2000) (Euspira sp)	2
Turritellopsis stimpsoni	0	4	I	S	–	Borja et al. (2000) (Turritella sp)	2
Yoldia myalis	0	3	I	S	–	Borja et al. (2000) (Y. limatula)	2

Table 12 – Reference conditions used for the calculation of Multivariate AZTI Marine Biotic Index (M-AMBI), for shallow and deep stations. S = taxa richness, H = Shannon index, AMBI = AZTI Marine Biotic Index.

	'Bad' status	**'High' status**
Shallow stations		
S	2.5	16.25
H'	0.35	2.01
AMBI	2.88	0.72
Deep stations		
S	7	21.95
H'	1.14	2.54
AMBI	2.43	0.78

Table 13 – Mean (standard error) and extremum values of habitat parameters sampled (organic matter, grain-size classes) and modelled (heavy metal concentrations), calculated for shallow (< 15 m) and deep (> 15 m) stations. SE = standard error.

	Shallow stations (n = 26)			**Deep stations (n = 82)**		
Variable	**Mean (SE)**	**Min**	**Max**	**Mean (SE)**	**Min**	**Max**
Organic matter (%)	1.09 (0.2)	0.25	4.42	1.69 (0.1)	0.23	3.71
Gravel (%)	1.8 (0.7)	0	12.4	3.7 (1.4)	0	76.5
Sand (%)	56.4 (5.1)	0.8	96.8	47.8 (2.4)	0	97
Silt (%)	39.4 (4.8)	3.2	86.9	46 (2.6)	0	84.5
Clay (%)	2.4 (1.7)	0	37.1	2.5 (1.6)	0	97.9
Arsenic ($mg.kg^{-1}$)	3.48 (0.3)	1.5	7.6	3.84 (0.3)	1.4	16
Cadmium ($mg.kg^{-1}$)	0.15 (0)	0.08	0.23	0.13 (0)	0.06	0.19
Chromium ($mg.kg^{-1}$)	60.89 (3.8)	28	110.7	55 (1.7)	27.8	86.5
Copper ($mg.kg^{-1}$)	12.93 (1.2)	2.9	28.7	11.3 (0.5)	3.6	21.4
Iron ($g.kg^{-1}$)	53.55 (2.5)	32.65	78.47	53.87 (1.8)	28.36	151.23
Manganese ($g.kg^{-1}$)	1.22 (0.1)	0.46	3.44	1.1 (0.1)	0.42	2.96
Mercury ($mg.kg^{-1}$)	0.02 (0)	0	0.05	0.02 (0)	0	0.09
Lead ($mg.kg^{-1}$)	5.8 (0.4)	2.4	12.1	5.12 (0.2)	1.7	9.3
Zinc ($mg.kg^{-1}$)	63.79 (4.3)	33.5	141	57.74 (1.7)	27.6	93.9


−66.5 °E
−66.4 °E
−66.3 °E
50.25 °N
50.2 °N
50.15 °N
50.1 °N
(A)
1 225.5 450 674.5 899
Total abundance
(B)
0 28.35 56.69 85.04 113.38
Total biomass
(C)
−0.01 0.01 0.04 0.06 0.09
W-Statistic index
(D)
1 6.75 12.5 18.25 24
Specific richness
(E)
0 0.68 1.36 2.03 2.71
Shannon index
(F)
0 1.32 2.63 3.95 5.26
Margalef index
(G)
0 0.23 0.46 0.68 0.91
Simpson index
(H)
0 0.25 0.5 0.75 1
Pielou evenness
Sept-Îles
Gulf of
St. Lawrence

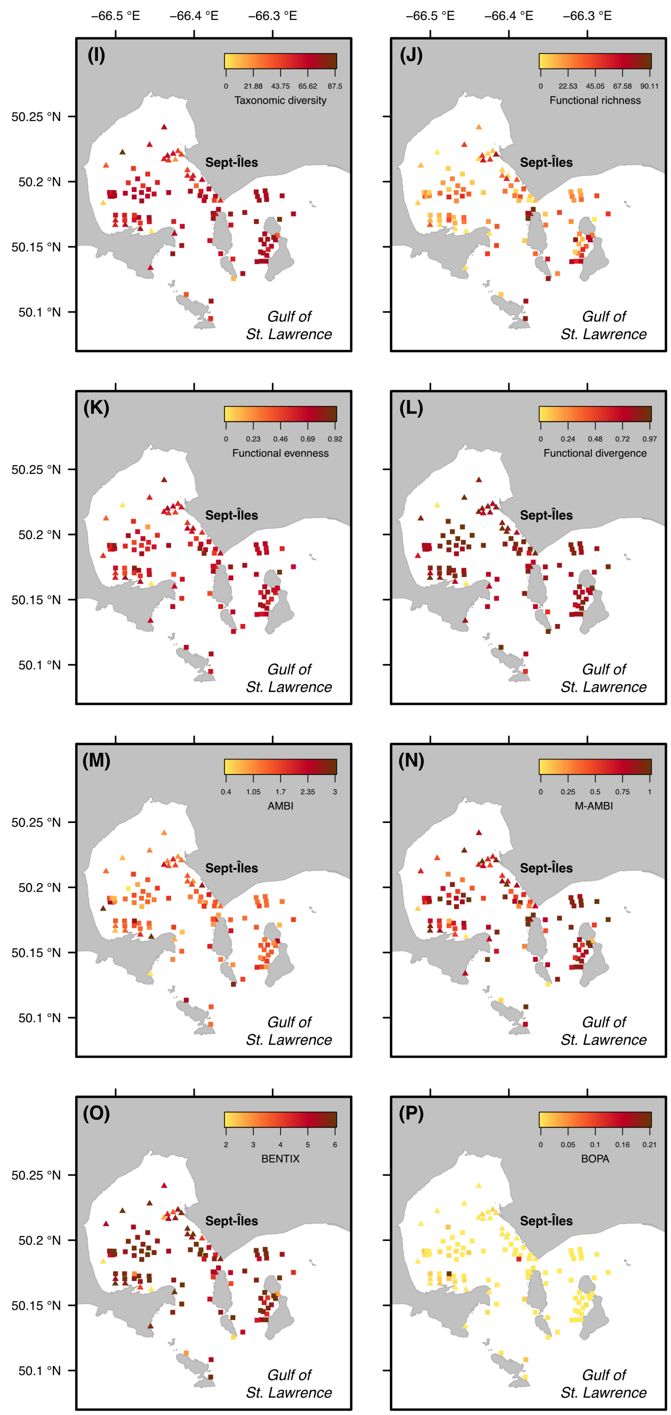


Figure 16 – Values of environmental indicators at each sampled station. Triangles and squares indicate shallow (< 15 m) and deep (> 15 m) stations, respectively. (A) total density, (B) total biomass, (C) W-Statistic index, (D) taxa richness, (E) Shannon index, (F) Margalef index, (G) Simpson index, (H) Pielou evenness, (I) taxonomic diversity, (J) functional richness, (K) functional evenness, (L) functional divergence, (M) AMBI score, (N) M-AMBI score, (O) BENTIX score, (P) BOPA score.

ARTICLE 3
DESCRIPTION DE L'EXPOSITION DES ÉCOSYSTÈMES BENTHIQUES CÔTIERS AUX MULTIPLES ACTIVITÉS HUMAINES À L'ÉCHELLE LOCALE

3.1 Résumé

L'influence anthropique est un phénomène affectant tous les écosystèmes marins de la planète, dont la majorité est influencée par de multiples activités humaines. L'évaluation des impacts cumulés permet de comprendre comment les communautés et les habitats peuvent être affectés par des stresseurs d'origine anthropique, notamment grâce à l'étude de l'exposition et de la vulnérabilité des écosystèmes. De telles évaluations ont été développées à des échelles régionale ou mondiale afin de détecter des tendances à large échelle, et il peut être intéressant d'appliquer ces méthodes à fine résolution spatiale pour améliorer la gestion environnementale locale. Cette étude s'est intéressée à développer et appliquer un indice pour calculer l'exposition locale des écosystèmes à de multiples activités humaines. Les écosystèmes benthiques côtiers de la région de Sept-Îles (Québec, Canada) ont été sélectionnés, une zone portuaire industrielle dans le golfe du Saint-Laurent où de nombreuses activités humaines sont présentes. L'exposition a été calculée grâce à un modèle de diffusion particulaire et aux évènements de pêche dans la région, et les activités considérées ont été l'aquaculture, l'influence de la ville, l'influence des industries, le dragage des sédiments, la navigation commerciale, les égouts et les pêcheries. Une faible exposition a été détectée à l'échelle de la baie, avec des zones d'exposition cumulée devant la ville et les zones industrielles. Les modèles joints de distribution d'espèces ont détecté des relations significatives entre l'assemblage macrobenthique et des prédicteurs tels que les paramètres abiotiques

et les indices d'exposition, permettant ainsi de rendre compte de la structure des communautés selon différents scénarios anthropiques. Cette étude présente des résultats intéressants sur les liens entre les activités humaines et les communautés benthiques à l'échelle locale, ouvrant la voie à des évaluations environnementales plus holistiques.

L'article associé à ce chapitre, "*Describing exposure from multiple human activities on coastal benthic ecosystems at the local scale*", a été co-rédigé avec Rémi M Daigle, David Beauchesne, Julie Carrière, Christopher W. McKindsey et Philippe Archambault. Il est prévu de le soumettre dans la revue *PLoS ONE* au courant de l'année 2021. J'ai établi les objectifs de ce chapitre avec Philippe Archambault. J'ai développé l'indice d'exposition avec Rémi M Daigle et David Beauchesne. Je me suis basé sur les données obtenues lors de la campagne d'échantillonnage en 2017 effectuée pour le premier chapitre, en collaboration avec Julie Carrière, pour étudier les liens entre communautés benthiques et indices d'exposition. J'ai dirigé la rédaction de l'article, où l'ensemble des co-auteurs a contribué à l'interprétation des résultats en fonction de leur expertise et à la révision générale. Certains résultats issus de ce chapitre ont été présentés lors de la Réunion Scientifique du *Canadian Healthy Oceans Network* II à Ottawa en novembre 2018, la *Global Change on Estuarine and Coastal Ecosystems Conference* (CHEERS) à Bordeaux en novembre 2019 et la Réunion Scientifique Annuelle de Québec-Océan à Québec en mars 2020.

Dreujou, E., Daigle, RM., Beauchesne, D., Carrière, J., McKindsey, CW., Archambault, P. (in prep). Describing exposure from multiple human activities on coastal benthic ecosystems at the local scale.

Les sections suivantes sont celles de l'article en préparation.

DESCRIBING EXPOSURE FROM MULTIPLE HUMAN ACTIVITIES ON COASTAL BENTHIC ECOSYSTEMS AT THE LOCAL SCALE

3.2 Abstract

Anthropogenic influence is a worldspread phenomenon affecting marine ecosystems, the majority of which is influenced by multiple human activities. Assessment of cumulative impacts provide information on how communities and habitats may be changed by anthropogenic stressors, through the study of exposure and vulnerability of ecosystems. Such assessments have been developed at a regional or a global scale to detect large-scale trends, and there is a need to operationalize their use for fine-scale applications so that local environmental management may be enhanced. The objective of this study was to develop and apply an index to determine the exposure of benthic coastal ecosystems to multiple local human activities. Coastal benthic ecosystems in the region of Sept-Îles (Québec, Canada) were selected, a major industrial harbour area in the Gulf of St. Lawrence where many human activities co-occur. Exposure was calculated using a particle diffusion model and fishing events data in the region, and the activities considered were aquaculture operations, city influence, industry influence, sediment dredging, commercial shipping, sewers and fisheries. A generally low exposure was obtained at the bay-scale, with areas of cumulative exposure in front of the city and industrial areas. Joint species distribution models detected significant relationships between the macrobenthic assemblage and predictors such as abiotic parameters and exposure indices, which will be useful to predict the structure of the communities under different anthropogenic scenarios. This study presents valuable results on the links between multiple human activities and benthic communities at the local scale, paving the way towards more holistic environmental assessments.

Keywords: anthropogenic exposure, multiple activities, coastal benthos, macrofauna, Gulf of St. Lawrence

3.3 Introduction

In order to guide protection and conservation initiatives, management of coastal ecosystems requires efficient monitoring of ecosystem components, including human activities. Considering the widespread anthropogenic influence on marine ecosystems (Halpern et al., 2019), environmental assessments need to consider emerging trends, such as cumulative impacts, where multiple activities co-occur so that descriptions of current and future patterns in the ecosystem are accurate (Crain et al., 2008; Brown et al., 2014; Côté et al., 2016).

Non-additive effects, such as antagonistic effects (*i.e.* perceived effect lower than the sum of individual effects) or synergistic effects (*i.e.* greater effect than the sum of individual effects), have been documented in a variety of ecosystems using *in situ* observations or experimental setups (Séguin et al., 2014; Piggott et al., 2015; Galic et al., 2018; Hodgson et al., 2019; Carrier-Belleau et al., 2021). However, these effects are complex to study as each interaction between effects induced by activities is unique, resulting in a multitude of potential interactions between activities that are typically time- and location-specific (Crain et al., 2008; Darling and Côté, 2008). Describing all interactions between human activities increases exponentially with the number of activities considered (Côté et al., 2016): as an example, considering 10 human activities yields 45 two-way interactions and around 1,000 three-way and more interactions.

Assessing, managing and monitoring the effects of multiple human activities in coastal ecosystems require integrative and holistic methods, such as ecosystem-based management or marine spatial planning (Margules and Pressey, 2000; Link, 2002; Pikitch et al., 2004; Levin et al., 2009; Santos et al., 2019; Dreujou et al., 2020a). These frameworks consider ecosystems as networks with a complex structure by including multiple ecological components (*e.g.* biological communities, habitats, human activities) along with various stakeholders (from scientists to decision-makers and industries). Their implementation relies on ecological groundwork and dedicated tools, such as indicators of ecological status, so that ecosystems are properly assessed according to local conditions and specific objectives (Pinto et al., 2009; Borja et al., 2012; Teixeira et al., 2016).

As an example, Halpern et al. developed an impact score from multiple stressors, updated in 2015 and 2019 (Halpern et al., 2008, 2015, 2019). This score is calculated by assessing exposure and vulnerability of ecosystems to human activities. Exposure corresponds to the susceptibility of ecosystems to be impacted by a perturbation, measured for example by describing co-occurence of human activities and ecosystem components, and vulnerability is a parameter describing how ecosystem components react to this perturbation (Wilson et al., 2005; Halpern et al., 2007). Such impact scores were measured for 17 human activities leading to an integrative cumulative impact score obtained by adding individual scores (Halpern et al., 2008). The results showed that stress on marine ecosystems from human activities is ubiquitous and that few regions are exempt of anthropogenic influence, which provides a major breakthrough to characterize cumulative effects globally. Such methods provide valuable information on the effects of multiple human activities in a context of ecosystem management, and their application in different ecosystems and at multiple spatial scales would greatly increase their value. To our knowledge, this score has been computed at finer spatial resolutions, such as the regional-scale (Ban and Alder, 2008; Beauchesne et al., 2020a), but it was not yet applied to local-scale ecosystems (*e.g.* a harbour or an estuary under anthropogenic influence). There is thus a need to understand how such methods may be used to evaluate human activity influence on ecosystems locally (*e.g.* < 100 km).

In this study, we focused on coastal ecosystems and on macrobenthic invertebrates, one of the most diverse biological communities whose links with human activities have been described in a variety of ecosystems (Pearson and Rosenberg, 1978; Grall and Glémarec, 1997; Teixeira et al., 2016). Many of these species are characterized by a sedentary lifestyle and a relatively long life span which tends to reflect medium-term conditions, resulting in adaptation or local extinction when perturbated (*e.g.* Dauer, 1993; Borja et al., 2000; Wei et al., 2020). Because characterization of vulnerability requires extensive data on physiological responses of species and how *influence* translates to *impact*, here we focused on the exposure of benthic communities to human activities (such as variable $S_{j,x}$ in the score by Halpern et al. (2019), representing intensity of each stressor at a grid cell) to provide an operational tool to study human activities (individual and cumulative exposures) in a local context.

The industrial harbour area of Sept-Îles (Québec, Canada) was used as a study case. Located in the Gulf of St. Lawrence, one of the management areas designated by Fisheries and Oceans Canada and a major economic region for Québec (Department of Fisheries and Oceans, 2009; Beauchesne et al., 2016; Daigle et al., 2017; Schloss et al., 2017), Sept-Îles is the fourth largest Canadian port in terms of total exchanged goods and the second largest in Québec as of 2019 (Statistics Canada, 2011; Binkley, 2020; Port de Sept-Îles, 2020; Ferrario and Archambault, in press). Available ecological data on ecosystems in this region are limited, which justified the initiation of many ecological projects to characterize benthic ecosystems and their relation to coastal human activities (Canadian Healthy Oceans Network, 2016; Carrière, 2018; Dreujou et al., 2020b, 2021).

The specific objectives of this study are to (i) model the exposure of benthic ecosystems to human activities locally and (ii) assess community structure as a function of individual and cumulative anthropogenic exposure. We expect that the structure of communities within high cumulative exposure areas (‘anthropogenic hotspots’) will present evidence of perturbation (such as lower diversity and presence of opportunistic species) compared to the rest of the study area.

3.4 Methods

3.4.1 Study area

Studied ecosystems are located on the northern shore of the Gulf of St. Lawrence, within the industrial harbour area of Sept-Îles (Figure 17). This region includes the Baie des Sept Îles and the archipelago at its entrance, covering around 200 km^2 (Figure 17). Bathymetry is shallow within the bay, with a maximum depth of 50 m at its entrance, then becoming deeper in the archipelago (as deep as 200 m) (Dutil et al., 2012). The general sediment profile is sandy-silty. Benthic communities are diverse with a high density of annelids, arthropods and molluscs (Dreujou et al., 2020b).

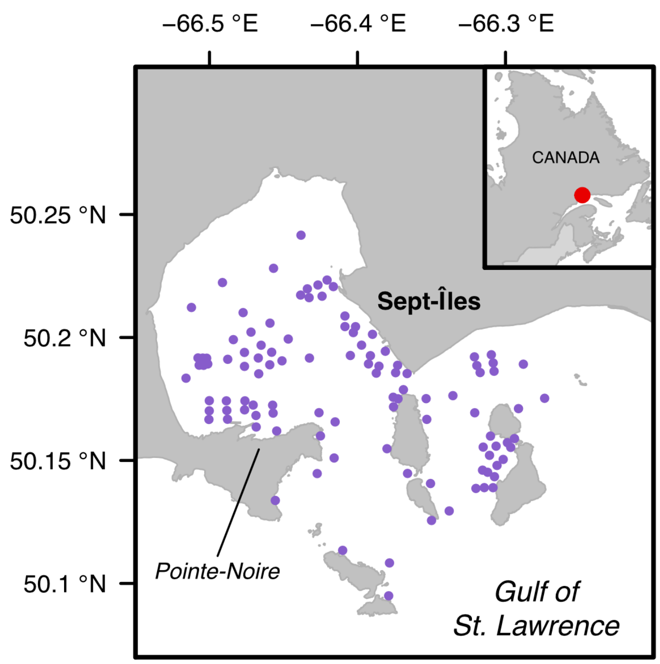


Figure 17 – Map of the study area with the location of the sampled stations.

The main industrial sectors in the area are aluminium production in plants at the Pointe-Noire sector and the south-eastern part of the city of Sept-Îles, international shipping of iron ore through bulk carriers (reaching 29.3 MT in 2019) and coastal fisheries targeting fishes (*Clupea harengus*, *Gadus morhua*), arthropods (*Chionoecetes opilio*, *Cancer irroratus*, *Pandalus borealis*) and molluscs (*Buccinum* sp, *Mactromeris polynyma*) (Department of Fisheries and Oceans, 2019; Port de Sept-Îles, 2020).

This region has sub-Arctic environmental conditions, with sea ice formation in November/December and important freshwater run-off due to snowmelt in April (Demers et al., 2018). A complete circulation model in the Baie des Sept-Îles is not available, but Shaw (2019) identified strong tidal currents, resulting in an overall estuarine circulation throughout the bay (*i.e.* an inflow from a bottom layer and an outflow from a top layer). Average currents at the entrance of the bay (located between the city Sept-Îles and the Pointe-Noire terminal) were as high as 14.4 $cm.s^{-1}$, and many streams are discharging in the eastern and northern sections (Shaw, 2019).

3.4.2 Exposure to human activities

Human activities occurring in the area were identified by a compilation of data from local organizations (Port de Sept-Îles, Ville de Sept-Îles and Institut Nordique de Recherche en Environnement et en Santé au Travail) and by referring to databases such as Halpern et al. (2019) and Beauchesne et al. (2020b). This initial review resulted in the consideration of seven human activities: aquaculture (a mussel farm in the archipelago), city influence (coastline linked to the city), industry influence (coastline linked to industries), sediment dredging (collecting and dumping sites), commercial shipping (mooring sites), sewers (city and industry drains) and fisheries. Spatial layers were produced with R v4.0 and packages 'raster' and 'sf' (Pebesma, 2018; Hijmans, 2020; R Core Team, 2020).

In order to analyze how human activities influence benthic ecosystems, we computed an index of exposure E for each activity. Because a complete circulation model is not available and to reduce complexity of coastal ecosystems, we considered a static environment (without spatial or temporal dynamics) so that the index will generate a 'snapshot' of the ecosystems. The index E has been computed differently for land/sea-based activities and for fisheries, then standardized between 0 (low exposure) and 1 (high) scale to allow comparisons. A cumulative exposure index was obtained by summing the seven indices.

3.4.2.1 Land/sea-based human activities

Indices of exposure for land/sea-based activities were obtained using a model of theoretical particle diffusion. Particles are the resultant of an activity, which can model many types of anthropogenic influence, such as chemical, physical or biological. These particles are introduced in the ecosystem by a point source from the surface, and we considered longitudinal and latitudinal movement (2D environment). An identical number of particles are released by each activity and we focused on their journey within the bay from the source(s) of activity. The number of particles linearly decreases with the distance from the source so that, for a certain location, if the length of the journey is small then particle density will be high, indicating a high exposure to the activity, and vice versa.

Because each human activity is not related to the same environmental components, we modelled exposure as a combination of diffusion patterns from different particle types (to reduce complexity, density was considered equivalent for particles within a same class). Three theoretical particle archetypes have been considered, considering biological and abiotic elements of the ecosystem (Table 13).

Table 13 – Types of theoretical particles considered in the diffusion model.

Particle type	**Size range**	**Examples**
High diffusion	<4 μm	clay, dissolved organic matter, small bacteria, viruses, chemical components, proteins
Medium diffusion	4 μm - 2 mm	sand, silt, small particulate organic matter large bacteria, small dead organisms
Low diffusion	>2 mm	gravel, large particulate organic matter large dead organisms

We identified 11 sources of human activity in our study area, from punctual sources, *e.g.* sewers drains, to diffused sources, *e.g.* runoff from city (Table 14, Figure 18), that will act as sources of particles in the diffusion model. Each source has been modelled as a combination of the three particle types, using proportion ratios p (sum of these three ratios equals one) (Table 14) obtained by expert opinion. A standardized survey was sent out to a number of researchers in January 2020, whose results were averaged to obtain p for each activity and compared to dedicated studies (such as Callier et al. (2008) for aquaculture).

Table 14 – Human activities and sources considered in this study, along with the proportion of theoretical particles considered in the diffusion model (when applicable). Proportions of high, medium and low diffusion particles corresponds to ratios assigned to parameter p in the calculation of indices of exposure E.

		Diffusion behaviour		
Human activity	**Identified sources**	**High**	**Medium**	**Low**
Aquaculture	Mussel farm	0.15	0.8	0.05
City influence	Water runoff	0.15	0.1	0.75
	Wharves and marina	0.45	0.45	0.1
Commercial shipping	Mooring sites	0.45	0.45	0.1
	Traffic routes	0.45	0.45	0.1
Fisheries	Bottom-trawls	—	—	—
	Dredges	—	—	—
	Nets	—	—	—
	Traps	—	—	—
Industry influence	Water runoff	0.1	0.3	0.6
	Wharves and docking	0.45	0.45	0.1
Sediment dredging	Collection sites	0.1	0.2	0.7
	Dumping sites	0.15	0.25	0.6
Sewers	Rainwater drains	0.05	0.15	0.8
	Wastewater drains	0.05	0.15	0.8

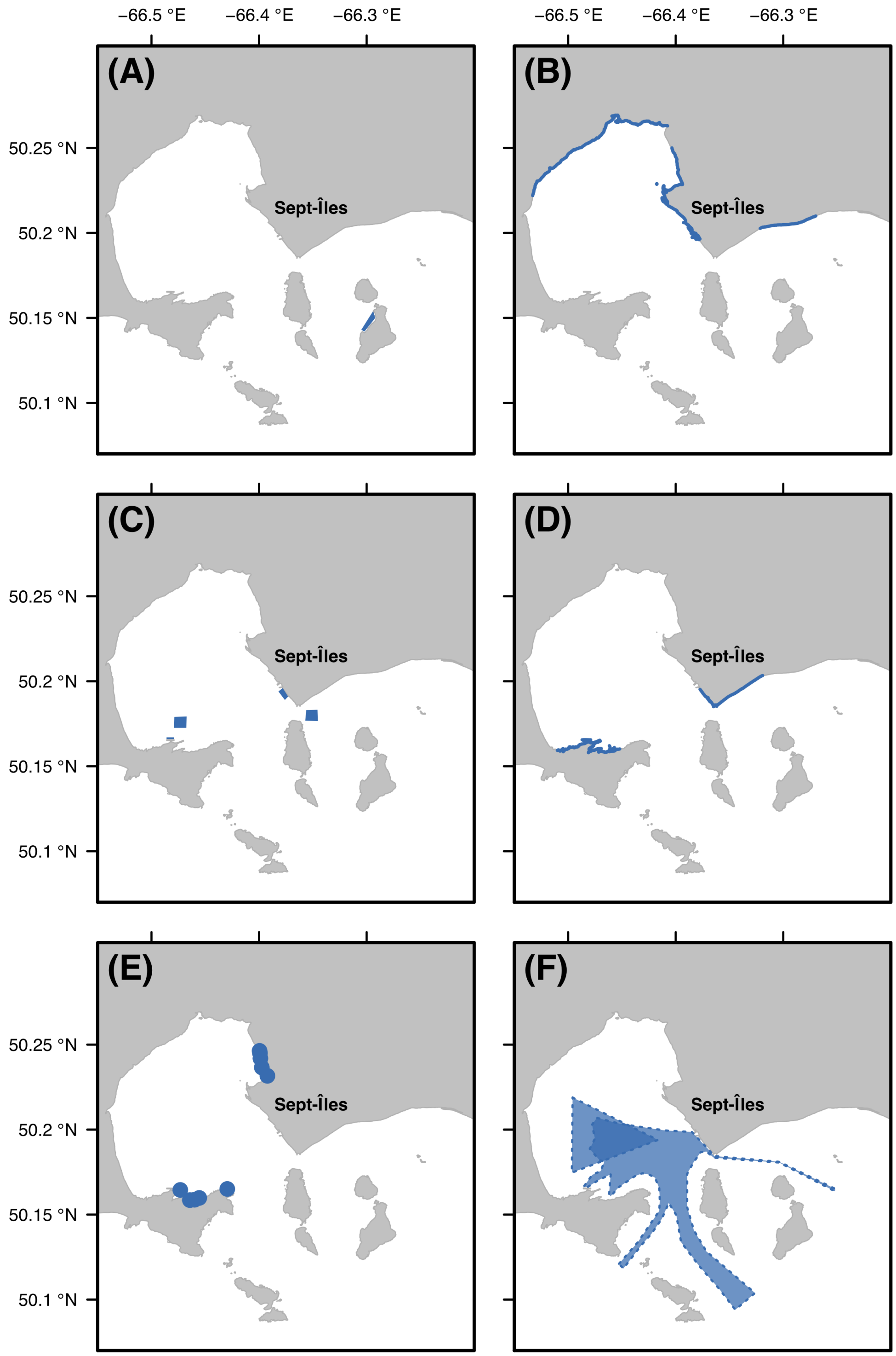


Figure 18 – Map of the considered sources of land/sea based human activities in the study area. (A) Aquaculture exposure, (B) City exposure, (C) Dredging exposure, (D) Industry exposure, (E) Sewers exposure, (F) Shipping exposure.

Diffusion behaviour for each particle type was simulated using package ‘gdistance’ (van Etten, 2017). We created a 100 x 100 m grid raster of the study area, where we established a connectivity matrix in a chess queen configuration (each cell to its eight direct neighbors using horizontal, vertical and diagonal directions). The cost of moving from one cell to another was computed considering two constraints: (i) particles only diffuse in the marine environment, (ii) particles cannot move when they have sedimented. To implement these constraints, we used coasts as boundaries (cost to select land cells is infinite) and a distance threshold (cost to select cells after a certain distance is infinite). The latter is specific for each particle type, initialized using Chamley (1989) and Bach et al. (2012). A least-cost pathfinding algorithm computed the distance D from the source(s) of human activity to a specific cell (Dijkstra, 1959; van Etten, 2017).

These exposure indices E were calculated at each cell using D in an inverse logarithmic relationship, to account for dispersion in a 2D environment while reducing the contribution of the highest values, and the proportion of particle types p for each activity (Table 14):

$$E_{ij} = \sum_{k}^{s,m,l} \left(\frac{1}{ln(D_{ijk} + 1)} . p_{jk} \right)$$

where i is a cell, j is a human activity and k is a particle type.

3.4.2.2 Fisheries

The index of exposure E for fisheries was calculated by considering the number of fishing event by gear type: areas with a high number of events will indicate a high exposure, and vice versa.

Data was extracted from the eDrivers plateform in the industrial harbour area of Sept Îles (Beauchesne et al., 2020b), for events recorded between 2010 and 2015. Fishing events were compiled in a raster file for four types of fishing gear: traps, bottom-trawls, nets and dredges. Because fisheries information was not available every year, we averaged the number of events to obtain a proxy of fishing intensity per gear G.

We obtained the exposure index E by combining G from the four gear types and a weighting parameter f_k, specific to each gear to differenciate their contribution to the exposure index, using the following formula:

$$E_i = \frac{\sum_k \left(G_{ik} . f_k\right)}{4}$$

where i is a cell and k is a gear type.

3.4.2.3 Cumulative exposure

To obtain an index of cumulative exposure C_i, describing the overall influence of the 7 selected activities, we added each individual index E:

$$C_i = \sum_j E_{ij}$$

where j is a human activity. This index thus varies between 0 (low cumulative exposure) and 7 (high cumulative exposure).

3.4.3 Habitat and biological samples

Ecological samples were collected in July 2017. A total of 108 stations were selected using a semi-randomization algorithm, in the bay and the archipelago. Stations were constrained between 0 m and 80 m deep with an increased sampling effort in areas where sources of human activities where present (Figure 17). Station depth was obtained from the navigation sonar, then corrected with respect to tide height at time of sampling. A Ponar grab (0.05 m^2) was deployed at each station from a boat with two independent casts.

The first cast collected sediment in which two samples were acquired for the analyses of organic matter content and sediment grain-size. These samples were stored at -20 °C until processing in the laboratory. The percentage of total organic matter (*i.e.* sum of organic carbon and organic nitrogen) in the sediment was obtained by using the Loss-on-Ignition method (Davies, 1974). Grain-size analysis was done on a sieving column for the fraction with particles larger than 2 mm and with a Laser Diffraction Particle Size Analyser for the smaller fractions. Results from both techniques were combined to yield a unified distribution range from 0.04 μm to 26.5 mm. From this, percentages of gravel, sand, silt and clay were calculated as defined by Wentworth (1922) and Folk (1980).

All the sediment obtained with the second cast was conserved for benthic macrofauna identification, then sieved on a 0.5 mm mesh size. Retained individuals were preserved in a solution of BORAX-buffered formalin (4 %) and these samples were sorted using a stereomicroscope. Taxa were identified to the lowest taxonomic level possible with reference manuals and identification guides, and names were validated with the World Register of Marine Species (WoRMS Editorial Board, 2020). Taxon density and wet biomass were recorded at each station by counting and weighting individuals collected per grab, respectively.

In addition to these parameters, we included heavy metal concentrations in the sediment. Concentrations were calculated based on values obtained in the same area in 2014 and 2016, retrieved from the database of Institut Nordique de Recherche en Environnement et Santé au Travail (Carrière, 2018), using Inverse Distance Weighting interpolation (Dale and Fortin, 2014). We focused on metals for which toxicity criteria have been defined in the Biological Effects Database for Sediments (Environment Canada and Ministère du Développement Durable de l'environnement et des Parcs du Québec, 2007; Centre d'Expertise en Analyse Environnementale du Québec, 2014): arsenic, cadmium, chromium, copper, mercury, lead and zinc; we also added iron and manganese to account for possible contamination from local ore industries.

3.4.4 Statistical analyses

Links between community characteristics – taxa richness, total density of individuals, total biomass, Shannon index (base e logarithm) and Pielou evenness (Magurran and McGill, 2011) – and exposure indices were studied using multiple linear regressions. We added depth as a covariate to account for bathymetric variation between stations. Variables were transformed (logarithm or square root) if the assumptions of normality and homoscedasticity were not respected (Quinn and Keough, 2002). We also explored relationships between the taxa assemblage and exposure indices through non-parametric multivariate regression with distance-based linear modelling (DistLM, 9999 permutations) (McArdle and Anderson, 2001). Statistical analyses were done using R v4.0 (R Core Team, 2020).

The structure of benthic communities has been modelled using Hierarchical Modelling of Species Communities (HMSC), a joint species distribution model method (Ovaskainen et al., 2017). Variables considered for this model were abiotic parameters (only those that were not correlated, *i.e.* organic matter, gravel, silt, clay, arsenic, cadmium, copper and mercury) and indices of exposure as predictors, with presence/absence of taxa collected at each station. HMSC models used a Probit distribution and non-informative priors (Ovaskainen et al., 2017). Monte-Carlo Markov Chains considered 100,000 iterations (1000 used for burning and a thinning parameter of 100). Tjur's pseudo R^2 was calculated to estimate predictive power of the models (Tjur, 2009). We validated the quality of the outcomes by randomly splitting the available data in a training dataset (85 % of the available data, corresponding to 92 stations) and a test dataset (the remaining stations, *i.e.* 16 stations) to compare observed and predicted results. Taxa presence was set with a probability threshold of 0.5, below which the taxon is assumed absent.

3.5 Results

3.5.1 Calculation of the indices of exposure

Overall, exposure values were low, with means varying between 0.04 (aquaculture) and 0.28 (shipping) (Figure 19). Only stations close to sources of activity presented higher values, with a limited area of influence for each human activity. This result is consistent with how indices were calculated, as they considered either distance from the source or occurrence of fishing events as proxies of exposure. Shipping was the activity with the highest number of stations where exposure reached a value above 0.9, which may be explained by the large extent of its source and high contents of high and medium diffusion particles (Figures 18, 19). Dredging, city and industry presented a similar behaviour, where exposure dropped relatively quickly away from sources (Figure 18). The exposure for dredging and sewers showed a large area of low exposure (between 0.1 and 0.2) while highest exposure were very close to sources (Figure 18). Fisheries exposure was quite different from other activities, with highly localized exposure (Figure 18).

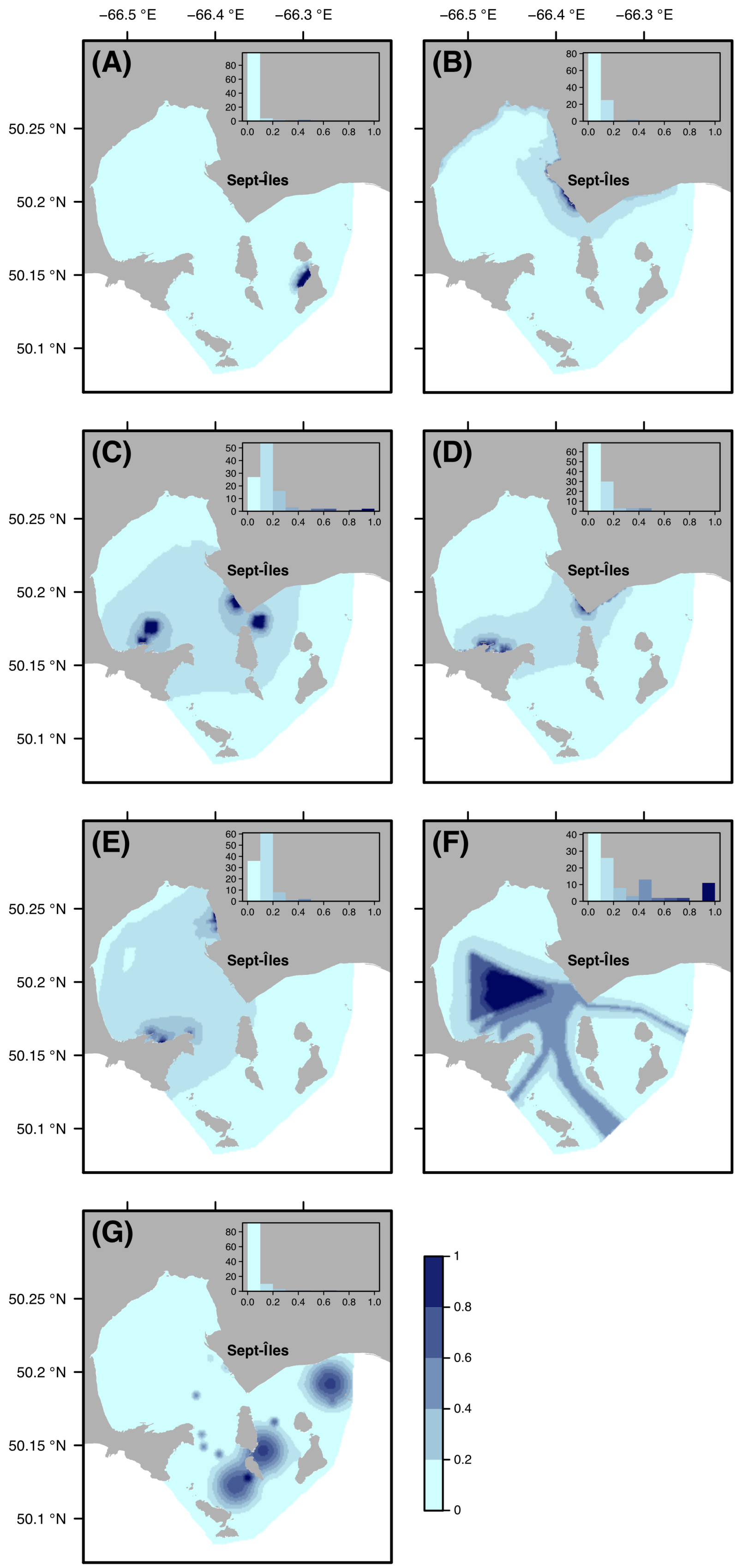

(A)
(B)
(C)
(D)
(E)
(F)
(G)
Sept-Îles
−66.5 °E
−66.4 °E
−66.3 °E
50.25 °N
50.2 °N
50.15 °N
50.1 °N
0.0 0.2 0.4 0.6 0.8 1.0
1
0.8
0.6
0.4
0.2
0

Figure 19 – Indices of exposure calculated in the study area. Histograms represent the number of stations along the value of the index, and colours correspond to exposure classes (light blue = low exposure, dark blue = high exposure). (A) Aquaculture exposure, (B) City exposure, (C) Dredging exposure, (D) Industry exposure, (E) Sewers exposure, (F) Shipping exposure, (G) Fisheries exposure.

The cumulative exposure index at each station varied between 0.184 and 1.955, reaching an average of 0.86 (standard error of 0.04) at the bay-scale (Figure 20). Unsurprisingly, the highest values were detected close to activity sources, especially in front of the city of Sept-Îles and the Pointe-Noire sector (Figure 20).

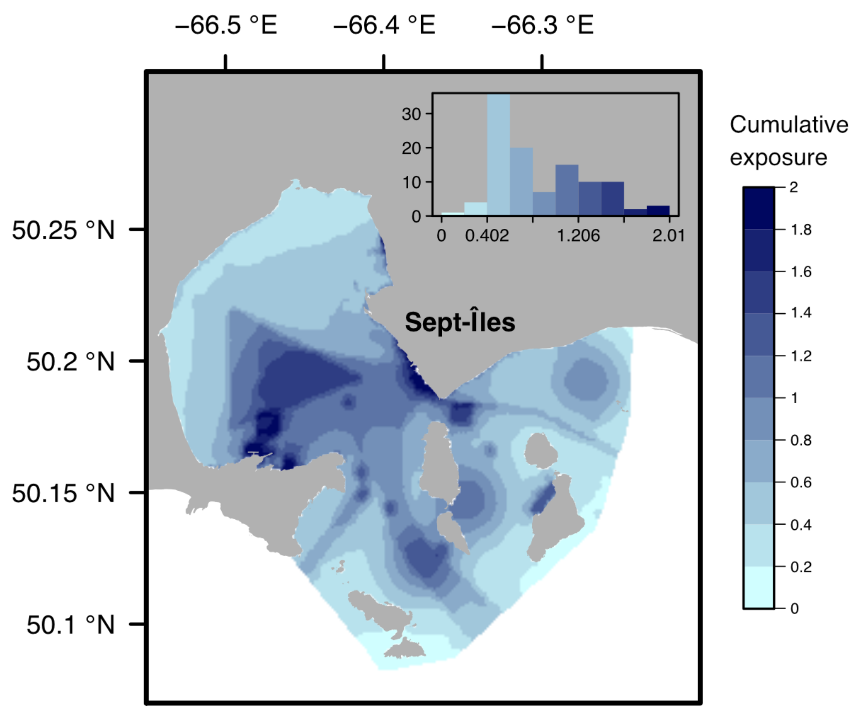


Figure 20 – Indices of cumulative exposure calculated in the study area. Histograms represent the number of stations along the value of the index, and colours correspond to cumulative exposure (light blue = low exposure, dark blue = high exposure).

3.5.2 Relationships with benthic communities

Predictive power of the multiple regressions was the highest for Shannon index (adjusted $R^2 = 0.3$) followed by taxa richness (0.2) and Pielou eveness (0.15), while total density and total biomass reached very low values (both 0.02) (Table 15). Marginal tests for depth reported significant positive relationships with taxa richness, Shannon index and Pielou evenness (Table 15). Four human activities presented similar influences on these variables: aquaculture and city with positive coefficients, industry and sewers with negative coefficients (Table 15). Only sewers presented a significant relationship with total biomass, while the rest of the coefficients seldom exceeded 0.1.

Table 15 – Predictor coefficients (and standard error) from the multiple linear regression models of community characteristics. S = taxa richness, N = total density, B = total biomass, H = Shannon index, J = Pielou evenness, Aq = aquaculture, Ci = city, Dr = dredging, Fi = fisheries, In = industry, Se = sewers, Sh = shipping. Significant p-values of marginal tests on predictors are highlighted in bold.

	Intercept	**Depth**	**Aq**	**Ci**	**Dr**	**Fi**	**In**	**Se**	**Sh**	$R2_{adj}$
S	0.01 (0.09)	0.22 (0.1)	0.13 (0.1)	0.03 (0.1)	- 0.03 (0.11)	0.17 (0.1)	- 0.14 (0.14)	- 0.12 (0.14)	0.11 (0.1)	0.12
p	1	**0.0279**	0.174	0.8152	0.7779	0.0783	0.2895	0.3823	0.255	
N	0.01 (0.1)	- 0.2 (0.11)	- 0.01 (0.11)	0.07 (0.11)	- 0.09 (0.12)	0.09 (0.11)	- 0.18 (0.15)	0.15 (0.15)	- 0.07 (0.11)	0.02
p	1	0.0692	0.9855	0.491	0.4687	0.4271	0.2242	0.34	0.4931	
B	0.01 (0.1)	- 0.19 (0.11)	- 0.16 (0.11)	- 0.2 (0.11)	0.01 (0.12)	- 0.02 (0.11)	0.2 (0.15)	- 0.38 (0.15)	- 0.1 (0.11)	0.02
p	1	0.0981	0.1456	0.0698	0.946	0.8335	0.1875	**0.0154**	0.4138	
H	0.01 (0.08)	0.49 (0.1)	0.1 (0.09)	0.08 (0.09)	0.12 (0.1)	- 0.04 (0.09)	- 0.18 (0.13)	- 0.1 (0.13)	0.03 (0.09)	0.3
p	1	**<0.0001**	0.263	0.3622	0.2372	0.6839	0.1621	0.4616	0.7489	
J	0.01 (0.09)	0.43 (0.1)	0.01 (0.1)	0.09 (0.1)	0.17 (0.11)	- 0.18 (0.1)	- 0.17 (0.14)	- 0.03 (0.14)	- 0.08 (0.1)	0.15
p	1	**<0.0001**	0.9087	0.3299	0.1333	0.0843	0.2159	0.8619	0.4423	

DistLM regression on the taxa assemblage had a R^2 of 0.22, and the result of the ancillary constrained ordination is shown on Figure 21. Two groups of exposure indices with similar influence were obtained: sewers/shipping to one side, and aquaculture/fisheries to the other; city, dredging and industry had lower influence. Interestingly, depth was not correlated to indices, but it was highly correlated to the benthic community structure, as expected. No specific structure may be described based on the similarity between stations, except for a group of stations with a high cumulative exposure which seems have different communities than the rest of the stations (Figure 21). These stations presented high abundances (density or biomass) of the annelid Errantia *Micronephtys neotena*, the cumacean *Eudorellopsis integra* and the bivalve *Macoma calcarea*.

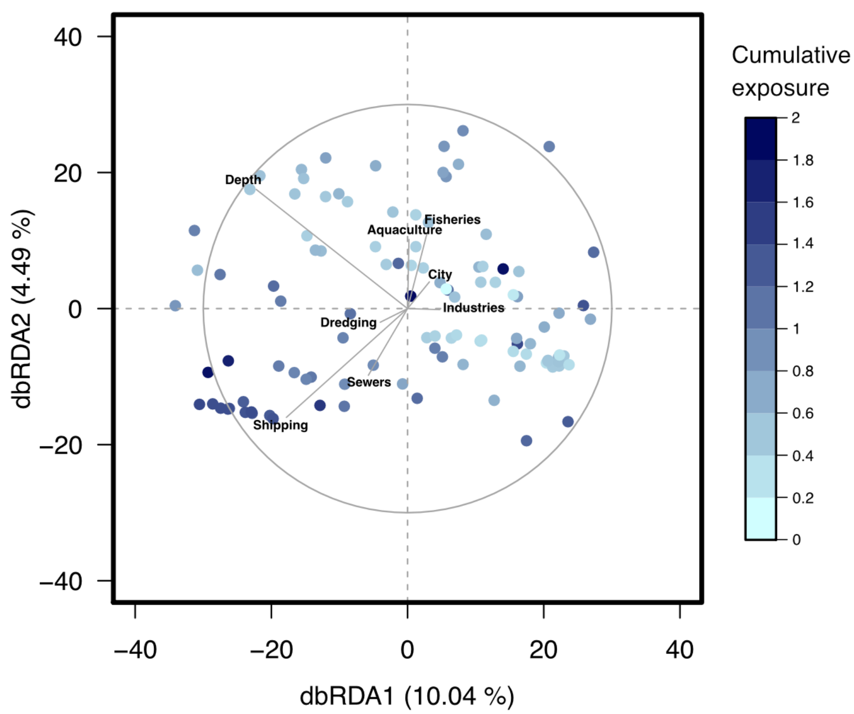


Figure 21 – Constrained ordination with a distance-based Redundancy Analysis on the taxa assemblage. Axes percentages are the proportion of variance explained by the regression model. Colours correspond to the cumulative exposure (light blue = low exposure, dark blue = high exposure).

The probability of taxa presence was calculated using predictive models with HMSC (Table 16). Models for 52 taxa (39.4 % of the total sampled taxa) had a pseudo-R^2 higher or equal to 0.15, the highest being for Nematoda (0.45). Predictor coefficients were variable between models, but some overall trends may be described. A majority of positive coefficients were observed for silt (for 76 % of the taxa), copper (86 %) and sewers (83 %) presented a majority of positive coefficients, while a majority of negative coefficients were obtained for organic matter (76 %), gravel (61 %), clay (87 %), arsenic (91 %), cadmium (87 %), mercury (82 %), aquaculture (67 %), city (80 %), dredging (82 %), industry (93 %), shipping (64 %) and fisheries (67 %). The validation process showed a relatively low number of models correctly predicting taxa presence, the highest being for *M. calcarea*, *M. neotena* and *E. integra*.

3.6 Discussion

We proposed an exposure index to study influence from human activities on coastal benthic ecosystems locally, based on sources of human activity with a particle diffusion model and able to consider multiple activities in a holistic framework. This index can be used in data-poor regions, as only the source of activities and their particle contents are needed to compute an anthropogenic exposure. In particular, this provides an interesting way to model exposure in ecosystems where complete hydrodynamic data on local currents are not available. This allows for application in different ecosystems and at different scales, a huge advantage when studying anthropogenic impacts on marine ecosystems.

When applied to ecosystems in the industrial harbour area of Sept-Îles, we detected some relationships between the exposure indices for seven human activities (aquaculture, city influence, industry influence, sediment dredging, commercial shipping, sewers and fisheries). Few stations have been detected free of cumulative exposure, which reinforces the importance of studying influence of multiple human activities in an integrative way (Côté et al., 2016; Dreujou et al., 2020a). In a management perspective, these areas can be considered anthropogenic 'hotspots', *i.e.* area where multiple activities are influencing ecosystems simultaneously, which is a valuable product to guide environmental protection and sustainable development.

Activities considered in this study have been chosen based on local conditions, and it would be interesting to add other activities (such as tourism cruises, underwater noise or recreational boating) along with environmental drivers (such as freshwater or terrigenous inputs), allowing an increased generalization of this index. Some of these activities may not be accurately modelled with the particle diffusion model, so future iterations of this index should include other physical parameters (*e.g.* tidal or river plumes currents, sediment variability, temperature or salinity) to properly represent local anthropogenic influence. Furthermore, the cumulative index assumed an additive relationship with an equal weighting of all seven activities. This can introduce bias in the interpretation, as some activities may produce higher stress compared to others (*e.g.* Halpern et al., 2007). It is thus necessary to understand the links between exposure and vulnerability of species in order to properly study anthropogenic impacts.

Regressions between community characteristics and exposure indices resulted in models with moderate predictive power and relatively low coefficients. The significant effect of depth on communities is coherent with patterns of biodiversity in marine ecosystems (Gray and Elliott, 2009; Levinton, 2013; Piacenza et al., 2015). The lack of strong relationships for these models (especially for total density and total biomass) indicates that biodiversity of benthic communities is not particularly influenced by the proposed exposure indices, which may be linked to low exposure resulting in low structuring impacts for benthic communities. Another hypothesis is that community characteristics may not be the best descriptors to properly account for perturbation, in particular because of their univariate nature (Drouin et al., 2011).

We thus evaluated how the taxa assemblage respond to exposure using multivariate regression. Overall, exposure indices do not present a particularly strong influence on the composition of communities, except concerning a group of stations with a high cumulative exposure score where communities were similar. Dominant taxa at these stations are tolerant to perturbation, according to the list from Borja et al. (2000), and they have been found within perturbated areas in this region by Dreujou et al. (2020b). This describes a moderately perturbated profile, where some sensitive taxa are absent compared to a relative dominance of tolerant taxa without opportunists (Pearson and Rosenberg, 1978; Grall and Glémarec, 1997; Borja et al., 2000).

These results, along with the relatively low cumulative exposure index, are coherent with the general good portrait described by Carrière (2018) and Dreujou et al. (2020b). Highest values of cumulative exposure are located in the same areas where communities presented a moderately perturbed profile, as detected by Dreujou et al. (2020b) (*i.e.* stations close to Sept-Îles and the Pointe-Noire terminal). Furthermore, indicators such as the AZTI Marine Biotic Index (Borja et al., 2000), calculated by Dreujou et al. (2021) in the area, highlighted an overall 'good' to 'high' ecological status for benthic ecosystems. This further reinforces the strength of this index proposed in this study to detect anthropogenic influence at the local scale.

HMSC models allowed to predict taxa presence in relation to exposure indices, with the advantage of considering joint distributions and a Bayesian framework (Ovaskainen et al., 2017). Taxa that presented the highest predictive power (Nematoda, *E. tenuis*, *E. integra*, *Goniada maculata* and *M. neotena*) are among the most abundant taxa in the study area, which may explain the increased model accuracy. It is interesting that high coefficients where obtained for some exposure indices compared to abiotic parameters and in particular depth, a strong structuring factor in bentic ecosystems. This supports the methodology to compute exposure and their relevance to detect human influence on marine ecosystems. Future work should explore the possibility to use HMSC models by including both abiotic parameters and exposure indices as predictors, so that it is possible to compute variance partitioning analyses and to increase the predictive power of the models.

A majority of negative influences has been detected for exposure indices with HMSC, indicating a general 'sensitivity' of the communities to anthropogenic effects, while some taxa may benefit from it (such as *M. neotena* or *M. calcarea*). This point needs to be discussed with notions of taxa vulnerability (Wilson et al., 2005; Halpern et al., 2007), as high exposure do not necessarily mean high impacts on communities (the opposite is also true). Links described with this model provide valuable insights on how species may react to abiotic parameters and exposure from human activities. This is particularly relevant for stakeholders, as models allow predicting benthic taxa distribution under scenarios affecting both sets of variables, such as climate change models or anthropogenic development.

3.7 Conclusion

The exposure index developed in this study allowed to highlight regions where human activities could influence ecosystems, by describing individual and cumulative exposure from local human activities (aquaculture, city influence, industry influence, sediment dredging, commercial shipping, sewers and fisheries). These results contribute to a better understanding of the relationships between ecosystems and multiple human activities at the local scale.

The use of methods considering gradients of anthropogenic exposure is promising, as they allow to study influence without the definition of reference conditions which are often biased by lack of historical data or pristine ecosystems. This index has the possibility to be applied in a variety of regions, making it interesting to compare local ecosystems and generalize conclusions for cumulative impact assessments.

3.8 Acknowledgements

The authors would like to acknowledge and thank all the people that helped during the field campaigns, lab work and data analysis: Claudy Dechêsne, Serge Gallienne, Cindy Grant, Lisa Tréau de Coeli, Laure de Montety, Philippe-Olivier Dumais, Raphaël Bouchard, Simon Bélanger, Raphaël Mabit and Carlos Araujo. This study is a contribution to the research program of FRQNT Strategic Network Québec-Océan, which also provided scientific support.

3.9 Supplementary material

Table 16 – Predictor coefficients, Tjur's pseudo-R^2 and validation results obtained by *Hierarchical Modelling of Species Communities*. Taxa with a R^2 higher than 0.15 are displayed here. Validation results show the number of stations where taxa is observed and predicted (in green) and where taxa are observed but not predicted (red). Aq = aquaculture, Ci = city, Dr = dredging, Fi = fisheries, In = industry, Se = sewers, Sh = shipping, OM = organic matter, As = arsenic, Cd = cadmium, Cu = copper, Hg = mercury.

			Exposure indices						
	R^2	**Intercept**	**Aq**	**Ci**	**Dr**	**Fi**	**In**	**Se**	**Sh**
Nematoda	0.47	-0.596	0.265	-0.339	-0.203	-0.022	-0.598	0.61	0.225
Eudorellopsis integra	0.42	0.302	-0.087	-0.199	0.173	-0.032	-0.415	0.174	0.479
Goniada maculata	0.4	-0.486	0.127	-0.114	-0.046	0.144	-0.325	0.22	0.348
Crenella decussata	0.39	-2.86	0.107	-0.52	-0.136	-0.105	-0.257	0.004	-0.072
Ennucula tenuis	0.39	-0.259	0.344	0.021	0.241	0.08	-0.413	0.241	0.398
Pontoporeia femorata	0.38	-0.949	-0.261	0.275	-0.21	-0.205	-0.007	0.276	-0.061
Leucon (Leucon) nasicoides	0.37	-0.346	0.072	-0.086	-0.121	-0.028	-0.369	0.351	0.32
Hediste diversicolor	0.36	-1.66	-0.373	0.0197	-0.101	-0.197	0.282	0.05	-0.46
Cumacea	0.34	-2.67	-0.039	-0.357	-0.108	-0.092	-0.299	0.137	0.077
Echinarachnius parma	0.33	-2.05	0.266	-0.113	0.176	0.023	-0.033	0.123	-0.069
Nephtys incisa	0.33	-0.942	0.106	-0.157	-0.055	-0.067	-0.514	0.194	0.323
Akanthophoreus gracilis	0.3	-1.05	0.337	-0.316	-0.009	0.126	-0.522	0.188	0.317
Micronephthys neotena	0.3	1.39	0.134	0.324	0.28	0.098	-0.436	0.367	0.392
Strongylocentrotus sp.	0.29	-3.3	-0.288	-0.537	-0.197	0.159	-0.161	-0.102	-0.105
Halacaridae spp.	0.27	-3.5	-0.2	-0.437	-0.118	0.202	-0.075	-0.083	-0.123
Protomedeia grandimana	0.27	-0.274	-0.05	0.33	-0.052	-0.138	-0.519	0.395	0.126
Chlamys islandica	0.26	-3.56	-0.206	-0.445	-0.114	0.203	-0.059	-0.097	-0.138
Thyasira gouldi	0.26	-0.95	0.157	-0.246	-0.136	0.01	-0.525	0.417	0.405
Musculus discors	0.25	-3.47	-0.205	-0.412	-0.119	0.194	-0.075	-0.094	-0.145
Edotia montosa	0.24	-2.95	-0.184	-0.286	-0.247	-0.174	0.275	0.087	-0.261
Maldanidae spp.	0.24	-1.03	0.165	-0.153	-0.103	-0.073	-0.416	0.27	0.394
Chaetodermatida	0.23	-1.39	0.147	-0.11	-0.003	0.06	-0.46	0.265	0.188
Eudorella emarginata	0.23	-2.15	-0.215	-0.226	-0.183	-0.152	-0.252	-0.005	0.305
Retusa obtusa	0.23	-2.1	-0.177	0.322	0.017	-0.023	0.171	0.056	-0.252
Axinopsida orbiculata	0.22	-0.951	-0.114	0.226	0.238	0.23	-0.347	-0.142	0.05
Harpacticoida	0.22	-0.242	-0.313	0.29	-0.305	-0.005	-0.107	0.151	0.097
Oligochaeta	0.22	-1.64	-0.172	0.297	-0.043	-0.066	-0.272	-0.016	0.337
Scoletoma sp.	0.22	-3.08	-0.146	-0.362	0.275	0.194	-0.088	-0.14	-0.174
Bathymedon longimanus	0.21	-2.45	0.319	-0.001	0.221	-0.073	-0.15	-0.031	-0.07
Solariella sp.	0.21	-2.54	-0.107	-0.348	-0.086	0.175	-0.223	-0.005	-0.005
Byblis gaimardii	0.2	-2.92	-0.147	-0.459	-0.179	0.155	-0.324	-0.064	-0.023
Cistenides granulata	0.2	-1.25	0.008	-0.249	0.147	0.128	-0.32	-0.045	0.063
Macoma calcarea	0.2	0.846	0.078	0.375	0.138	0.099	-0.337	0.293	0.09
Philomedes sp.	0.2	-2.84	-0.15	-0.448	-0.106	0.025	-0.344	-0.016	0.031
Scoletoma fragilis	0.19	-1.82	-0.25	-0.296	-0.095	-0.178	0.094	0.078	-0.2
Sipuncula	0.19	-1.18	0.112	0.101	0.010	-0.078	-0.224	0.133	0.184
Ampelisca vadorum	0.18	-3.6	-0.214	-0.503	-0.207	-0.025	-0.129	-0.071	-0.11
Boreochiton ruber	0.18	-3.02	-0.162	-0.443	-0.118	0.075	-0.09	-0.122	-0.176

	R^2	Intercept	Exposure indices						
			Aq	Ci	Dr	Fi	In	Se	Sh
Lamprops fuscatus	0.18	-1.64	0.277	-0.268	0.209	0.015	0.076	0.095	0.125
Lamprops quadriplicata	0.18	-2.96	-0.159	-0.28	-0.218	-0.159	0.021	0.299	-0.235
Ophiura robusta	0.18	-2.86	-0.16	-0.394	-0.109	0.034	-0.145	0.033	-0.139
Propebela turricula	0.18	-2.39	-0.1	-0.452	-0.133	0.084	-0.375	0.202	0.286
Quasimelita formosa	0.18	-0.885	0.257	0.052	0.052	0.026	-0.481	0.368	0.233
Brachydiastylis sp.	0.17	-3.34	-0.195	-0.511	-0.204	0.018	-0.246	-0.068	-0.036
Caprella septentrionalis	0.17	-2.49	-0.293	0.01	-0.235	-0.224	-0.126	0.072	-0.334
Cirripedia	0.16	-3.33	-0.245	-0.337	-0.114	-0.069	0.117	-0.105	-0.252
Glycera sp.	0.16	-2.37	-0.338	-0.192	-0.272	-0.213	-0.103	-0.084	-0.308
Ischyrocerus anguipes	0.16	-2.51	-0.3	-0.072	-0.217	-0.191	-0.043	-0.122	-0.357
Lepeta caeca	0.16	-2.46	-0.198	-0.287	-0.156	0.058	-0.183	-0.005	-0.163
Lysianassidae spp.	0.16	-3.17	0.21	-0.356	-0.103	-0.064	0.171	-0.12	-0.061
Nephtyidae spp.	0.16	-1.72	-0.054	-0.149	0.254	0.254	-0.301	-0.069	-0.166
Polynoidae spp.	0.16	-0.976	-0.187	-0.057	-0.259	-0.198	-0.436	0.367	0.06

(Table 16 continued)

	Abiotic parameters									Validation
	Depth	OM	Gravel	Silt	Clay	As	Cd	Cu	Hg	
Nematoda	0.259	-0.246	-0.131	-0.331	-0.323	-0.228	-0.568	-0.073	-0.19	5 ; 3
Ennucula tenuis	0.31	0.611	-0.175	-0.115	0.023	0.122	-0.157	-0.457	0.437	6 ; 3
Eudorellopsis integra	0.629	0.223	-0.405	-0.096	-0.2	-0.374	0.043	-0.268	0.231	7 ; 5
Goniada maculata	-0.052	-0.71	0.279	0.063	-0.277	-0.783	-0.599	0.508	-0.509	3 ; 2
Micronephthys neotena	0.676	0.271	-0.066	-0.191	0.156	-0.163	-0.413	-0.243	0.276	11 ; 3
Leucon (Leucon) nasicoides	-0.107	0.363	0.029	0.267	-0.3	-0.07	0.096	-0.152	0.451	4 ; 2
Pontoporeia femorata	0.562	0.616	-0.249	-0.307	0.097	-0.087	-0.125	-0.258	0.399	2 ; 2
Crenella decussata	-0.709	0.107	0.094	0.504	-0.262	-0.166	0.093	-0.153	0.075	0 ; 1
Echinarachnius parma	0.426	-0.168	-0.137	-0.145	0.29	-0.407	-0.168	0.508	-0.089	1 ; 3
Nephtys incisa	-0.152	-0.615	-0.148	0.047	-0.334	-0.715	-0.431	0.425	-0.566	1 ; 7
Strongylocentrotus sp	0.647	0.364	-0.031	-0.148	0.197	-0.316	-0.349	0.01	0.303	0 ; 0
Harpacticoida	0.399	0.052	-0.207	-0.137	0.178	-0.209	-0.396	0.15	-0.031	3 ; 7
Retusa obtusa	0.231	0.904	-0.163	-0.035	0.051	0.082	-0.046	-0.661	0.485	0 ; 0
Hediste diversicolor	-0.206	-0.645	0.17	0.388	-0.346	-0.704	-0.427	0.518	-0.488	0 ; 4
Halacaridae	-0.185	-0.653	-0.013	0.314	-0.343	-0.726	-0.221	0.625	-0.395	0 ; 0
Musculus discors	0.337	0.65	-0.042	-0.174	0.058	0.142	-0.352	-0.321	0.34	0 ; 0
Protomedeia grandimana	-0.18	-0.67	0.012	0.322	-0.335	-0.744	-0.227	0.633	-0.392	5 ; 2
Chlamys islandica	0.493	0.121	-0.133	-0.039	-0.196	-0.359	-0.517	0.119	0.004	0 ; 0
Akanthophoreus gracilis	-0.2	-0.623	-0.007	0.311	-0.327	-0.695	-0.201	0.59	-0.379	0 ; 7
Maldanidae	-0.472	-0.494	-0.019	0.04	-0.396	-0.542	-0.095	0.533	-0.424	2 ; 1
Oligochaeta	0.383	0.296	-0.261	0.017	0.134	-0.232	-0.273	0.275	-0.011	0 ; 4
Propebela turricula	0.536	-0.019	-0.326	-0.363	0.182	-0.409	-0.275	0.174	-0.011	0 ; 0
Solariella sp	-0.052	0.154	0.235	0.322	-0.198	-0.277	-0.142	0.233	0.125	0 ; 2
Thyasira gouldi	-0.349	-0.03	-0.031	0.287	-0.282	-0.033	0.309	0.016	-0.16	0 ; 6
Chaetodermatida	0.218	0.221	-0.18	-0.166	0.187	0.128	0.145	-0.309	-0.202	0 ; 2
Cistenides granulata	-0.353	0.112	-0.099	-0.162	-0.264	0.129	-0.048	-0.349	-0.046	0 ; 2
Eudorella emarginata	-0.173	0.204	-0.068	0.244	-0.209	-0.229	0.02	0.056	0.067	0 ; 0
Scoletoma sp	-0.229	-0.613	-0.041	0.285	-0.32	-0.648	-0.179	0.501	-0.376	0 ; 0
Axinopsida orbiculata	-0.174	-0.273	-0.073	0.075	-0.248	-0.584	-0.063	0.307	-0.347	0 ; 2
Cumacea	0.168	-0.372	0.073	0.231	-0.249	-0.605	-0.414	0.242	-0.371	0 ; 0

	Abiotic parameters									Validation
	Depth	OM	Gravel	Silt	Clay	As	Cd	Cu	Hg	
Macoma calcarea	0.192	-0.419	0.003	-0.008	-0.218	-0.673	-0.339	0.42	-0.253	11 ; 4
Philomedes sp	-0.053	-0.052	-0.037	0.045	-0.194	-0.401	-0.314	-0.016	-0.129	0 ; 0
Ampelisca vadorum	-0.041	0.475	-0.161	0.004	-0.263	0.091	-0.155	-0.658	0.002	0 ; 0
Brachydiastylis sp	0.374	-0.459	0.054	-0.042	-0.178	-0.642	-0.367	0.476	-0.266	0 ; 0
Byblis gaimardii	-0.384	0.097	-0.002	0.504	-0.102	-0.367	-0.206	0.269	-0.152	0 ; 0
Diastylis sculpta	0.172	0.317	-0.144	0.142	0.168	0.145	-0.119	-0.075	-0.093	0 ; 2
Ostracoda	-0.071	-0.641	0.245	0.247	-0.298	-0.748	-0.328	0.623	-0.394	3 ; 3
Edotia montosa	-0.323	-0.618	0.104	0.324	-0.35	-0.74	-0.418	0.437	-0.468	0 ; 1
Glycera sp	-0.079	-0.347	-0.172	0.041	-0.272	-0.6	-0.181	0.414	-0.247	0 ; 1
Thracia septentrionalis	-0.295	-0.477	-0.022	-0.012	-0.389	-0.502	-0.115	0.388	-0.278	0 ; 4
Astarte sp	-0.096	-0.613	-0.028	0.131	-0.346	-0.732	-0.499	0.404	-0.451	0 ; 2
Boreochiton ruber	0.594	-0.245	-0.159	0.07	-0.163	-0.519	-0.272	0.391	0.022	0 ; 0
Nephtyidae	0.332	0.202	-0.101	0.044	-0.199	-0.246	-0.23	0.013	-0.142	0 ; 2
Ophiura robusta	0.111	-0.612	0.124	0.137	-0.256	-0.719	-0.364	0.571	-0.366	0 ; 0
Phoxocephalus holbolli	-0.606	-0.315	0.013	0.088	-0.191	-0.515	-0.065	0.213	-0.235	0 ; 4
Sipuncula	-0.328	-0.569	0.041	0.299	-0.356	-0.691	0.217	0.36	-0.357	0 ; 0
Amphipoda	-0.514	-0.075	0.069	0.394	-0.339	-0.409	0.044	0.299	-0.221	0 ; 6
Anonyx lilljeborgi	-0.633	-0.147	-0.01	0.139	-0.34	-0.477	0.136	0.359	-0.262	0 ; 2
Axiothella catenata	-0.192	-0.487	0.204	0.305	-0.336	-0.575	-0.331	0.278	-0.339	0 ; 0
Bathymedon longimanus	-0.234	-0.593	-0.013	0.274	-0.287	-0.78	-0.296	0.699	-0.522	0 ; 0
Caprella septentrionalis	0.115	-0.022	-0.256	-0.102	-0.156	-0.118	0.051	0.035	0.003	0 ; 1
Serripes groenlandicus	0.099	0.346	-0.15	0.065	-0.275	-0.341	-0.297	-0.09	0.088	0 ; 0

CONCLUSION GÉNÉRALE

Cette thèse de doctorat a considéré trois objectifs principaux : (i) étudier les communautés et les habitats d'une zone industrialo-portuaire subarctique pour comprendre leur structure et les facteurs qui les influencent, (ii) déterminer le statut écologique de ces écosystèmes au moyen de méthodes couramment utilisées dans les évaluations environnementales, et (iii) développer un modèle permettant de décrire l'influence anthropique sur les écosystèmes benthiques côtiers, en considérant de multiples activités humaines à l'échelle locale. La Figure 22 complète le schéma conceptuel de l'introduction en y ajoutant les différentes contributions de chaque chapitre, discutées dans les paragraphes suivants.

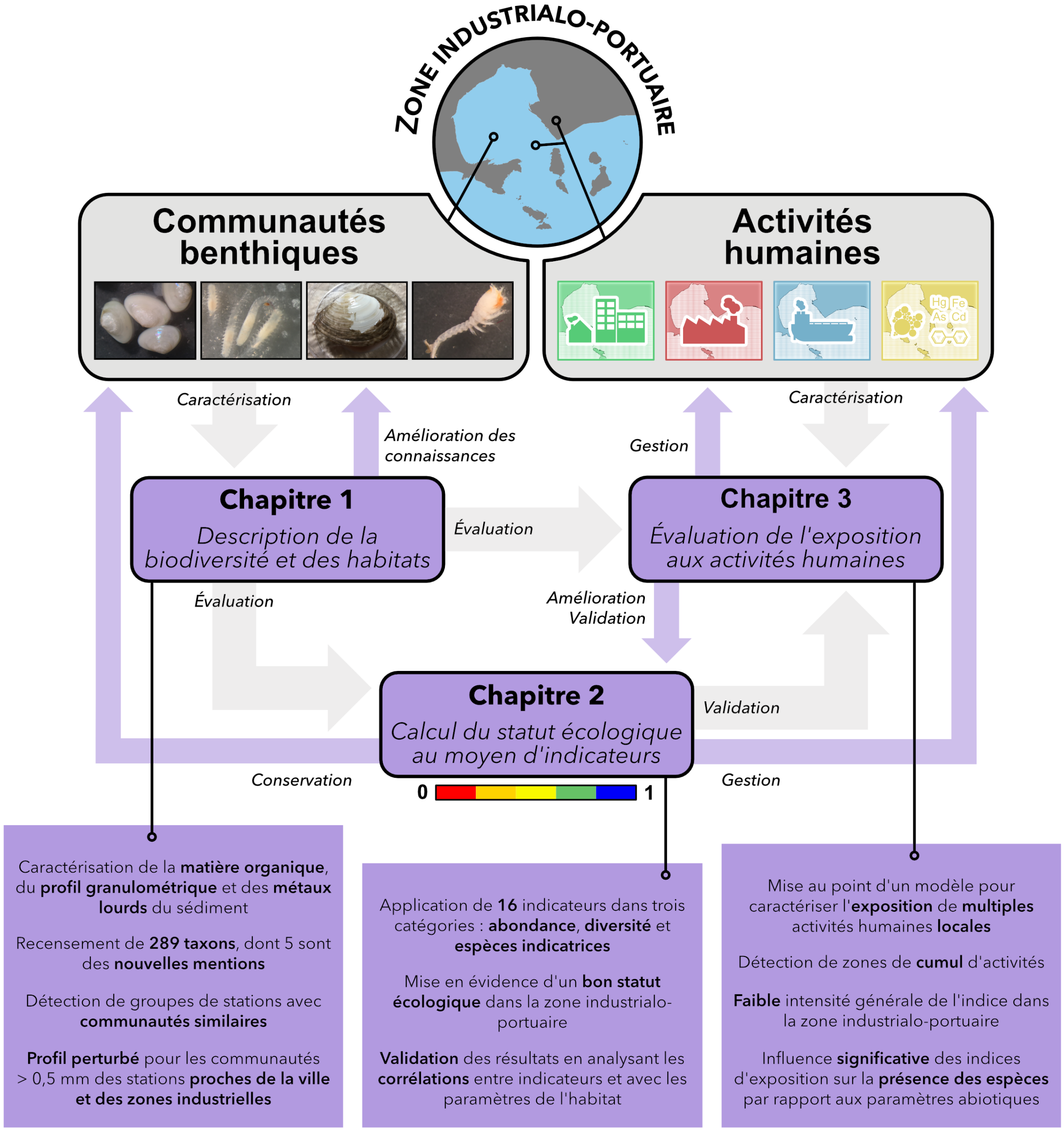


FIGURE 22 – Diagramme intégratif représentant les liens entre les différents chapitres de la thèse de doctorat, ainsi que les contributions principales de chaque chapitre. Les flèches grises correspondent aux liens entre composantes considérées des écosystèmes de la région d'étude (encadrés gris) et les chapitres (encadrés pourpres). Les flèches pourpres correspondent au débouché de chaque chapitre.

Contributions de la thèse

Augmentation des connaissances sur les écosystèmes côtiers subarctiques sous influence anthropique

Mon premier chapitre constitue le premier relevé de biodiversité à l'échelle de la région de Sept-Îles. Ce travail a constitué la base de nombreuses analyses des chapitres suivants, et a permis d'augmenter les connaissances sur les écosystèmes benthiques de la région où peu d'informations sont disponibles. Les précédents recensements d'espèces dans la baie des Sept Îles, par exemple sur la plateforme *Ocean Biodiversity Information System* (OBIS, 2020), ont visé en priorité les espèces pélagiques, telles que les poissons ou les crustacés d'intérêt commercial. Au total, 289 taxons ont été identifiés, la plupart présents dans le Golfe du Saint-Laurent et cinq sont des nouvelles mentions dans la région (*Bathyporeia quoddyensis*, *Glycera alba*, *Kirkegaardia* sp., *Pholoe minuta tecta*, *Tricellaria arctica*). L'ajout de ces informations permet ainsi de réaliser un portrait cohérent des différentes composantes de l'écosystème. Ceci est d'autant plus utile que les écosystèmes subarctiques sont souvent sous-représentés dans les études écologiques évaluant l'impact anthropique.

L'utilisation de deux mailles différentes (0,5 mm et 1 mm) pour tamiser les échantillons a permis de distinguer deux ensembles d'espèces spécifiques et d'identifier la méthodologie à privilégier pour étudier ces écosystèmes. L'étude de la similarité entre les communautés a démontré que les trois groupes identifiés pour les communautés $> 0{,}5$ mm pouvaient être reliés à un état de perturbation, où des espèces tolérantes à la perturbation et opportunistes étaient présentes aux stations proches des villes et zones industrielles. Cet état n'a pas été détecté pour les communautés > 1 mm, où la variabilité naturelle était le principal moteur de la variabilité observée. La détection d'un profil de perturbation relativement faible dans la zone industrialo-portuaire de Sept-Îles, hors quelques stations en face de la ville et du terminal de Pointe-Noire, est un résultat intéressant pour les initiatives de gestion des écosystèmes dans la région. Il serait intéressant d'étudier spécifiquement les capacités de résilience et de résistance des espèces présentes, par exemple en étudiant leur évolution selon des gradients de perturbation, afin de mieux comprendre les réponses des communautés face aux activités humaines. Dans un contexte d'augmentation de l'empreinte anthropique sur les écosystèmes, notamment grâce à une forte demande en ressources fossiles et à l'accroissement de la population, les données collectées permettent de définir des conditions de référence, c'est-à-dire des conditions caractéristiques d'un faible état d'anthropisation du milieu, dont l'apport sera particulièrement utile à la conservation des écosystèmes de Sept-Îles à long terme.

Application d'indicateurs du statut écologique pour des écosystèmes canadiens

Le deuxième chapitre s'est intéressé au statut écologique des écosystèmes marins. En me basant sur les travaux de Rice (2003) et Salas et al. (2006), j'ai appliqué un ensemble de seize indicateurs sur les données des communautés $>$ 0,5 mm, divisé en trois catégories méthodologiques : indicateurs basés sur les abondances de taxons, des mesures de diversité ou des espèces caractéristiques. Cet exercice a permis de décrire un état écologique généralement élevé, où les communautés sont diversifiées, sans taxon dominant ni abondance élevée d'opportunistes, confirmant ainsi les hypothèses proposées au chapitre précédent. Les indicateurs considérant des espèces caractéristiques (*AZTI Marine Biotic Index*, BENTIX et *Benthic Opportunistic Polychaetes Amphipods ratio*) ont été développés pour décrire un état écologique, en fonction de conditions de référence. Il est cependant important de mentionner que la plupart de ces références sont basées sur les réponses de taxons à un type particulier de perturbation (l'augmentation en matière organique) et qu'elles ont été définies pour des écosystèmes particuliers (Pearson et Rosenberg, 1978; Borja et al., 2013). Bien que plusieurs études aient montré la pertinence d'utiliser ces indicateurs dans d'autres régions, peu se sont spécifiquement intéressées aux écosystèmes canadiens (lire par exemple Borja et al., 2008a; Callier et al., 2008; Robert et al., 2013; Gillett et al., 2015). Ceci rend les résultats de ce chapitre particulièrement pertinents, mais aussi sujets à caution car l'assignation des espèces canadiennes aux groupes écologiques basés sur les espèces européennes peut introduire un biais dans le calcul des indicateurs. Il sera donc nécessaire de poursuivre ce travail afin de valider les choix de classification des espèces, par exemple en testant expérimentalement leur réponse à différentes perturbations.

L'utilisation de techniques de rééchantillonnage statistique des données (comme l'algorithme *bootstrap*) a permis de tester la robustesse des indicateurs basés sur l'abondance et la diversité. Il s'agit d'une étape importante dans l'évaluation environnementale, et les différences relativement faibles entre les moyennes observées et les moyennes rééchantillonnées renforcent ainsi les conclusions obtenues. Cette analyse a été complétée par l'étude des liens entre indicateurs et paramètres de l'habitat. Ainsi, les résultats ont mis en évidence quels indicateurs produisent des interprétations similaires, ce qui permet de guider les protocoles d'évaluation environnementale tout en augmentant leur efficacité, ainsi que les types de perturbations qu'ils sont capables de détecter. La forte corrélation entre indicateurs et concentrations en métaux lourds, en particulier pour les stations profondes, est un résultat intéressant mais à explorer davantage, car les indicateurs utilisés n'ont pas été développés spécifiquement pour détecter ce type de perturbation.

Développement et application d'un outil pour évaluer l'exposition anthropique locale

Le dernier chapitre a mis en relation les données des chapitres précédents avec les sources d'activités humaines présentes à Sept-Îles. Je me suis basé sur une méthode d'évaluation d'impact cumulé, l'indice de Halpern et al. (2008 ; mis à jour par Halpern et al., 2015, et Halpern et al. (2019)), pour calculer un score local d'exposition considérant l'influence de sept activités humaines : l'aquaculture, l'influence de la ville, l'influence des industries, le dragage des sédiments, la navigation commerciale, les égouts et la pêche. Le principal avantage de cet outil réside dans sa capacité à être appliqué dans de nombreux écosystèmes, car il demande peu de données pour être utilisé. L'utilisation d'un modèle de diffusion particulaire s'est révélée prometteuse pour caractériser l'exposition anthropique, ce qui est particulièrement pertinent pour les écosystèmes où peu de données (notamment sur la circulation océanique) sont disponibles.

L'exposition individuelle des activités humaines considérées est généralement faible sauf à proximité immédiate des sources d'activité, ce qui est cohérent avec les résultats des chapitres précédents. Les zones où l'exposition cumulée est la plus élevée (en face de la ville de Sept-Îles et des opérations industrielles du secteur de Pointe-Noire) coïncident avec les stations où les communautés benthiques possèdent un profil perturbé (détectées par l'analyse de similarité du chapitre 1). La présence de ces zones de superposition de l'exposition anthropique ("*hotspots*") est ainsi concomitante à un certain état de perturbation des communautés benthiques, ce qui renforce les conclusions obtenues grâce à cet outil. Cependant, il est important de noter que les indices d'exposition ont été appliqués sur un environnement statique en 2D, c'est-à-dire sans variations dynamiques des masses d'eau dans l'espace et le temps ni avec la profondeur. Shaw (2019) a montré que la Baie des Sept Îles est particulièrement influencée par les courants de marée en s'intéressant à un transect à l'entrée de la baie, ce qui indique que l'indice développé détecte une exposition théorique sans considération de la "dilution" possible des particules hors de la baie ou de la "concentration" dans des zones spécifiques. De plus, il est tout à fait envisageable que des dynamiques source-puits, c'est-à-dire des transferts de matière et d'énergie entre écosystèmes voisins, soient présentes dans cette région, ce qui complexifierait l'évaluation environnementale et demanderait l'utilisation d'une analyse à plus large échelle.

Une meilleure compréhension des facteurs environnementaux impactant les communautés

Les modèles de régression décrits aux chapitres 1 et 3 augmentent la compréhension des différentes composantes des écosystèmes côtiers subarctiques. Il a été possible de mettre en évidence des relations significatives expliquant la diversité de la communauté avec les paramètres abiotiques d'une part, et avec les indices d'exposition d'autre part, où le pouvoir explicatif (R^2) a atteint jusqu'à 0,5. Une influence importante de la profondeur y a été détectée, ce qui est cohérent avec les patrons de diversité en milieu marin (Gray et Elliott, 2009; Levinton, 2013; Piacenza et al., 2015). Plusieurs paramètres abiotiques (comme la matière organique ou les concentrations de métaux lourds) ont aussi montré un effet significatif sur les communautés, et l'utilisation des indices d'exposition est une possibilité intéressante pour comprendre comment les activités humaines affectent ces variables, et ainsi mieux comprendre les facteurs structurant les communautés benthiques. S'ajoute à ce portrait la modélisation des communautés, obtenue à partir du *Hierarchical Modelling of Species Communities*, dont les résultats permettent de prédire la présence d'espèces particulières dans les écosystèmes considérés. Ce type de modèle offre l'avantage de calculer des probabilités jointes pour les espèces, considérant ainsi les interactions entre les distributions des taxons présents (Ovaskainen et Abrego, 2020). Il est possible d'inclure des données complémentaires à cette méthode pour améliorer les prédictions, comme des traits biologiques et des informations phylogénétiques, ce qui la rend particulièrement pertinente dans le cadre d'évaluations environnementales holistiques (Ovaskainen et al., 2017 ; 2020). Ce point est notamment discuté dans l'étude en annexe, en particulier au sein de la priorité de recherche V (Dreujou et al., 2020a). Enfin, la possibilité de prédire la distribution des espèces est particulièrement appropriée dans le cas d'espèces dites "indicatrices" (telles que définies dans le chapitre 2) : une meilleure compréhension de leur dynamique et des facteurs qui influencent leur distribution permet ainsi de renforcer les conclusions obtenues lors de l'utilisation d'indicateurs environnementaux, ainsi que d'estimer l'état écologique des communautés selon différents scénarios d'influence anthropique.

Perspectives

Accroître le volume de données écologiques en milieu subarctique

De nombreux habitats côtiers figurent parmi les écosystèmes plus susceptibles à l'influence humaine, de par la présence d'activités humaines variées et de nombreux habitats qui y sont vulnérables (Halpern et al., 2019). Je me suis intéressé aux liens entre les écosystèmes côtiers benthiques et l'influence anthropique, en prenant comme région d'étude la zone industrialo-portuaire de Sept-Îles. Ce type de structure regroupe un nombre et une diversité élevés d'activités humaines dans une zone réduite, induisant ainsi la possibilité d'augmenter sensiblement et durablement l'influence humaine sur les communautés et les habitats. Il s'agit d'une étude pionnière au Canada, et en particulier pour les écosystèmes subarctiques. La formation de la banquise et de glace sur les côtes en fin d'automne (induisant par exemple un isolement des écosystèmes ou l'érosion du sédiment) et sa fonte au printemps (modifiant drastiquement la salinité et la température) constituent des facteurs environnementaux qui impactent considérablement la structure des écosystèmes (par ex. Demers et al., 2018). De par ces conditions, et par le fait que ces écosystèmes soient souvent difficiles d'accès, il existe un manque de données à propos des espèces et des habitats, ce qui complique la description de l'influence anthropique sur ces milieux. Alors que les différents scénarios liés au changement climatique prévoient une augmentation des activités humaines sur les écosystèmes marins, il existe un besoin urgent d'améliorer nos connaissances sur ces écosystèmes (Moritz et al., 2002; Piepenburg, 2005; Yamanouchi, 2011; Arrigo et al., 2020). Différentes voies permettraient d'atteindre cet objectif, au moyen d'études dédiées sur le terrain ou en laboratoire, ainsi qu'en mettant en place des observatoires annuels fixes pour obtenir de séries temporelles écologiques fiables (voir par exemple les travaux de Galbraith et al., 2020, pour une application annuelle à l'échelle du Golfe du Saint-Laurent), ce qui consituerait des atouts importants pour les stratégies de gestion.

Au cours de cette thèse, j'ai considéré les macro-invertébrés benthiques pour répondre aux objectifs des différents chapitres. Ce choix repose sur les modèles ayant mis en évidence leur capacité à rendre compte des perturbations de l'écosystème (voir les travaux fondateurs de Pearson et Rosenberg, 1978). Cependant, des liens entre perturbations et d'autres compartiments écologiques comme le phytoplancton, le zooplancton ou les espèces pélagiques ont aussi été décrits dans des écosystèmes marins subarctiques, tout comme l'importance du compartiment microbien dans les différents cycles chimiques des écosystèmes. L'ajout d'informations sur ces groupes permettrait d'avoir un portrait intégré des écosystèmes, notamment en considérant l'évolution des réseaux trophiques en fonction des activités humaines (Beauchesne et al., 2020a).

Vers une meilleure compréhension des écosystèmes de la zone industrialo-portuaire de Sept-Îles

L'échantillonnage réalisé à Sept-Îles a permis d'obtenir de nombreuses données sur les habitats et les communautés benthiques subarctiques. Les stations ont été sélectionnées afin d'obtenir une couverture complète de la baie et de l'archipel des Sept Îles, de façon aléatoire avec un effort accru près de sources d'activité humaine (Quinn et Keough, 2002; Eleftheriou et McIntyre, 2005; Underwood, 2012). Ceci a permis la découverte de relations significatives, que ce soit au niveau de la similarité entre communautés, du calcul du statut écologique ou des liens avec l'exposition anthropique. Un biais d'échantillonnage a pu être introduit de par l'impossibilité de considérer des stations plus profondes que 80 m à cause de limitations logistiques. En effet, il est alors impossible d'évaluer comment les communautés profondes présentes dans les chenaux de l'archipel (jusqu'à 200 m de profondeur) réagissent aux activités humaines. Même s'il est probable que cette influence serait faible, ceci limite les interprétations, en particulier pour les indices d'exposition : une exposition cumulée relativement élevée a été détectée dans les chenaux profonds, mais il n'est pas possible de savoir si elle impacte les communautés de façon effective. De plus, le recensement de biodiversité obtenu dans cette région est certainement une sous-estimation de la richesse taxonomique totale. Les futures études dans cette région devront ainsi intégrer un échantillonnage représentatif des stations profondes, afin de pouvoir améliorer la portée des interprétations.

La principale limitation rencontrée tout au long de ce doctorat consiste en l'absence d'un modèle détaillé des courants et de leur intensité dans la Baie des Sept Îles. Les travaux pionners de Shaw (2019) ont permis d'acquérir des connaissances de base mais de plus amples études sont à effectuer par exemple grâce au déploiement de bouées dérivantes. Ceci permettra en particulier de comprendre la connectivité entre les habitats ainsi que le temps de résidence des contaminants au sein de la baie, deux informations particulièrement importantes dans le cadre d'une stratégie de gestion intégrée à l'échelle locale.

La baie des Sept Îles possède une géographie particulière dans la région Côte-Nord, avec une baie relativement abritée par l'archipel comparativement au reste des côtes voisines. Le chapitre 1 a permis de comparer les conditions au sein de la baie comparativement à trois autres portions de côte. Certaines différences intéressantes ont été détectées (voir clusters B *vs* C et D/E *vs* F), cependant il est possible que les communautés à Sept-Îles soient différentes de celles d'autres régions à cause de conditions environnementales différentes (en lien avec la géographie, la connectivité et l'hydrodynamisme), et non à cause d'une influence anthropique. Ceci est d'autant plus vrai pour le secteur de Port-Cartier, une région anthropisée à l'ouest de Sept-Îles, qui possède une plus forte similarité avec les secteurs des rivières Pentecôte et Manitou qu'avec celui de Sept-Îles.

Cette spécificité géographique peut être un obstacle pour la généralisation des résultats, notamment concernant la définition de conditions de référence. La comparaison avec d'autres écosystèmes côtiers dans le golfe du Saint-Laurent, tels que la zone industrialo-portuaire de Baie-Comeau, permettrait ainsi de mieux comprendre la dynamique et la variabilité naturelle des écosystèmes.

Les analyses décrites dans cette thèse ont ciblé les écosystèmes subtidaux à substrat meuble, afin d'étudier des habitats semblables (ce qui est nécessaire pour les indicateurs du chapitre 2). La baie des Sept Îles présente d'autres habitats singuliers, en particulier des herbiers à *Zostera marina* dans la partie nord-ouest et des côtes rocheuses le long des îles à l'entrée de la baie. Dans le but d'obtenir une évaluation intégrée de l'état écologique, la considération de ces habitats serait particulièrement pertinente. Il est probable que l'indice d'exposition développé au chapitre 3 puisse être adapté pour évaluer spécifiquement d'autres types d'habitats. Selon les résultats obtenus avec l'indice d'exposition, la région où se trouvent les herbiers possède une exposition cumulée particulièrement faible ce qui pourrait indiquer que cet habitat n'est que peu influencé par les activités humaines considérées. Les travaux de Simon Bélanger, de Carlos Araujo et de l'INREST indiquent en effet une bonne santé de l'herbier de Sept-Îles, avec une augmentation de son étendue depuis 1985 malgré le déclin de ce type d'habitat à l'échelle globale (Orth et al., 2006; Waycott et al., 2009; Paquette et al., 2018; Murphy et al., 2019; Zimmerman, 2021). Cette observation pourrait corroborer l'hypothèse du bon état de l'herbier, mais des études dédiées sur les liens entre la physiologie de cette espèce, les communautés benthiques et les influences anthropiques sont nécessaires pour la confirmer.

Enfin, j'ai réalisé une campagne pilote sur les écosystèmes intertidaux à sédiment meuble en 2016, dans le but d'explorer la variabilité entre cinq portions de côte (secteurs des rivières Pentecôte et Manitou, de Port-Cartier, de Sept-Îles et de la baie Sheldrake). Les communautés présentes dans ces habitats ont montré peu de variabilité entre les régions considérées, qu'elles soient anthropisées ou non, ce qui peut être expliqué par une forte tolérance naturelle de ces espèces aux facteurs de stress ou par une faible influence anthropique générale (telle que décrite pour les écosystèmes subtidaux). Les données sur les habitats et les communautés collectées lors de cette campagne n'ont pas fait l'objet d'une publication, et sont disponibles dans la section d'annexe de cette thèse.

Évaluation intégrée du statut écologique des communautés

Les indicateurs environnementaux choisis pour le chapitre 2 sont documentés et supportés par un corpus conséquent d'études écologiques (lire Pinto et al., 2009; Teixeira et al., 2016). Leur utilisation est intégrée dans divers protocoles d'évaluation environnementale (par ex. la *Marine Strategy Framework Directive*, European Commission, 2008), dans le but de guider la prise de décision et la gestion des écosystèmes marins. Bien que l'utilisation d'indicateurs présente de nombreux avantages (par ex. des approches quantitatives, la capacité à synthétiser la complexité des écosystèmes ou la considération de multiples acteurs écosystémiques), il faut les choisir adéquatement en fonction de l'écosystème et des objectifs écologiques (DEVOTES, 2012; McQuatters-Gollop et al., 2019). Il est certain que des indicateurs supplémentaires seraient applicables pour les écosystèmes de Sept-Îles, mais il est probable que l'information qui en résulterait soit redondante avec les indicateurs appliqués (*c.f.* les corrélations significatives obtenues entre les indicateurs utilisés).

L'intégration de différents compartiments de l'écosystème au sein d'une approche intégrée est une perspective pertinente pour poursuivre le travail entrepris durant cette thèse. Ceci permettrait de définir le statut environnemental autrement qu'avec des indicateurs basés sur des mesures univariées ou des espèces indicatrices, tout en faisant écho aux perspectives discutées en annexe (Dreujou et al., 2020a). À cet effet, certains indicateurs, tels que *Ecosystem-Based Quality Index*, considèrent l'écosystème comme un réseau, où chaque composante peut être évaluée en comparant les valeurs de différents paramètres (par ex. le couvert algal ou l'abondance) à une valeur considérée comme représentative d'un état non-perturbé (Rastorgueff et al., 2015). En complément d'une valeur globale, cet indicateur présente un résultat pour chaque composante considérée, mettant ainsi en évidence lesquelles peuvent être possiblement perturbées. Cet indicateur possède un fort potentiel d'intégration, propriété désirable pour les évaluations environnementales, néanmoins le calcul se base sur des connaissances servant de référence à chaque composante ce qui nécessite un haut niveau de connaissance intrinsèque de l'écosystème. Le volume de données nécessaire peut donc augmenter drastiquement avec le nombre de composantes et d'espèces considérées, ce qui complexifie l'application de cet indicateur (surtout dans le cas où la physiologie et l'écologie des espèces sont méconnues). Une méthode complémentaire pourrait être la description de “gradients d'anthropisation” depuis la ou les sources d'activité pour calculer un statut environnemental. Cette logique se rapproche de l'indice qui a été développé pour le chapitre 3, apportant l'avantage de ne pas avoir à établir de conditions de référence autres que les bornes minimale et maximale du gradient.

Exposition aux activités humaines et évaluation environnementale

Les chapitres 2 et 3 ont été confrontés à la définition des concepts utilisés lors des évaluations d'impact. La sémantique utilisée pour décrire les effets mesurés sur les écosystèmes est diverse, et cet ensemble se complexifie davantage avec les correspondances linguistiques, par exemple de l'anglais vers le français (par ex. pour *stressor*), et les correspondances entre disciplines scientifiques (Gari et al., 2014; Orr et al., 2020). Judd et al. (2015) ont proposé des définitions pour uniformiser ce lexique, mais des travaux supplémentaires, ainsi qu'une concertation entre les acteurs de ces évaluations, sont nécessaires pour rendre ces recommandations universelles.

Pour calculer l'influence des activités humaines sur les écosystèmes benthiques, j'ai choisi de modéliser une *exposition* des écosystèmes aux activités humaines et non un *impact* ou un *stress*. Ce choix est supporté par le fait que l'échantillonnage n'a pu rendre compte de la vulnérabilité des espèces présentes (par exemple en étudiant les modifications de leur physiologie ou de leur distribution spatiale) (Wilson et al., 2005; Halpern et al., 2007, 2008). De plus, l'initialisation des paramètres utilisés dans le modèle de diffusion, à savoir la distance d'influence maximale et les proportions en particules à sédimentation lente, intermédiaire et rapide, a été effectuée au moyen d'une concertation scientifique appuyée par une lecture d'articles spécialisés. Un tel processus a pu introduire un biais de protocole dans le calcul des scores d'exposition, et il est important d'augmenter le nombre d'experts questionnés sur ces paramètres pour rendre ces choix plus robustes. De plus, il est nécessaire d'intégrer des étapes de validation, telles qu'une analyse de la sensibilité des résultats en fonction d'une modification aléatoire des paramètres, ou la comparaison des résultats obtenus avec des jeux de données de référence. Dans le chapitre 3, la couverture spatiale de l'échantillonnage et le nombre élevé de stations ont permis d'utiliser un sous-ensemble de stations pour entrainer les modèles de distribution d'espèces basés sur l'exposition, ainsi qu'un modèle complémentaire permettant de valider les prédictions de façon indépendante. Ceci a montré une certaine efficacité du modèle dans le cas des espèces abondantes, quoique plus limitée pour les espèces rares, indiquant un potentiel intéressant pour cet indice.

Les résultats du *Hierarchical Modelling of Species Communities* ont permis de mettre en évidence les paramètres (variables de l'habitat et indices d'exposition) qui influencent la distribution des communautés. Une perspective intéressante serait d'utiliser ces modèles afin de prédire la structure des communautés en fonction de scénarios impactant les variables structurantes, tels que les changements climatiques ou des projets de développement anthropique. En utilisant les modèles actuels, il est par exemple possible de prédire la valeur de la richesse spécifique dans la zone industrialo-portuaire, dont un exemple se trouve à la Figure 23. La comparaison qualitative entre les valeurs prédites par le modèle et les valeurs observées aux stations échantillonnées montre une certaine correspondance, malgré une sous-estimation générale de la biodiversité. En adressant les recommandations présentées ci-dessus, un tel exercice a donc la possibilité d'être utilisé au sein d'évaluations environnementales dans le but de mieux comprendre les liens entre biodiversité et perturbation anthropique.

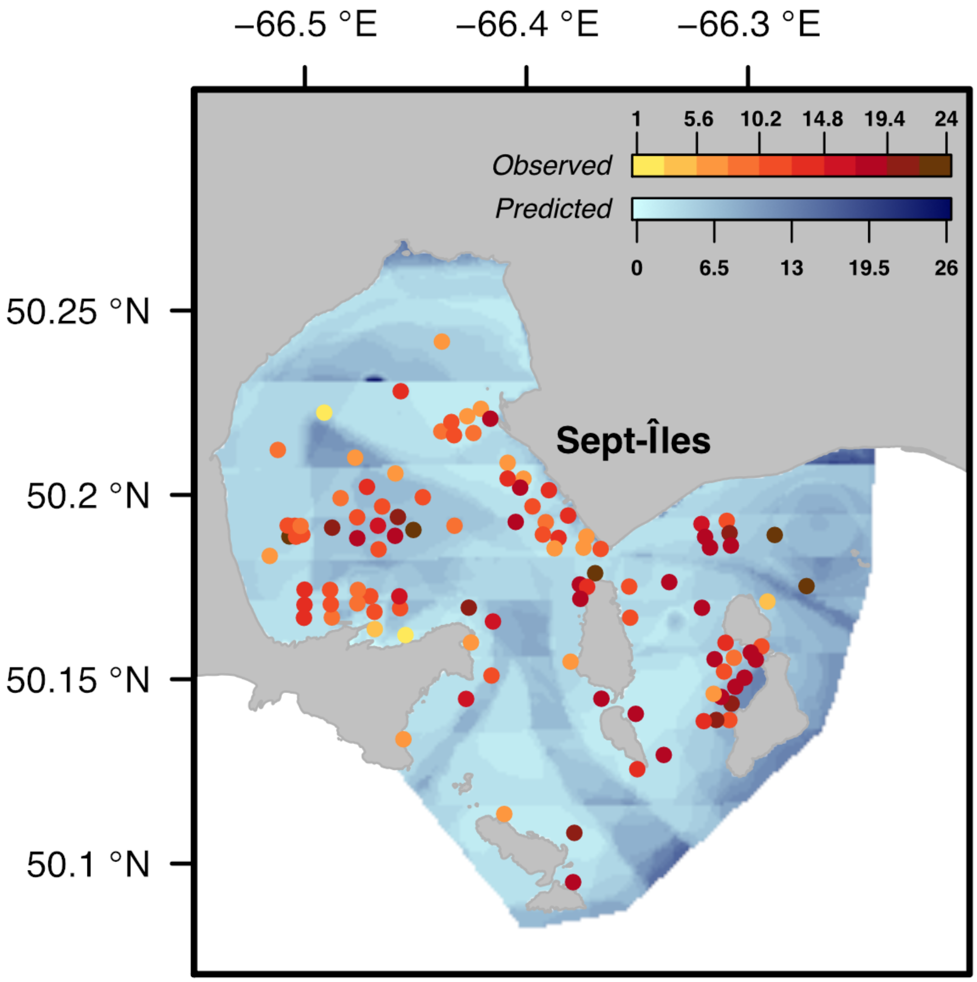


FIGURE 23 – Exemple de prédiction de la richesse spécifique basée sur les modèles *Hierarchical Modelling of Species Communities* utilisant les indices d'exposition comme prédicteurs. Le fond de carte correspond à l'exposition cumulée (bleu ciel = faible, indigo = élevée) et les points représentent les stations échantillonnées avec la richesse spécifique observée (jaune = faible, carmin = élevée).

L'exposition cumulée calculée dans cette étude a été basée sur la somme des indices d'exposition individuels. Cependant, de nombreux travaux ont considéré des réponses non-linéaires pour les impacts cumulés afin d'explorer des phénomènes émergents (Crain et al., 2008; Brown et al., 2014; Piggott et al., 2015; Côté et al., 2016; Galic et al., 2018). Il serait intéressant de tester l'influence que pourraient avoir d'autres types de relations entre les activités humaines sur nos résultats, comme les effets antagonistes (résultat inférieur à la somme des phénomènes individuels) ou synergiques (résultats supérieurs à cette somme). Ceci pourrait être réalisé en s'intéressant aux réponses physiologiques des espèces à plusieurs influences anthropiques simultannées, notamment en utilisant des expériences *in situ* ou en laboratoire. L'étude de Carrier-Belleau et al. (2021) constitue un exemple pertinent, car les interactions mises en évidence entre la température, la salinité et la concentration en nutriments sur la mortalité des bivalves considérés (*Limecola balthica* et *Mytilus* sp) permettent de mieux comprendre comment l'influence cumulée de plusieurs stresseurs se traduit à l'échelle de l'individu. En se basant sur ces résultats, il sera possible de s'intéresser aux effets sur les communautés et l'évolution des écosystèmes. De nombreux auteurs ont étudié ces liens dans différents types d'écosystèmes (lire par exemple Pearson et Rosenberg, 1978), et l'ajout de données en milieu côtier subarctique et anthropisé représenterait un ajout particulièrement pertinent pour de futurs projets.

Conclusion

Cette thèse a permis d'augmenter les connaissances sur les milieux côtiers subarctiques et de rendre compte des liens entre l'exposition anthropique et les écosystèmes benthiques à l'échelle locale. En plus d'apporter de nombreuses données sur les écosystèmes de la zone industrialo-portuaire de Sept-Îles, les méthodes d'évaluation du statut environnemental et de l'empreinte anthropique proposées ont la possibilité d'être appliquées à d'autres régions côtières dans le but d'obtenir des réseaux d'observation partageant des stratégies similaires.

Les questions écologiques étudiées durant ce doctorat s'inscrivent parfaitement dans la vision de la Décade 2021-2030 sur les Sciences de l'Océan pour un Développement Durable (United Nations, 2020) :

> *"The Decade is embracing a participative and transformative process so that scientists, policy makers, managers, and service users can work together to ensure that ocean science delivers greater benefits for both the ocean ecosystem and for society. This Decade will be designed to facilitate global communication and mutual learning across research and stakeholder communities. It will work to meet the needs of scientists, policy makers, industry, civil society and the wider public, but it will also support new, collaborative partnerships that can deliver more effective science-based management of our ocean space and resources."*

Forte d'initiatives telles que les *Aichi Biodiversity Targets* ou l'Accord de Paris sur le climat, l'humanité est dans une position favorable pour accroître sa compréhension des écosystèmes marins, condition *sine qua non* à une gestion environnementale efficace. Les réalisations découlant de ces travaux permettront d'améliorer l'intégration des communautés biologiques, habitats et activités anthropiques dans un ensemble cohérent et holistique.

ANNEXE I
APPROCHES ENVIRONNEMENTALES HOLISTIQUES ET OBJECTIFS D'AICHI POUR LA BIODIVERSITÉ : ACCOMPLISSEMENTS ET PERSPECTIVES POUR LES ÉCOSYSTÈMES MARINS

4.1 Résumé

Afin de préserver la biodiversité des changements mondiaux, la Conférence des Parties a élaboré un Plan Stratégique pour la Biodiversité pour la période 2011-2020 qui comprenait une liste de vingt objectifs spécifiques connus sous le nom d'objectifs d'Aichi pour la biodiversité. Cette période arrivant à sa fin, et malgré des progrès majeurs dans la conservation de la biodiversité, les preuves suggèrent que la majorité des objectifs ne seront probablement pas atteints. Cet article fait partie d'une série d'articles de perspective en lien avec la 4^{e} *World Conference on Marine Biodiversity* (mai 2018, Montréal, Canada) pour identifier les prochaines étapes vers une conservation réussie de la biodiversité en milieu marin. Nous avons considéré les études avec une approche environnementale holistique (AEH) et leur contribution à l'atteinte des objectifs. Notre analyse est axée sur différentes approches environnementales pouvant être considérées comme holistiques, et nous discutons de la manière dont les AEH peut contribuer aux objectifs d'Aichi pour la biodiversité à l'avenir. Nous avons constaté que seuls quelques AEH considéraient un objectif de biodiversité spécifique dans leurs recherches, et que l'objectif 11, qui porte sur les aires marines protégées, était le plus souvent cité. Nous proposons cinq priorités de recherche pour améliorer les AEH pour la conservation de la biodiversité marine au-delà de 2020 : (i) étendre l'utilisation d'approches holistiques

dans les évaluations environnementales, (ii) normaliser le vocabulaire utilisé par les AEH, (iii) améliorer la collecte, le partage et la gestion des données, (iv) envisager la variabilité spatio-temporelle des écosystèmes et (v) intégrer les services écosystémiques dans les AEH. La prise en compte de ces priorités favorisera la valeur des AEH et profitera au Plan Stratégique pour la Biodiversité.

L'article associé à ce chapitre, "*Holistic Environmental Approaches and Aichi Biodiversity Targets: accomplishments and perspectives for marine ecosystems*", a été co-rédigé avec Charlotte Carrier-Belleau, Jesica Goldsmit, Dario Fiorentino, Radhouane Ben-Hamadou, Jose H Muelbert, Jasmin A Godbold, Rémi M Daigle et David Beauchesne. Il a été publié dans le journal *PeerJ*, le 25 février 2020. Cet article d'opinion a été réalisé dans le cadre du programme de mentorat organisé lors de la 4e *World Conference on Marine Biodiversity*. J'ai dirigé l'équipe de rédaction, composée d'étudiants et de chercheurs (mentors scientifiques), pendant et après la conférence pour déterminer la direction des discussions, identifier les priorités de recherche et en discuter les implications, en lien avec les objectifs définis avec Jesica Goldsmit, Rémi M Daigle et David Beauchesne. J'ai effectué la revue de littérature, et j'ai dirigé la rédaction de l'article, où l'ensemble des co-auteurs a contribué à l'écriture des priorités de recherche et à la révision générale.

Dreujou, E., Carrier-Belleau, C., Goldsmit, J., Fiorentino, D., Ben-Hamadou, R., Muelbert, JH., Godbold, JA., Daigle, RM., Beauchesne, D. (2020). Holistic Environmental Approaches and Aichi Biodiversity Targets: accomplishments and perspectives for marine ecosystems. *PeerJ* 8:e817. DOI:10.7717/peerj.8171.

Les sections suivantes correspondent à celles de l'article publié.

HOLISTIC ENVIRONMENTAL APPROACHES AND AICHI BIODIVERSITY TARGETS: ACCOMPLISHMENTS AND PERSPECTIVES FOR MARINE ECOSYSTEMS

4.2 Abstract

In order to help safeguard biodiversity from global changes, the Conference of the Parties developed a Strategic Plan for Biodiversity for the period 2011-2020 that included a list of twenty specific objectives known as the Aichi Biodiversity Targets. With the end of that timeframe in sight, and despite major advancements in biodiversity conservation, evidence suggests that the majority of the Targets are unlikely to be met. This article is part of a series of perspective pieces from the 4th World Conference on Marine Biodiversity (May 2018, Montréal, Canada) to identify next steps towards successful biodiversity conservation in marine environments. We specifically reviewed holistic environmental assessment studies (HEA) and their contribution to reaching the Targets. Our analysis was based on multiple environmental approaches which can be considered as holistic, and we discuss how HEA can contribute to the Aichi Biodiversity Targets in the near future. We found that only a few HEA articles considered a specific Biodiversity Target in their research, and that Target 11, which focuses on marine protected areas, was the most commonly cited. We propose five research priorities to enhance HEA for marine biodiversity conservation beyond 2020: (i) expand the use of holistic approaches in environmental assessments, (ii) standardize HEA vocabulary, (iii) enhance data collection, sharing and management, (iv) consider ecosystem spatio-temporal variability and (v) integrate ecosystem services in HEA. The consideration of these priorities will promote the value of HEA and will benefit the Strategic Plan for Biodiversity.

Keywords: marine conservation, research priorities, holistic approaches, strategic plan for biodiversity, Aichi Biodiversity Targets

4.3 Introduction

In 2010, the 10th Conference of the Parties revised and updated the Strategic Plan for Biodiversity from the Convention on Biological Diversity (CBD), which included the Aichi Biodiversity Targets for 2011-2020 (Secretariat of the Convention on Biological Diversity, 2010). The mission of the Strategic Plan for Biodiversity is to "take effective and urgent action to halt the loss of biodiversity in order to ensure that by 2020 ecosystems are resilient and continue to provide essential services" (Secretariat of the Convention on Biological Diversity, 2010). According to the United Nations (1992), biodiversity refers to "the variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems". Yet, despite recent small- and large-scale conservation and management efforts, including the development of global protected area networks (Butchart et al., 2015), evidence suggests that most of the Targets are unlikely to be met as species declines and extinctions continue to occur (Secretariat of the Convention on Biological Diversity, 2014; Tittensor et al., 2014).

With the end of the Strategic Plan for Biodiversity in sight, the time is ripe to reflect on accomplishments thus far and to identify the next steps towards successful biodiversity conservation in marine ecosystems. These steps will be critical to meet the Sustainable Development Goal 14, which aims for the conservation and sustainable use of the oceans, seas, and marine resources by 2030 (SDG, 2020). These topics were tackled during the 4th World Conference on Marine Biodiversity held in Montréal, Canada, in May 2018, which gathered marine biodiversity experts from around the world. A mentoring program was devised to bring senior and early-career scientists together to address this challenge, which resulted in a series of perspective pieces, including this article. Holistic Environmental Approaches (HEA) were identified as crucial to marine biodiversity conservation by program participants.

In the present study, we define HEAs as environmental planning, assessment, management, or monitoring strategies that use a whole-system approach to explicitly consider and prioritize ecosystem complexity. Holism is dependent on components, connections, and boundaries of the considered ecosystems. While HEAs focus on natural ecosystems, they may include additional dimensions (*e.g.*, social, cultural and economic) relevant to the ecosystem under consideration. There is little doubt that the complexity of ecosystems must be considered for successful marine biodiversity conservation, yet the contribution of HEAs to marine biodiversity conservation in general and to the Aichi Biodiversity Targets in particular is unclear. In this perspective paper, we review the prevalence of HEAs in the peer-reviewed marine biodiversity literature and discuss their relevance to reaching the Aichi Biodiversity Targets, with a focus on the ecological dimension of HEAs. We then propose research priorities to enhance HEAs for marine biodiversity conservation beyond 2020.

4.4 Litterature review

4.4.1 Methodology

To better understand the uses of HEAs and their relevance for the Strategic Plan for Biodiversity, we searched the peer-reviewed scientific literature between January 1990 and July 2019 (inclusive). We used the *ISI Web of Knowledge database* and we queried the title, keywords, and abstract of original research articles. Non-peer reviewed literature, such as technical reports or assessment tools, were not included in this review as we considered that it could produce an important bias by the selection of studies related only to a specific region or for a specific use. We constrained our search to environmental studies by using the search terms *ecology*, *ecosystem*, *environment*, *habitat*, *species* and *biodiversity* as an initial filtering criteria (Table 17). We then further selected articles that focused on marine environments only.

A list of HEAs was established by gathering expert opinion from researchers in the field of marine ecology and environmental conservation. This process led to the inclusion of nine HEAs: *adaptive management* (Stankey et al., 2005), *cumulative impact assessment* (Jones, 2016), *ecosystem-based management* (Link, 2002; Pikitch et al., 2004; Levin et al., 2009), *integrated management* (Cicin-Sain and Belfiore, 2005), *marine spatial planning* (Santos et al., 2019), *social-ecological networks* (Baggio and Hillis, 2018), *strategic environmental assessment* (Gunn and Noble, 2009, 2011), *sustainable resource management* (Bringezu and Bleischwitz, 2009), and *systematic conservation planning* (Margules and Pressey, 2000). We used all HEA collectively as a search query on the initial corpus, then each HEA was queried individually to determine their prevalence in the literature (Table 17). Finally, we used the search term *Aichi Targets* in order to determine if and how Aichi Biodiversity Targets were considered in HEA studies.

Table 17 – Search terms used in the *ISI Web of Knowledge* to characterize the relevance of Holistic Environmental Approaches (HEAs) to achieving the Strategic Plan for Biodiversity. The different queries were limited from January 1990 to July 2019 (inclusive). Queries and search terms have been formatted with a regular expression syntax (REGEX) structured with conditional statements in italics, except for queries 2.x which have searched only for one type of HEA at a time.

ID	Query	Articles
1	Criteria AND HEAs	1,648
2	Criteria AND HEAs AND "marine"	505
2.1	Adaptive management	69
2.2	Cumulative impact assessment	2
2.3	Ecosystem-based management	223
2.4	Integrated management	43
2.5	Marine spatial planning	159
2.6	Social-ecological network	1
2.7	Strategic environmental assessment	5
2.8	Sustainable resource management	5
2.9	Systematic conservation planning	83
3	Criteria AND HEAs AND "marine" AND "Aichi"	12

Criteria: (ecolog∗ OR ecosystem OR environment∗ OR habitat OR species) AND "biodiversity" carriage return.

HEAs: "adaptive management" OR ("cumulative effect∗ assessment" OR "cumulative impact∗ assessment") OR "ecosystem.based management" OR ("integrated management" OR "integrative management") OR "marine spatial planning" OR "social.ecological network∗" OR "strategic environmental assessment" OR "sustainable resource management" OR "systematic conservation planning".

4.4.2 Prevalence of HEAs in the marine biodiversity literature

Our review identified 1,648 research articles related to biodiversity studies that used any of the identified HEAs, with 505 articles targeting marine environments. We found that the term *ecosystem-based management* was the most represented HEA (40.2%), followed by *marine spatial planning* (31.5%) (Figure 24). Other HEAs were less represented in the scientific literature, with *systematic conservation planning*, *adaptive management* and *integrative management* referred to in 16.4%, 13.7%, and 8.5% of the identified literature, respectively (Figure 24). Overall, few studies have considered multiple HEAs simultaneously, with 39 articles having the highest overlap between *ecosystem-based management* and *marine spatial planning*. When analyzing the keywords that were used in the reviewed articles, the most prevalent HEAs were "ecosystem-based management" and "marine spatial planning". Another common keyword was "marine protected areas", highlighting the relatively common use of this tool in marine conservation programs.

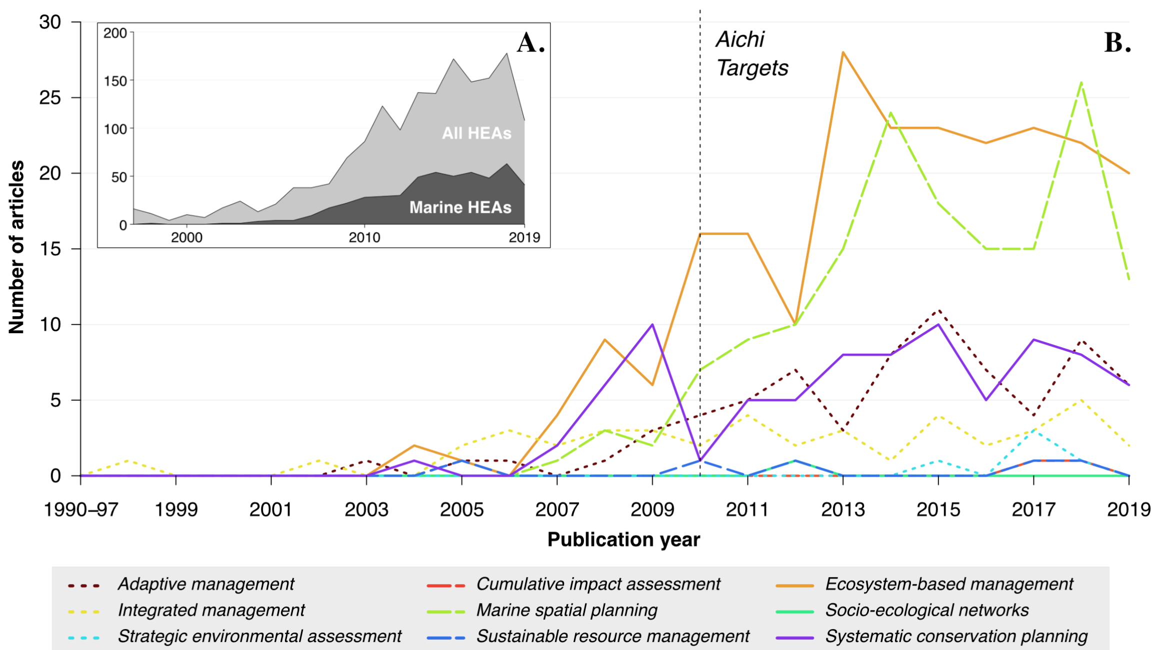


Figure 24 – Number of articles per year adopting a Holistic Environmental Approach (HEA) identified in *ISI Web of Knowledge*. (A) Number of HEA studies conducted in terrestrial, freshwater and marine environments (light grey), including studies focusing only on marine environments (dark grey). (B) Prevalence of each HEA within studies targeting marine environments only. Searches queried the title, abstract and keywords of peer-reviewed articles. Publication of the Aichi Biodiversity Targets in 2010 is represented by the black dashed vertical line.

The results show that HEAs were rarely discussed before 2006, and the number of HEA articles peaked in 2013, 2014, and 2018 (Figure 24). Overall, there was a steady increase in the number of HEA articles since 2000. This is particularly true for marine HEAs where the number of studies increased notably two years after the development of the Aichi Biodiversity Targets in 2010 (Figure 24). This increase after 2012 appears to be largely driven by a rise in the number of *ecosystem-based management* and *marine spatial planning* studies, which is likely a reflection of the time required for Aichi-related frameworks to be implemented in research supporting the management of socio-ecological systems (*e.g.*, White et al., 2010).

Of all the studies on HEAs, only 12 specifically used the term *Aichi Targets*, representing 2.4% of the papers originally identified (Table 18). This is a low proportion of HEAs contributing to the Strategic Plan for Biodiversity, even if we acknowledge that a study does not need to focus on a specific Target to allow a contribution. In addition, nine studies explicitly considered Targets in their research objectives (Table 18). The most frequently mentioned Aichi Biodiversity Target was Target 11, which aims for the conservation of 10% of coastal and marine areas by 2020 (Secretariat of the Convention on Biological Diversity, 2010). This Target is one of the few that specifically identifies quantitative thresholds for protected areas (Harrop, 2011), which supports the development of well specified and measurable objectives and tools such as simple, measurable, accurate, realistic, time-bound indices (SMART). HEAs could use SMART objectives, although there are few examples of their use in this context (Ehler, 2017). Specifying SMART objectives can be a difficult task, but their measurable component can highlight successful accomplishment of expected thresholds (Ehler, 2017, and references therin). Many studies selected in our literature review evaluated progress and developments of marine protected areas (*e.g.*, Amengual and Alvarez-Berastegui, 2018; Jantke et al., 2018; Rees et al., 2018). Target 11 has also been used to evaluate case studies (Diz et al., 2018), and to identify the sustainable use of specific marine protected areas as part of workshops and wider consultations (Johnson et al., 2014;

Sarker et al., 2019). Other selected studies considered either a specific Aichi Biodiversity Target, such as Target 12 in Davidson and Dulvy (2017) or Target 19 in Lagabrielle et al. (2014), or multiple Targets, such as Targets 1, 3, 6, and 17 in Cisneros-Montemayor et al. (2018) or Targets 6, 10, 11, and 12 in Davies et al. (2017) (Table 18). Five articles did not use Aichi Targets in their specific objectives, but were included to set the wider context of the article (*e.g.*, Lagabrielle et al., 2014; Yamakita et al., 2015; Davidson and Dulvy, 2017; Davies et al., 2017; Novaczek et al., 2017) (Table 18).

Table 18 – Links between articles adopting a Holistic Environmental Approach (HEA) obtained for Query 3 of the literature review and the Aichi Biodiversity Targets.

Article	Type of HEA considered	Targets considered	Targets as objectives?
Amengual and Alvarez-Berastegui (2018)	Marine spatial planning	11	Yes
Cisneros-Montemayor et al. (2018)	Adaptive management	1, 3, 6, 17	Yes
Davidson and Dulvy (2017)	Systematic conservation planning	11, 12	No
Davies et al. (2017)	Systematic conservation planning	6, 10, 11, 12	No
Diz et al. (2018)	Marine spatial planning	11	Yes
Jantke et al. (2018)	Systematic conservation planning	11	Yes
Johnson et al. (2014)	Ecosystem-based management	6, 11	Yes
Lagabrielle et al. (2014)	Marine spatial planning	11, 19	No
Novaczek et al. (2017)	Adaptive management	11	No
Rees et al. (2018)	Marine spatial planning	11	Yes
Sarker et al. (2019)	Integrated management	11	Yes
Yamakita et al. (2015)	Strategic environmental assessment	11	No

4.4.3 Linking HEAs and the strategic plan for biodiversity

Strategic Goals have been identified by the CBD as the steps necessary to safeguard biodiversity by 2020 (Figure 25A). These Goals include mainstreaming biodiversity across government and society (Goal A), reducing direct pressures on biodiversity (Goal B), improving the status of biodiversity (Goal C), enhancing benefits from biodiversity and ecosystem services (Goal D) and enhancing implementation of the established measures (Goal E) (Secretariat of the Convention on Biological Diversity, 2010). Aichi Biodiversity

Targets have been set within each Goal, with specific objectives or quantitative thresholds to reach (Figure 25B). Our literature review gathered a large number of HEA studies where a few referred to Targets in their objectives and methods (Table 17). Collectively, we found that these studies have focussed on eight Targets, with five being specified as objectives of the study (Table 18, Figure 25C). This provides examples of how HEAs can contribute to the Strategic Plan for Biodiversity while also providing feedback to reach specific Targets (Figure 25B-C). We will discuss some examples of these relationships in more detail in the section below.

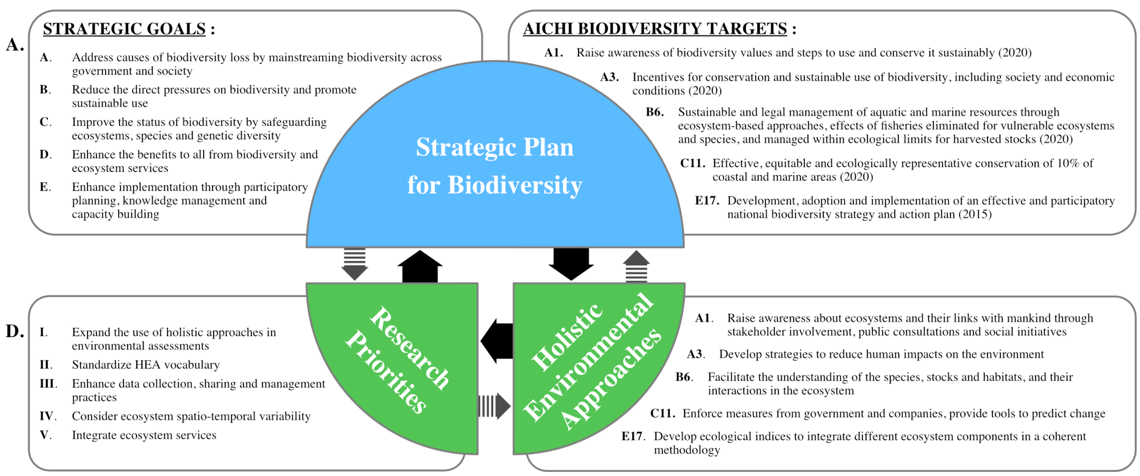


Figure 25 – Conceptual diagram of interactions and relationships between the Strategic Goals (A), the Aichi Biodiversity Targets (B), Holistic Environmental Approaches (C), and the identified research priorities (D). Targets have been summarized from Secretariat of the Convention on Biological Diversity (2010), and the letter before their number corresponds to the Goal to which they belong. Solid arrows represent direct relationships between sections, and dashed arrows represent secondary feedback.

Modern sustainable development objectives include minimizing cross-scale human impacts on biodiversity; concurrently, management plans are increasingly integrating social and economic dimensions (Intergovernmental Panel on Climate Change, 2014; Steffen et al., 2015). Thus, by also considering these same dimensions, HEAs explicitly include stakeholder involvement, public consultations or social initiatives, which is in accordance to Target 1. When made available to the public, the use of a whole-system approach within HEAs, in order to embrace ecosystem complexity, can raise awareness about biodiversity (Palerm, 2000; Portman, 2009; Jarvis et al., 2015). Implementation of conservation actions are usually complicated due to the variety of people concerned and the commercial interests of the different stakeholders (Margules and Pressey, 2000), but also because marine settings are particularly challenging, as stakeholders and objectives tend to be less well-defined (Cisneros-Montemayor et al., 2018). HEAs that take into account the natural variability of ecosystems, such as adaptive management or ecosystem-based management, should include social and political involvement (Stankey et al., 2005).

HEAs should also favor whole-system approaches to prioritize management actions based on ecosystem services, which relates to human use of environments (Carpenter et al., 2009; Chan and Ruckelshaus, 2010; Kareiva et al., 2011; Queiroz et al., 2015). Cumulative impact assessments, for example, focus on drivers of change and mechanistic pathways of impact in order to prioritize management efforts and take into account ecosystem services and thus, socio-economic dimensions (Brown et al., 2013; Cook et al., 2014; Cisneros-Montemayor et al., 2018). These approaches can be linked with Target 3's objectives to decrease negative effects on biodiversity and encourage conservation and sustainable use of biodiversity.

Target 6 states that fish and invertebrate stocks and aquatic plants are managed and harvested sustainably, legally, and applying ecosystem-based approaches, so that overfishing is avoided, recovery plans and measures are in place for all depleted species, fisheries have no significant adverse impacts on threatened species and vulnerable ecosystems, and the impacts of fisheries on stocks, species and ecosystems are within safe ecological limits (Secretariat of the Convention on Biological Diversity, 2010). HEAs, such as ecosystem-based management, resource management and adaptive management (along with all the processes linked to theses approaches) will provide the tools to a better understanding of the species, stocks, and habitats as well as their interactions in ecosystems (Arkema et al., 2006). These tools may be applied to a variety of concrete case studies, ranging from the conservation of marine mammals to coral reef protection (Maggs et al., 2013; Authier et al., 2017), but also to discuss the adequacy and performance of management strategies (Johnson et al., 2014; Cisneros-Montemayor et al., 2018).

With the aim of improving the status of biodiversity, governments and companies are required to enforce measures to safeguard ecosystems and all components therein (Secretariat of the Convention on Biological Diversity, 2010). In this context, HEAs can provide tools to accurately predict ecosystem consequences for systems threatened by multiple drivers of change (Nilsson and Dalkmann, 2001). For example, for Target 11 and the conservation of marine and coastal areas, HEAs have a direct contribution by being related and concerned with management, planning, and conservation. HEAs can also be helpful in the identification and assessment of threats by being able to manage the multiple and simultaneous drivers of change and stress.

The implementation of plans and strategies through participatory actions, such as proposed in Target 17, requires the production of concrete tools to manage environmental use. The correct implementation of HEAs can support the development of ecological indices to integrate different ecosystem components in a coherent methodology, since the need for operational tools within management plans has been highlighted (Arkema et al., 2006; Cisneros-Montemayor et al., 2018). In fact, these types of assessments are better undertaken when they are done strategically and expressed in a measurable way, *e.g.*, using SMART objectives (Jones, 2016).

4.5 Research priorities

HEAs should integrate all components of the studied ecosystems. However, logistical, technical and monetary considerations may limit the feasibility of such a goal. Nonetheless, 'partial' HEAs are often more valuable than specific environmental assessments (Jones, 2016). The complexity and breadth of knowledge needed for 'full' HEAs makes them exceedingly difficult to implement, which may likely explain the relatively small number of studies found applying holistic approaches to ecosystem management (Table 17). In order to achieve the goals set by the Strategic Plan for Biodiversity, there is a need to develop management actions beyond 2020 (Secretariat of the Convention on Biological Diversity, 2010). Discussions to identify the strategic direction for a post-2020 global biodiversity framework are taking place (*e.g.*, IX Trondheim Conference on Biodiversity), and the need for holistic management actions for a sustainable environment has been highlighted.

With this in mind, research priorities for the application of HEAs in marine environments were identified during the 4th World Conference on Marine Biodiversity as part of a mentoring program. Participants worked individually to identify research priorities before the conference, in order to provide a comprehensive list for the conference. This list was then used by participants to collectively curate a list of the top research priorities. This selection was discussed with conference attendees through panel discussions during the conference and comments were used to refine priorities post-conference. This process yielded a list of five research priorities (Figure 25D). The steps to undertake, in order to develop and promote the use of holistic approaches for marine biodiversity conservation, are discussed below.

4.5.1 Priority I: expand the use of holistic approaches in environmental assessments

Marine biodiversity spans different levels of biological organization (Hagen et al., 2012). The various biological components of a given ecosystem are continuously interacting with their environment within complex ecological networks. However, many environmental assessments focus on a single species or a single component of the ecosystem, overlooking important abiotic and biotic interactions that significantly affect the way organisms interact with their environment and mediate ecosystem functioning (Crain et al., 2008; Bulleri, 2009; van der Plas, 2019). Therefore, accurately assessing ecological functioning of marine ecosystems and their environmental, social and economic sustainability requires a holistic approach (Burton et al., 2014; Ma et al., 2017b).

Characterization of marine biodiversity and ecosystem functioning can be achieved through theoretical, numerical, experimental or monitoring approaches (Costello et al., 2017; Eriksen et al., 2018). Emerging environmental DNA techniques consisting of DNA metabarcoding and metagenomics (*e.g.*, Thomsen and Willerslev, 2015) offer potentially powerful new tools to monitor marine biodiversity and detect new species introductions. This allows reduced investment in traditional taxonomic techniques and biodiversity sampling and provides new opportunities to assess challenging and remote locations (Brown et al., 2016; Lacoursière-Roussel et al., 2018). Moreover, scientific research vessels now deploy vast arrays of equipment and gears simultaneously to answer increasingly complex research questions about whole ecosystems rather than as individual components (*e.g.*, Pesant et al., 2015). These new emerging methodologies and technologies can complement current holistic approaches such as cumulative effects assessments (Halpern et al., 2008, 2015) or systematic conservation planning (Margules and Pressey, 2000; Ball et al., 2009; Daigle et al., 2020). Managers increasingly recognize the need to shift towards holistic approaches to generate informed actions more inclusive of the relationships between ecosystem components than those obtained by traditional single- species efforts (*e.g.*, Manley et al., 2004; Beever, 2006).

The use of HEAs is relevant to all Targets within Goals B and C. In particular, expanding the use of holistic approaches could benefit Target 11's conservation objectives and perspectives by considering the complexity of the ecosystems (Rees et al., 2018).

4.5.2 Priority II: standardize HEAs vocabulary

What is a "driver of change", and when does it become a "stressor"? What constitutes an "impact"? The need to adopt a common vocabulary is especially important for multidisciplinary approaches in which communication between actors with a variety of backgrounds is often impeded by semantics (Holt et al., 2011). For example, the scientific community frequently uses the expression "cumulative effects assessment", but the underlying principles are often poorly understood, which may impact the interpretation of these assessments. Along with the definition of a concept, the origins behind the terminology must be explored and the terms standardized prior to their application across disciplines (Judd et al., 2015).

Analytical frameworks such as DPSIR (Drivers, Pressures, State, Impact, Response) models are useful for HEAs if all of the included elements are well defined and consistent (Kelble et al., 2013). However, Lewison et al. (2016) and Gari et al. (2014) found that despite the widespread application of individual terms across disciplines and projects, there is still no consensus on the definitions of "pressure" and "impact". These different interpretations decrease the understanding and operationalization of HEAs across scientists, stakeholders, and decision makers (Gari et al., 2014). The strengths of DPSIR frameworks, such as the capacity to describe linkages between human activity and environmental issues, encourage transdisciplinary research and will benefit many disciplines once its components are clarified (Kelble et al., 2013; Lewison et al., 2016).

While vocabulary standardization does not contribute directly to a specific Aichi Biodiversity Target, it will promote the applicability of HEAs by facilitating communication between actors, which could ultimately be advantageous to all Strategic Goals and Aichi Biodiversity Targets.

4.5.3 Priority III: enhance data collection, sharing, and management practices

The application of HEAs is highly dependent on efficient data collection, sharing, and management. However, constructing large datasets for holistic approaches is a challenging endeavour whose complexity is compounded by decentralized digital infrastructure and heterogenous practices (Wilkinson et al., 2016). To this end, we have identified three steps to promote data collection, sharing, and management efficiency for use in HEAs. Firstly, it is imperative to develop proper mechanisms to incentivize researchers to share their data publicly. In order to accelerate scientific discoveries and optimize research investments, many scientific journals and governmental agencies have initiated strong policies to promote public data archiving (Tenopir et al., 2011; Poisot et al., 2013; Roche et al., 2015). Regardless, many researchers remain reluctant to share their data publicly (Tenopir et al., 2011; Hampton et al., 2013; Roche et al., 2014), highlighting the lack of widespread mechanisms to give proper scientific value to data products (Wilkinson et al., 2016).

Secondly, our digital infrastructure should be improved so that data needed for HEAs are easily and openly accessible to all practitioners, scientists, and the public. We recommend adhering and promoting the FAIR Data Principles, which states that data must be Findable, Accessible, Interoperable, and Reusable (Wilkinson et al., 2016; Tanhua et al., 2019). This emerges as a crucial step to foster proper data management practices and to provide quality data and knowledge relevant to HEAs. Open-access data resources such as the Ocean Biodiversity Information System (OBIS, 2020) and the Global Biodiversity Information Facility (GBIF, 2020) exemplify excellent and easily accessible sources that can be used by researchers to share their data.

Finally, we should strive for global standardization of ocean practices. Defining clear standards and protocols will favour compatibility and pave the way towards efficient HEAs by facilitating the aggregation of local and regional datasets into large, holistic datasets. Initiatives that seek such standardization in practices, such as the Essential Ocean Variables from the Global Ocean Observing System (GOOS, 2020) and the Ocean Best Practices repository (OBP, 2020) from the International Oceanographic Data and Information Exchange, should thus be highly promoted.

Addressing these three steps will enhance data and protocol management, along with knowledge transfer and interoperability, which are necessary for efficient and robust HEAs. This will, in turn, facilitate education and outreach, management and conservation actions, evaluation of ecosystem services, data sharing and capacity building, which are the cornerstones of the Strategic Plan for Biodiversity.

4.5.4 Priority IV: consider ecosystem spatio-temporal variability

Ecosystem studies widely recognize the importance of spatial and temporal scales, as they influence ecosystem components (*e.g.*, fauna, flora), and characterize ecological processes (Legendre, 1993; Hagen et al., 2012; Pittman, 2017). Organism-environment interactions occur across a variety of spatio-temporal scales (*e.g.*, Legendre and Gauthier, 2014; Kraan et al., 2015; Yeager et al., 2017; Ryo et al., 2019), but only few HEA studies have acknowledged the need to consider these variations, for example by comparing different seasons or locations (*e.g.*, de la Vega et al., 2018b,a). Despite available methodologies to investigate spatio-temporal patterns within ecosystems (*e.g.*, Baselga, 2010; Legendre and Gauthier, 2014), we are unaware of environmental assessments that investigated multiple spatio-temporal structures concurrently in marine environments.

In addition to the organism-environment interactions, spatio-temporal structures can also affect human activities in an economic context. This can be seen with fisheries management, where activities occur across multiple spatio-temporal scales by involving single boat and fleet activities and managed to exploit targeted resources most efficiently (Hilborn, 2007; Watson et al., 2018). For example, tuna fisheries may be three times more profitable if fishing on strong oceanographic fronts (*i.e.*, Lagrangian coherent structures) (Watson et al., 2018). This implies that the effects of a physical feature of the water column can trickle through the local food web, ultimately affecting fisheries profitability at the spatial and temporal scale of the physical feature.

Human activities can interact directly and indirectly with a variety of natural drivers, such as shear stress, storms or currents, at different spatio-temporal scales (van Denderen et al., 2015; Watson et al., 2018). These interactions may trigger biodiversity responses that consequently appear at different levels of organization, influencing both faunal composition and functions that ultimately impact ecosystem functions and services. In order to develop successful conservation actions, HEAs require further understanding of the spatio-temporal structure of ecosystems and the scales of variability of related ecological patterns and processes, in order to adapt to their variability.

With respect to the Aichi Biodiversity Targets, assessing scales of spatio-temporal variability through HEAs will assist in reducing the impacts of human activities on ecosystems and species (Goal B), and to enhance management strategies to improve the status of critical areas (Goal C).

4.5.5 Priority V: integrate ecosystem services

The concept of "ecosystem services" has initiated the creation of a set of principles to be used by researchers and managers to support ecosystem conservation initiatives (de Groot et al., 2002; Beaumont et al., 2007). Ecosystem services are the benefits that humans gain from the natural environment (Millennial Environmental Assessment, 2005). They include provisioning (*e.g.*, production of food or raw materials), regulating (*e.g.*, water purification, carbon sequestration), supporting (*e.g.*, soil production, primary production) and cultural services (*e.g.*, aesthetic, recreation) (Beaumont et al., 2007; Fisher et al., 2009; Atkins et al., 2011; Balmford et al., 2011). Such services may be used to find compromises between providing a hospitable environment for human populations, maintaining ecosystem patterns, and processes within a sustainable range of variation (Beaumont et al., 2007; Cardinale et al., 2012; Norris, 2012). Because ecosystem services consider multiple aspects of the ecosystems within integrative frameworks, they will be highly relevant in HEAs.

Management and consideration of each ecosystem service category is often not equivalent within policy, resulting in a possible mismatch with environmental assessment in terms of spatio-temporal scales (Srivastava and Vellend, 2005; Cardinale et al., 2012). In order to use ecosystem services for biodiversity and ecosystem conservation, many ongoing discussions between stakeholders are seeking a common ground in their respective objectives and agendas (Seddon et al., 2016; Dee et al., 2017a). For example, Holt et al. (2011) quantified the types of services most valued by the local community and stakeholders in a coastal wetland and established the legislative mismatches that exist for protecting those ecosystem processes and functions that are necessary to support the valued benefits. This represents an important step towards integration of ecosystem services in frameworks like HEAs. While we acknowledge the complexity of these discussions and the ongoing research on the topic (*e.g.*, Paterson et al., 2011; Langhans et al., 2019), we emphasize that the integration of ecosystem services by stakeholders and within HEAs will provide a great tool for the Strategic Plan for Biodiversity. To this end, approaches considering ecosystems through network theory may be a great tool to consider the complexity of ecosystems with the plurality of human influences and services (Dee et al., 2017b).

Considering ecosystem services in HEAs will benefit the safeguarding of ecosystems and the maximization of benefits as stated in Goal D. The literature review detected an absence of HEA studies specifically including Targets of this Goal, which highlights the need to better link HEAs and ecosystem services.

4.6 Conclusion

Holistic environmental assessments have the potential to enhance marine conservation and management initiatives significantly beyond 2020. The use of HEAs has been increasing steadily over the past decade and is likely related to the establishment of the Strategic Plan for Biodiversity. To date, only a few studies refer to specific Aichi Biodiversity Targets in their research objectives. If included, HEAs could improve ecological research related to these Targets in a variety of ways: from the development of ecological indices and increased understanding of species-ecosystem interaction, to the provision of tools for the prediction of multiple drivers of change and helping the establishment of frameworks for citizen science. All these actions could simultaneously increase understanding of ecosystem complexity in management schemes and decision-making in order to achieve biodiversity goals.

We proposed five research priorities that could increase the effectiveness of HEAs in attaining the Aichi Biodiversity Targets, with respect to their current state of completion. Holistic approaches must appropriately assess the ecological functioning of marine ecosystems and their environmental, social and sustainable economic development. There is a need to standardize the vocabulary used for environmental assessments. Data collection needs to integrate system complexity and data management needs to follow recognized international standards. Marine biodiversity monitoring must consider single and multiple ecosystem components, must observe variability at different scales and should link biodiversity conservation to ecosystem services to support their sustainable uses.

Considering these priorities will help raise the value of HEAs to managers, ensuring greater accuracy and predictive power in environmental management, and could greatly help preparation of the work beyond the Strategic Plan for Biodiversity.

4.7 Acknowledgements

This work is the result of the 4th World Conference on Marine Biodiversity mentoring program. We thank Stephanie Allen, Karen Mooney, Lidia Lins Pereira and Hashim Said, who were involved in the initial discussions from which resulted in this work. We also wish to thank Peter Duinker and Natalie Ban for their help during the literature review, along with two anonymous reviewers for their helpful comments during earlier stages of this manuscript. Finally, we particularly thank Philippe Archambault, Anna Metaxas and Paul Snelgrove for their implication in the mentoring committee, who initiated this project, and their helpful comments and suggestions during the writing process.

ANNEXE II
ÉTUDE PILOTE SUR LES ÉCOSYSTÈMES INTERTIDAUX DANS LA RÉGION DE SEPT-ÎLES

5.1 Résumé

Cette section présente les jeux de données obtenus lors d'une étude exploratoire effectuée à l'été 2016 sur les écosystèmes intertidaux des Rivière Pentecôte (PR), Côte de Port-Cartier (CPC), Baie des Sept-Îles (BSI) et Rivière Manitou (MR). Des données sur les habitats (matière organique, pigments photosynthétiques, distribution des particules du sédiment) et la macrofaune (espèces supérieures à 1 mm) benthiques ont été récoltées, dans le but d'explorer la structure des écosystèmes et de fournir des références pour de futures études dans ces régions.

PILOT STUDY ON INTERTIDAL ECOSYSTEMS IN THE REGION OF SEPT-ÎLES

5.2 Abstract

This section presents datasets obtained during a pilot study in Summer 2016 on intertidal ecosystems of Pentecôte River (PR), Coast of Port-Cartier (CPC), Baie des Sept Îles (BSI) and Manitou River (MR). Data on benthic habitats (organic matter content, photosynthetic pigments concentration, grain-size analysis) and on macrofauna (individual higher than 1 mm) were gathered, in order to explore what is the structure of the ecosystems and to obtain references for future studies in these regions.

5.3 Objective of the study

This study aims to obtain baseline data on intertidal ecosystems in different regions around Sept-Îles, by sampling benthic habitats and communities.

5.4 Data collection

Five sectors were selected on the coastline of each considered region (Figure 26), at which three stations were randomly sampled within a 40 m transect. Stations were placed at *circa* 0.7 m above the tide-corrected waterline to ensure homogeneous communities.

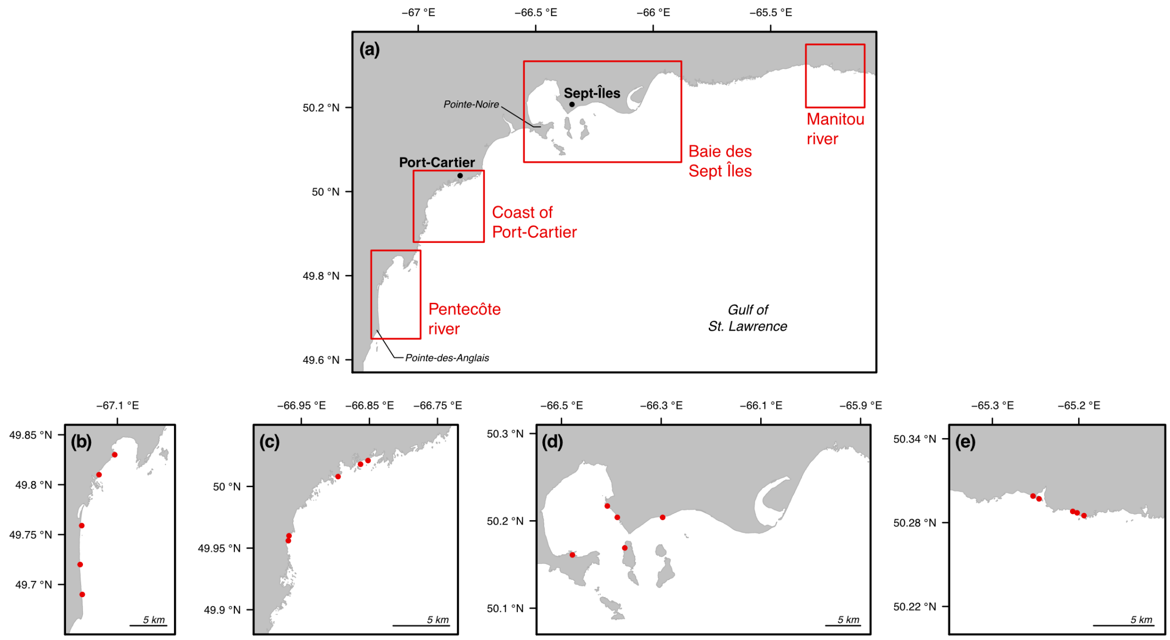


Figure 26 – Maps of the study area. (a) Location of the considered sectors; (b) Stations sampled in the Pentecôte River sector; (c) stations sampled in the coast of Port-Cartier sector; (d) stations sampled in Baie des Sept Îles sector; (e) stations sampled in the Manitou River sector.

At each station, a 0.25 m^2 quadrat was deployed and around 12 L of sediment was collected. This sediment was sorted on a 1 mm sieving mesh and retained individuals were preserved in a solution of BORAX-buffered formalin (4%). Around the quadrat, we collected three samples for the analyses of organic matter content, photosynthetic pigments and sediment grain-size, which were stored at -20 °C until processing in the laboratory.

The percentage of total organic matter (*i.e.*, sum of organic carbon and organic nitrogen) in the sediment was obtained by using the Loss-on-Ignition method (Davies, 1974). Concentrations of chlorophyll *a* and phaeopigments were obtained using a fluorimeter after pigment extraction in ethanol, according to the protocols of Dominique Lavallée at Institut des Sciences de la Mer (Rimouski) and Riaux-Gobin et al. (1989). Grain-size analysis was done on a sieving column for the fraction with particles larger than 2 mm and with a Laser Diffraction Particle Size Analyzer for the smaller fractions. Results from both techniques were combined to yield a unified distribution range from 0.04 μm to 26.5 mm. From this, percentages of gravel, sand, silt, and clay were calculated as defined by Wentworth (1922) and Folk (1980).

Samples for macrofauna identification were sorted using a stereomicroscope. Individuals were identified to the lowest taxonomic level possible with reference manuals and identification guides, and names were validated according to the World Register of Marine Species (WoRMS Editorial Board, 2020). Taxon density was recorded for each station by counting individuals collected per grab.

5.5 Datasets

5.5.1 Metadata

Table 19 – Metadata information for stations sampled during the intertidal campaign. ID is the unique identifier of each station used for the other datasets, height is calculated relative to the zero of nautical maps. NDD = northern decimal degrees, EDD = eastern decimel degrees.

ID	Station name	Region	Sampling date	Latitude (NDD)	Longitude (EDD)	Height (m)
1	PR_ST1_R1	Pentecôte River	2016-06-20	49.690083	-67.167467	0.71
2	PR_ST1_R2	Pentecôte River	2016-06-20	49.6904	-67.167433	0.73
3	PR_ST1_R3	Pentecôte River	2016-06-20	49.6905	-67.167417	0.7
4	PR_ST1_R4	Pentecôte River	2016-06-20	49.690533	-67.167433	0.64
5	PR_ST1_R5	Pentecôte River	2016-06-20	49.690667	-67.167433	0.67
6	PR_ST2_R1	Pentecôte River	2016-06-21	49.72775	-67.17125	0.73
7	PR_ST2_R2	Pentecôte River	2016-06-21	49.727633	-67.17125	0.7
8	PR_ST2_R3	Pentecôte River	2016-06-21	49.727717	-67.171283	0.7
9	PR_ST3_R1	Pentecôte River	2016-06-22	49.759717	-67.168533	0.68
10	PR_ST3_R2	Pentecôte River	2016-06-22	49.75985	-67.168533	0.69
11	PR_ST3_R3	Pentecôte River	2016-06-22	49.759983	-67.168533	0.72
12	PR_ST4_R1	Pentecôte River	2016-06-23	49.8109	-67.135183	0.71
13	PR_ST4_R2	Pentecôte River	2016-06-23	49.811133	-67.135117	0.61
14	PR_ST4_R3	Pentecôte River	2016-06-23	49.811217	-67.1351	0.67
15	PR_ST5_R1	Pentecôte River	2016-06-24	49.830767	-67.105417	0.68
16	PR_ST5_R2	Pentecôte River	2016-06-24	49.830817	-67.10545	0.68
17	PR_ST5_R3	Pentecôte River	2016-06-24	49.8309	-67.105467	0.71
18	MR_ST1_R1	Manitou River	2016-08-17	50.299367	-65.2538	0.67
19	MR_ST1_R2	Manitou River	2016-08-17	50.299233	-65.25365	0.7

ID	Station name	Region	Sampling date	Latitude (NDD)	Longitude (EDD)	Height (m)
20	MR_ST1_R3	Manitou River	2016-08-17	50.299167	-65.2536	0.71
21	MR_ST2_R1	Manitou River	2016-08-17	50.29715	-65.246483	0.72
22	MR_ST2_R2	Manitou River	2016-08-17	50.29715	-65.246483	0.72
23	MR_ST2_R3	Manitou River	2016-08-17	50.297117	-65.246167	0.75
24	MR_ST3_R1	Manitou River	2016-08-18	50.288533	-65.20665	0.7
25	MR_ST3_R2	Manitou River	2016-08-18	50.288583	-65.207	0.7
26	MR_ST3_R3	Manitou River	2016-08-18	50.288567	-65.206867	0.71
27	MR_ST4_R1	Manitou River	2016-08-18	50.287083	-65.202183	0.69
28	MR_ST4_R2	Manitou River	2016-08-18	50.287083	-65.202167	0.69
29	MR_ST4_R3	Manitou River	2016-08-18	50.28705	-65.2021	0.69
30	MR_ST5_R1	Manitou River	2016-08-16	50.285267	-65.19435	0.7
31	MR_ST5_R2	Manitou River	2016-08-16	50.285183	-65.194083	0.69
32	MR_ST5_R3	Manitou River	2016-08-16	50.285117	-65.193983	0.7
33	CPC_ST1_R1	Coast of Port-Cartier	2016-08-05	49.956633	-66.969367	0.68
34	CPC_ST1_R2	Coast of Port-Cartier	2016-08-05	49.956783	-66.969233	0.68
35	CPC_ST1_R3	Coast of Port-Cartier	2016-08-05	49.956817	-66.969233	0.69
36	CPC_ST2_R1	Coast of Port-Cartier	2016-08-03	50.008483	-66.896167	0.69
37	CPC_ST2_R2	Coast of Port-Cartier	2016-08-03	50.008583	-66.896083	0.69
38	CPC_ST2_R3	Coast of Port-Cartier	2016-08-03	50.00865	-66.89605	0.69
39	CPC_ST3_R1	Coast of Port-Cartier	2016-08-03	50.017983	-66.8638	0.67
40	CPC_ST3_R2	Coast of Port-Cartier	2016-08-03	50.018067	-66.863983	0.69
41	CPC_ST3_R3	Coast of Port-Cartier	2016-08-03	50.0181	-66.863983	0.71
42	CPC_ST4_R1	Coast of Port-Cartier	2016-08-04	50.020717	-66.852583	0.72
43	CPC_ST4_R2	Coast of Port-Cartier	2016-08-04	50.02075	-66.85255	0.73
44	CPC_ST4_R3	Coast of Port-Cartier	2016-08-04	50.020783	-66.852517	0.71
45	CPC_ST5_R1	Coast of Port-Cartier	2016-08-05	49.95945	-66.968	0.7
46	CPC_ST5_R2	Coast of Port-Cartier	2016-08-05	49.9595	-66.967983	0.69
47	CPC_ST5_R3	Coast of Port-Cartier	2016-08-05	49.9596	-66.967883	0.69
48	BSI_ST1_R1	Baie des Sept Îles	2016-07-20	50.160633	-66.478883	0.7
49	BSI_ST1_R2	Baie des Sept Îles	2016-07-20	50.160733	-66.478867	0.7
50	BSI_ST1_R3	Baie des Sept Îles	2016-07-20	50.160817	-66.478867	0.7
51	BSI_ST2_R1	Baie des Sept Îles	2016-07-24	50.217867	-66.408467	0.72
52	BSI_ST2_R2	Baie des Sept Îles	2016-07-24	50.217867	-66.408467	0.7
53	BSI_ST2_R3	Baie des Sept Îles	2016-07-24	50.217817	-66.4083	0.69
54	BSI_ST3_R1	Baie des Sept Îles	2016-07-23	50.204167	-66.387983	0.68
55	BSI_ST3_R2	Baie des Sept Îles	2016-07-23	50.204117	-66.387983	0.64
56	BSI_ST3_R3	Baie des Sept Îles	2016-07-23	50.204067	-66.38795	0.64
57	BSI_ST4_R1	Baie des Sept Îles	2016-07-22	50.169117	-66.37315	0.68
58	BSI_ST4_R2	Baie des Sept Îles	2016-07-22	50.1691	-66.3731	0.7
59	BSI_ST4_R3	Baie des Sept Îles	2016-07-22	50.168883	-66.3731	0.64
60	BSI_ST5_R1	Baie des Sept Îles	2016-07-19	50.204433	-66.297533	0.69
61	BSI_ST5_R2	Baie des Sept Îles	2016-07-19	50.204417	-66.29725	0.68
62	BSI_ST5_R3	Baie des Sept Îles	2016-07-19	50.20445	-66.297133	0.7

5.5.2 Habitat parameters

Table 20 – Habitat parameters at stations sampled during the intertidal campaign. ID refers to the unique station identifier used in Table 19 to link information between tables. OM = organic matter, Chla a = chlorophyll a, Phaeo = phaeopigments.

ID	OM (%)	Chl a (μg/g)	Phaeo (μg/g)	Gravel (%)	Sand (%)	Silt & Clay (%)
1	0.170254165	0.724639	0.0910011	0	1	0
2	0.147536146	0.906301	0.108756	0	1	0
3	0.172990428	1.20529	0.136393	0	1	0
4	0.1735107	0.986145	0.152852	0	1	0
5	0.204026115	1.12778	0.139498	0	1	0
6	0.184269227	0.373529	0.0538522	0	1	0
7	0.192270717	0.374648	0.0611087	0	1	0
8	0.158091406	0.307509	0.0501365	0	1	0
9	0.228957697	0.188597	0.0351598	0	1	0
10	0.211480363	0.196273	0.0401166	0	1	0
11	0.162683018	0.225632	0.0418445	0	1	0
12	0.219045423	1.10336	0.170026	0.0622382	0.937762	0
13	0.155697098	1.11875	0.160324	0	1	0
14	0.182799443	1.09852	0.149274	0	1	0
15	0.282271197	3.26944	0.414856	0	0.988695	0.0113052
16	0.130300877	3.02826	0.314379	0	0.983335	0.0166652
17	0.222196234	3.25628	0.438317	0	0.985987	0.014013
18	0.224	1.07371	0.111005	0	1	0
19	0.184297825	1.3449	0.157645	0	1	0
20	0.17455815	1.30606	0.0840853	0	1	0
21	0.208405696	1.00086	0.538239	0	1	0
22	0.200278649	1.42983	0.159539	0	1	0
23	0.09354537	1.50013	0.147093	0	1	0
24	0.056737589	2.51466	0.24176	0	1	0
25	0.057458056	2.42926	0.125234	0	1	0
26	0.029515939	2.47196	0.183497	0	1	0
27	0.137457045	2.08768	0.194564	0	1	0
28	0.067649844	2.09448	0.207155	0	1	0
29	0.054195646	1.93941	0.304992	0	1	0
30	0.160023276	2.61732	0.2122	0	1	0
31	0.166709413	2.50199	0.193257	0	1	0
32	0.1998002	2.44214	0.214908	0	1	0
33	0.845828527	2.20112	0.155546	0	1	0
34	0.189933523	1.9821	0.125533	0	1	0
35	0.190053849	2.36495	0.133784	0	1	0
36	0.14901279	1.23727	0.161582	0	1	0
37	0.166216497	1.06568	0.138332	0.0463085	0.953692	0
38	0.165772024	1.16165	0.159935	0	1	0
39	0.23923445	1.9924	0.304095	0.298525	0.641939	0.0595359
40	0.318794814	7.44382	2.47217	0	0.984977	0.0150228
41	0.398355179	5.24786	2.19768	0	0.972251	0.0277495
42	0.471225185	8.42282	1.74802	0	0.983485	0.0165153
43	0.434053467	7.65303	1.08279	0	0.989408	0.0105923

ID	OM (%)	Chl a (μg/g)	Phaeo (μg/g)	Gravel (%)	Sand (%)	Silt & Clay (%)
44	0.456323338	7.79113	1.10419	0	0.976679	0.0233213
45	0.216606498	2.5562	0.192048	0	1	0
46	0.228255646	2.60465	0.265335	0	1	0
47	0.218537087	2.57942	0.156342	0	1	0
48	0.71717705	1.01436	1.62405	0	0.166048	0.833952
49	0.7435653	0.72577	1.39594	0	3.51E-07	1
50	0.718051263	0.588036	1.39053	0	0.0110052	0.988995
51	0.232378002	2.26501	0.39856	0.0439861	0.870052	0.0859617
52	0.190497535	1.62219	0.313291	0.061164	0.914786	0.02405
53	0.175087544	2.21423	0.27113	0.059364	0.940636	0
54	0.210993892	0.771023	0.140497	0.210279	0.789721	0
55	0.280005091	0.813955	0.141085	0.144347	0.855653	0
56	0.194552529	1.91496	0.17602	0	1	0
57	0.157960059	0.972333	0.133434	0.13116	0.859811	0.00902819
58	0.224478888	0.64972	0.140853	0.220803	0.779197	0
59	0.235720762	1.0345	0.343567	0.325912	0.670333	0.0037547
60	0.187300993	1.83909	0.154193	0	1	0
61	0.145154087	1.18993	0.132369	0.0306736	0.969326	0
62	0.153806716	1.27531	0.134914	0	1	0
63	0.271403898	0.931408	0.161444	0	1	0
64	0.25390377	0.797018	0.153736	0.0187839	0.981216	0
65	0.228679042	0.705303	0.151392	0.0478096	0.95219	0
66	0.239275338	0.133881	0.028115	0.543316	0.456684	0
67	0.21293287	0.203464	0.047062	0.568153	0.428571	0.0032768
68	0.202354673	0.168988	0.0337339	0.517584	0.477716	0.00470014
69	0.428622455	0.0984226	0.0505498	0.560513	0.439487	0
70	0.27688048	0.106953	0.0298942	1	0	0
71	0.337552743	0.0962961	0.0212473	0.642971	0.357028	0
72	0.202558635	0.0976171	0.0214922	0.686753	0.310371	0.0028759
73	0.167732467	0.343566	0.0632044	0.500826	0.499174	0
74	0.231309376	0.129868	0.0293589	0.569432	0.430566	1.65E-06
75	0.353490721	0.0648362	0.0178126	1	0	0
76	0.227583068	0.185002	0.0487889	1	0	0
77	0.288808664	0.197223	0.226061	0.925495	0.0738898	0.00061484

5.5.3 Macrofauna

Table 21 – Density of macrofauna taxa at stations sampled during the intertidal campaign. ID refers to the unique station identifier used in Table 19 to link information between tables, density is in individual per quadrat.

ID	Phylum	Class	Order	Family	Accepted name	AphiaID	Density
1	Mollusca	Bivalvia		Mesodesmatidae	Mesodesma arctatum	156805	79
1	Nemertea				Nemertea	152391	1
1	Annelida	Polychaeta		Opheliidae	Ophelia limacina	130494	31
2	Arthropoda	Malacostraca	Amphipoda	Bathyporeiidae	Bathyporeia quoddyensis	158034	1
2	Mollusca	Bivalvia		Mesodesmatidae	Mesodesma arctatum	156805	90
2	Nemertea				Nemertea	152391	3
2	Annelida	Polychaeta		Opheliidae	Ophelia limacina	130494	92
2	Annelida	Polychaeta	Spionida	Spionidae	Spionidae	913	1
3	Arthropoda	Malacostraca	Amphipoda	Bathyporeiidae	Bathyporeia quoddyensis	158034	29
3	Mollusca	Bivalvia		Mesodesmatidae	Mesodesma arctatum	156805	74
3	Nemertea				Nemertea	152391	1
3	Annelida	Polychaeta		Opheliidae	Ophelia limacina	130494	64
3	Arthropoda	Malacostraca	Amphipoda	Tryphosidae	Wecomedon nobilis	1255501	10
4	Arthropoda	Malacostraca	Amphipoda	Bathyporeiidae	Bathyporeia quoddyensis	158034	39
4	Arthropoda	Malacostraca	Cumacea		Cumacea	1137	1
4	Mollusca	Bivalvia		Mesodesmatidae	Mesodesma arctatum	156805	80
4	Nemertea				Nemertea	152391	5
4	Annelida	Polychaeta		Opheliidae	Ophelia limacina	130494	109
4	Arthropoda	Malacostraca	Amphipoda	Tryphosidae	Wecomedon nobilis	1255501	4
5	Arthropoda	Malacostraca	Amphipoda	Bathyporeiidae	Bathyporeia quoddyensis	158034	59
5	Arthropoda	Malacostraca	Cumacea		Cumacea	1137	1
5	Mollusca	Bivalvia		Mesodesmatidae	Mesodesma arctatum	156805	184
5	Nemertea				Nemertea	152391	8
5	Annelida	Polychaeta		Opheliidae	Ophelia limacina	130494	145
6	Arthropoda	Malacostraca	Amphipoda	Bathyporeiidae	Bathyporeia quoddyensis	158034	20
6	Mollusca	Bivalvia		Mesodesmatidae	Mesodesma arctatum	156805	12
6	Nemertea				Nemertea	152391	1
6	Annelida	Polychaeta		Opheliidae	Ophelia limacina	130494	28
6	Arthropoda	Malacostraca	Amphipoda	Tryphosidae	Wecomedon nobilis	1255501	2
7	Arthropoda	Malacostraca	Amphipoda	Bathyporeiidae	Bathyporeia quoddyensis	158034	30
7	Mollusca	Bivalvia		Mesodesmatidae	Mesodesma arctatum	156805	17
7	Nemertea				Nemertea	152391	4
7	Annelida	Polychaeta		Opheliidae	Ophelia limacina	130494	13
7	Arthropoda	Malacostraca	Amphipoda	Tryphosidae	Wecomedon nobilis	1255501	1
8	Arthropoda	Malacostraca	Amphipoda	Bathyporeiidae	Bathyporeia quoddyensis	158034	32
8	Mollusca	Bivalvia		Mesodesmatidae	Mesodesma arctatum	156805	11
8	Nemertea				Nemertea	152391	2
8	Annelida	Polychaeta		Opheliidae	Ophelia limacina	130494	4
9	Arthropoda	Malacostraca	Amphipoda	Bathyporeiidae	Bathyporeia quoddyensis	158034	17
9	Mollusca	Bivalvia		Mesodesmatidae	Mesodesma arctatum	156805	458
9	Annelida	Polychaeta		Opheliidae	Ophelia limacina	130494	4
9	Arthropoda	Malacostraca	Amphipoda	Tryphosidae	Wecomedon nobilis	1255501	1
10	Arthropoda	Malacostraca	Amphipoda	Bathyporeiidae	Bathyporeia quoddyensis	158034	10
10	Mollusca	Bivalvia		Mesodesmatidae	Mesodesma arctatum	156805	311
10	Annelida	Polychaeta		Opheliidae	Ophelia limacina	130494	2

ID	Phylum	Class	Order	Family	Accepted name	AphiaID	Density
11	Arthropoda	Malacostraca	Amphipoda	Bathyporeiidae	Bathyporeia quoddyensis	158034	39
11	Mollusca	Bivalvia		Mesodesmatidae	Mesodesma arctatum	156805	183
11	Annelida	Polychaeta		Opheliidae	Ophelia limacina	130494	4
11	Arthropoda	Malacostraca	Amphipoda	Tryphosidae	Wecomedon nobilis	1255501	2
12	Arthropoda	Malacostraca	Amphipoda	Bathyporeiidae	Bathyporeia quoddyensis	158034	13
12	Mollusca	Bivalvia		Mesodesmatidae	Mesodesma arctatum	156805	50
12	Arthropoda	Malacostraca	Amphipoda	Tryphosidae	Wecomedon nobilis	1255501	2
13	Mollusca	Bivalvia		Mesodesmatidae	Mesodesma arctatum	156805	19
13	Arthropoda	Malacostraca	Amphipoda	Tryphosidae	Wecomedon nobilis	1255501	30
14	Mollusca	Bivalvia		Mesodesmatidae	Mesodesma arctatum	156805	41
15	Mollusca	Bivalvia	Cardiida	Tellinidae	Limecola balthica	880017	1
15	Mollusca	Bivalvia		Mesodesmatidae	Mesodesma arctatum	156805	4
15	Annelida	Polychaeta	Phyllodocida	Nephtyidae	Nephtys caeca	130355	5
15	Annelida	Polychaeta	Phyllodocida	Phyllodocidae	Phyllodocidae	931	1
15	Arthropoda	Malacostraca	Amphipoda	Tryphosidae	Wecomedon nobilis	1255501	17
16	Annelida	Polychaeta	Phyllodocida	Glyceridae	Glycera dibranchiata	157392	1
16	Mollusca	Bivalvia		Mesodesmatidae	Mesodesma arctatum	156805	3
16	Annelida	Polychaeta	Phyllodocida	Nephtyidae	Nephtys caeca	130355	2
16	Arthropoda	Malacostraca	Amphipoda	Tryphosidae	Wecomedon nobilis	1255501	13
17	Mollusca	Bivalvia		Mesodesmatidae	Mesodesma arctatum	156805	11
17	Arthropoda	Malacostraca	Amphipoda	Tryphosidae	Wecomedon nobilis	1255501	40
18	Arthropoda	Malacostraca	Amphipoda	Bathyporeiidae	Bathyporeia quoddyensis	158034	33
18	Mollusca	Bivalvia		Mesodesmatidae	Mesodesma arctatum	156805	45
18	Annelida	Polychaeta		Opheliidae	Ophelia limacina	130494	18
18	Arthropoda	Malacostraca	Amphipoda	Tryphosidae	Wecomedon nobilis	1255501	1
19	Arthropoda	Malacostraca	Amphipoda	Bathyporeiidae	Bathyporeia quoddyensis	158034	104
19	Mollusca	Bivalvia		Mesodesmatidae	Mesodesma arctatum	156805	143
19	Annelida	Polychaeta		Opheliidae	Ophelia limacina	130494	29
19	Arthropoda	Malacostraca	Amphipoda	Tryphosidae	Wecomedon nobilis	1255501	1
20	Arthropoda	Malacostraca	Amphipoda	Bathyporeiidae	Bathyporeia quoddyensis	158034	19
20	Mollusca	Bivalvia		Mesodesmatidae	Mesodesma arctatum	156805	179
20	Annelida	Polychaeta		Opheliidae	Ophelia limacina	130494	20
20	Arthropoda	Malacostraca	Amphipoda	Talitridae	Orchestia grillus	158123	1
21	Arthropoda	Malacostraca	Amphipoda	Bathyporeiidae	Bathyporeia quoddyensis	158034	23
21	Mollusca	Bivalvia		Mesodesmatidae	Mesodesma arctatum	156805	103
21	Annelida	Polychaeta		Opheliidae	Ophelia limacina	130494	31
21	Arthropoda	Malacostraca	Amphipoda	Tryphosidae	Wecomedon nobilis	1255501	1
22	Arthropoda	Malacostraca	Amphipoda	Bathyporeiidae	Bathyporeia quoddyensis	158034	28
22	Mollusca	Bivalvia		Mesodesmatidae	Mesodesma arctatum	156805	118
22	Annelida	Polychaeta		Opheliidae	Ophelia limacina	130494	36
23	Arthropoda	Malacostraca	Amphipoda	Bathyporeiidae	Bathyporeia quoddyensis	158034	1
23	Mollusca	Bivalvia		Mesodesmatidae	Mesodesma arctatum	156805	152
23	Annelida	Polychaeta		Opheliidae	Ophelia limacina	130494	5
24	Arthropoda	Malacostraca	Amphipoda	Bathyporeiidae	Bathyporeia quoddyensis	158034	35
24	Mollusca	Bivalvia		Mesodesmatidae	Mesodesma arctatum	156805	333
24	Annelida	Polychaeta		Opheliidae	Ophelia limacina	130494	47
24	Arthropoda	Malacostraca	Amphipoda	Tryphosidae	Wecomedon nobilis	1255501	1
25	Arthropoda	Malacostraca	Amphipoda	Bathyporeiidae	Bathyporeia quoddyensis	158034	21
25	Mollusca	Bivalvia		Mesodesmatidae	Mesodesma arctatum	156805	186
25	Annelida	Polychaeta		Opheliidae	Ophelia limacina	130494	65
26	Arthropoda	Malacostraca	Amphipoda	Bathyporeiidae	Bathyporeia quoddyensis	158034	17
26	Mollusca	Bivalvia		Mesodesmatidae	Mesodesma arctatum	156805	160
26	Annelida	Polychaeta		Opheliidae	Ophelia limacina	130494	42
27	Mollusca	Bivalvia		Mesodesmatidae	Mesodesma arctatum	156805	94

ID	Phylum	Class	Order	Family	Accepted name	AphiaID	Density
27	Annelida	Polychaeta		Opheliidae	Ophelia limacina	130494	13
28	Arthropoda	Malacostraca	Amphipoda	Bathyporeiidae	Bathyporeia quoddyensis	158034	1
28	Mollusca	Bivalvia		Mesodesmatidae	Mesodesma arctatum	156805	154
28	Annelida	Polychaeta		Opheliidae	Ophelia limacina	130494	20
29	Arthropoda	Malacostraca	Amphipoda	Bathyporeiidae	Bathyporeia quoddyensis	158034	1
29	Mollusca	Bivalvia		Mesodesmatidae	Mesodesma arctatum	156805	160
29	Annelida	Polychaeta		Opheliidae	Ophelia limacina	130494	19
30	Mollusca	Bivalvia	Cardiida	Tellinidae	Limecola balthica	880017	1
30	Mollusca	Bivalvia		Mesodesmatidae	Mesodesma arctatum	156805	28
30	Annelida	Polychaeta		Opheliidae	Ophelia limacina	130494	1
30	Arthropoda	Malacostraca	Amphipoda	Pleustidae	Pleusymtes glaber	103020	1
30	Arthropoda	Malacostraca	Amphipoda	Tryphosidae	Wecomedon nobilis	1255501	14
31	Arthropoda	Malacostraca	Amphipoda	Bathyporeiidae	Bathyporeia quoddyensis	158034	3
31	Mollusca	Bivalvia		Mesodesmatidae	Mesodesma arctatum	156805	28
31	Annelida	Polychaeta		Opheliidae	Ophelia limacina	130494	3
31	Arthropoda	Malacostraca	Amphipoda	Tryphosidae	Wecomedon nobilis	1255501	5
32	Echinodermata	Echinoidea	Clypeasteroida	Echinarachniidae	Echinarachnius parma	158062	1
32	Mollusca	Bivalvia		Mesodesmatidae	Mesodesma arctatum	156805	22
32	Arthropoda	Malacostraca	Amphipoda	Tryphosidae	Wecomedon nobilis	1255501	12
33	Mollusca	Bivalvia		Mesodesmatidae	Mesodesma arctatum	156805	39
33	Annelida	Polychaeta		Opheliidae	Ophelia limacina	130494	7
33	Arthropoda	Malacostraca	Amphipoda	Tryphosidae	Wecomedon nobilis	1255501	4
34	Mollusca	Bivalvia		Mesodesmatidae	Mesodesma arctatum	156805	42
34	Annelida	Polychaeta		Opheliidae	Ophelia limacina	130494	1
35	Mollusca	Bivalvia		Mesodesmatidae	Mesodesma arctatum	156805	33
35	Annelida	Polychaeta		Opheliidae	Ophelia limacina	130494	4
35	Arthropoda	Malacostraca	Amphipoda	Tryphosidae	Wecomedon nobilis	1255501	4
36	Mollusca	Bivalvia	Cardiida	Tellinidae	Limecola balthica	880017	1
36	Mollusca	Bivalvia	Myida	Myidae	Mya arenaria	140430	1
37	Mollusca	Bivalvia	Cardiida	Tellinidae	Limecola balthica	880017	27
37	Annelida	Polychaeta	Spionida	Spionidae	Marenzelleria viridis	131135	1
37	Mollusca	Bivalvia	Myida	Myidae	Mya arenaria	140430	2
38	Annelida	Polychaeta	Phyllodocida	Phyllodocidae	Eteone longa	130616	2
38	Arthropoda	Malacostraca	Amphipoda	Gammaridae	Gammarus	101537	5
38	Annelida	Polychaeta	Phyllodocida	Nereididae	Hediste diversicolor	152302	5
38	Mollusca	Bivalvia	Cardiida	Tellinidae	Limecola balthica	880017	29
38	Mollusca	Bivalvia		Mesodesmatidae	Mesodesma arctatum	156805	3
38	Mollusca	Bivalvia	Myida	Myidae	Mya arenaria	140430	2
38	Annelida	Polychaeta	Spionida	Spionidae	Spionidae	913	9
39	Mollusca	Bivalvia		Mesodesmatidae	Mesodesma arctatum	156805	7
40	Mollusca	Bivalvia		Mesodesmatidae	Mesodesma arctatum	156805	13
41	Mollusca	Bivalvia		Mesodesmatidae	Mesodesma arctatum	156805	11
42	Annelida	Polychaeta	Phyllodocida	Phyllodocidae	Eteone longa	130616	2
42	Mollusca	Bivalvia		Mesodesmatidae	Mesodesma arctatum	156805	7
42	Nemertea				Nemertea	152391	6
42	Annelida	Clitellata			Oligochaeta	2036	5
42	Arthropoda	Malacostraca	Amphipoda	Tryphosidae	Wecomedon nobilis	1255501	21
43	Mollusca	Bivalvia	Cardiida	Tellinidae	Limecola balthica	880017	2
43	Mollusca	Bivalvia		Mesodesmatidae	Mesodesma arctatum	156805	20
43	Nemertea				Nemertea	152391	1
43	Annelida	Clitellata			Oligochaeta	2036	1
43	Arthropoda	Malacostraca	Amphipoda	Tryphosidae	Wecomedon nobilis	1255501	18
43	Annelida	Polychaeta	Spionida	Spionidae	Spionidae	913	2
44	Arthropoda	Malacostraca	Amphipoda	Gammaridae	Gammarus	101537	1

ID	Phylum	Class	Order	Family	Accepted name	AphiaID	Density
44	Mollusca	Bivalvia		Mesodesmatidae	Mesodesma arctatum	156805	16
44	Annelida	Polychaeta	Phyllodocida	Nephtyidae	Nephtys caeca	130355	1
44	Arthropoda	Malacostraca	Amphipoda	Tryphosidae	Wecomedon nobilis	1255501	1
44	Annelida	Polychaeta	Spionida	Spionidae	Spionidae	913	1
45	Mollusca	Bivalvia		Mesodesmatidae	Mesodesma arctatum	156805	32
45	Nemertea				Nemertea	152391	2
45	Arthropoda	Malacostraca	Amphipoda	Tryphosidae	Wecomedon nobilis	1255501	12
46	Mollusca	Bivalvia		Mesodesmatidae	Mesodesma arctatum	156805	28
46	Arthropoda	Malacostraca	Amphipoda	Tryphosidae	Wecomedon nobilis	1255501	17
47	Mollusca	Bivalvia		Mesodesmatidae	Mesodesma arctatum	156805	42
47	Arthropoda	Malacostraca	Amphipoda	Tryphosidae	Wecomedon nobilis	1255501	12
48	Annelida	Polychaeta	Phyllodocida	Glyceridae	Glycera dibranchiata	157392	5
48	Annelida	Polychaeta	Phyllodocida	Nereididae	Hediste diversicolor	152302	1
48	Mollusca	Bivalvia	Cardiida	Tellinidae	Limecola balthica	880017	105
48	Mollusca	Bivalvia	Myida	Myidae	Mya arenaria	140430	54
49	Annelida	Polychaeta	Phyllodocida	Glyceridae	Glycera dibranchiata	157392	3
49	Annelida	Polychaeta	Phyllodocida	Nereididae	Hediste diversicolor	152302	2
49	Mollusca	Bivalvia	Cardiida	Tellinidae	Limecola balthica	880017	89
49	Mollusca	Bivalvia	Myida	Myidae	Mya arenaria	140430	80
50	Annelida	Polychaeta	Phyllodocida	Glyceridae	Glycera dibranchiata	157392	1
50	Annelida	Polychaeta	Phyllodocida	Nereididae	Hediste diversicolor	152302	2
50	Mollusca	Bivalvia	Cardiida	Tellinidae	Limecola balthica	880017	106
50	Mollusca	Bivalvia	Myida	Myidae	Mya arenaria	140430	74
51	Mollusca	Bivalvia	Cardiida	Tellinidae	Limecola balthica	880017	4
56	Mollusca	Bivalvia		Mesodesmatidae	Mesodesma arctatum	156805	2
57	Mollusca	Bivalvia		Mesodesmatidae	Mesodesma arctatum	156805	18
58	Arthropoda	Malacostraca	Amphipoda	Gammaridae	Gammarus oceanicus	102285	44
58	Mollusca	Bivalvia		Mesodesmatidae	Mesodesma arctatum	156805	70
59	Arthropoda	Malacostraca	Amphipoda	Gammaridae	Gammarus oceanicus	102285	2
59	Mollusca	Bivalvia		Mesodesmatidae	Mesodesma arctatum	156805	46
60	Arthropoda	Malacostraca	Amphipoda	Bathyporeiidae	Bathyporeia quoddyensis	158034	1
60	Mollusca	Bivalvia		Mesodesmatidae	Mesodesma arctatum	156805	69
60	Annelida	Polychaeta		Opheliidae	Ophelia limacina	130494	3
60	Arthropoda	Malacostraca	Amphipoda	Tryphosidae	Wecomedon nobilis	1255501	27
61	Mollusca	Bivalvia		Mesodesmatidae	Mesodesma arctatum	156805	77
61	Arthropoda	Malacostraca	Amphipoda	Tryphosidae	Wecomedon nobilis	1255501	4
62	Mollusca	Bivalvia		Mesodesmatidae	Mesodesma arctatum	156805	58
62	Arthropoda	Malacostraca	Amphipoda	Tryphosidae	Wecomedon nobilis	1255501	3

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