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A National Monitoring Framework for Coral and Sponge Areas Identified as Other Effective Area-Based Conservation Measures

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ABSTRACT

Canada has made national and international commitments to protect 25% of our marine and coastal areas by 2025, working toward 30% by 2030. Fisheries and Oceans Canada (DFO) uses Marine Protected Areas and Other Effective Area-based Conservation Measures (OECMs) to protect significant areas of our oceans. OECMs are areas that are conserved to achieve a biodiversity conservation benefit (BCB). These BCBs can be direct, meaning the species or habitat that are being targeted for protection, or indirect, which are the additional benefits occurring because of the protection measure. Many of Canada’s OECMs are areas containing significant aggregations and populations of cold-water corals and sponges and are protected using fisheries closures (also known as “marine refuges”). Now that these areas are protected, DFO needs to effectively monitor them to ensure that the direct and indirect BCBs are being maintained or improved. In this paper, we summarize information on Canada’s corals and sponge OECMs, including baseline information and knowledge gaps. We provide a list of known coral and sponge groups for each OECM and we suggest that using those groups can be helpful when developing monitoring plans. Indirect BCBs of corals and sponge habitat are described and reviewed for each group. Indicators to monitor corals and sponges and their associated BCBs are suggested. We describe the appropriateness of several state (ecosystem and environment), and pressure (stress) indicators for corals and sponges. Lastly, we examine various tools and methodologies that can be used to study these coral and sponge habitats. The feasibility and appropriateness of many research tools are explained, from commonly used methods such as trawling and imagery technologies, to emerging techniques like environmental DNA. Considerations and best practices for designing a monitoring program, such as the use of baseline data, important statistical consideration, and details on sampling design, are reviewed. We suggest that monitoring should focus on the direct BCBs, the corals and sponges, and that researchers can use targeted studies to confirm indirect BCBs. The information in this paper is presented in support of a Canadian Science Advisory process that took place December 1-3, 2020. The information contained in this report can be used by researchers and managers to develop appropriate and effective monitoring plans and strategies for coral and sponge OECMs. In addition, this framework has potential application for existing and proposed MPAs with cold-water coral and sponge conservation objectives.

1. INTRODUCTION

1.1. COLD-WATER CORALS AND SPONGES

Corals are benthic cnidarians in the classes Anthozoa and Hydrozoa which produce calcium carbonate and/or proteinaceous skeletons (Cairns 2007). In Canada, cold-water corals are found in the Atlantic, Pacific, and central and eastern Arctic Oceans, with over 120 reported species, and likely many more not yet identified (DFO 2010a). Key groups in Canada include reef-building corals (e.g., *Lophelia pertusa*) and cup corals in the Order Scleractinia, hydrocorals (Order Anthoathecata), black corals (Order Antipatharia), gorgonians (Order Alcyonacea), soft corals (Order Alcyonacea), and sea pens (Order Pennatulacea) (Jamieson et al. 2007, Wareham and Edinger, 2007; Kenchington et al. 2016). Cold-water corals mostly differ from their tropical relatives in that most do not live in symbiosis with zooxanthellae, and therefore do not require light for energy production. They feed mainly by capturing detritus and re-suspended particles, with some species feeding on benthic meiofauna and zooplankton (Campbell and Simms, 2009; Salvo et al., 2018; Sherwood et al., 2011, Sherwood et al. 2008).

Corals in Canada can form biogenic habitat by creating complex, three-dimensional structures on the seafloor. Scleractinian corals secrete calcium carbonate to build their skeletons (Stolarski et al., 2011), while black corals produce exclusively proteinaceous skeletons (Wagner et al. 2012). Many cold-water corals grow in patchy formations or in aggregations of individuals, also called fields or gardens (e.g., Boutillier et al., 2010; Long et al., 2020; Roberts et al., 2009). For reef-building corals, a reef framework is formed when older polyps die and the coral skeleton becomes subject to bioerosion or breakage (Roberts et al., 2006). These pieces of skeleton cause a build-up of sediment, which provide substrate for living corals to grow and form structures called bioherms (Boutillier et al., 2010).

Many cold-water corals are long-lived, with *Lophelia pertusa* (*Desmophyllum pertusum*) reefs estimated to be over 8,000 years old (Roberts et al., 2006). Non-reef forming corals can also attain high longevities, such as certain black corals which have been shown to achieve longevities of >4,000 years (Roark et al. 2009). Certain gorgonians and cup corals have been aged to 100-200 years and older (Mortensen and Buhl-Mortensen, 2005; Sherwood and Edinger, 2009; Boutillier et al., 2010), and some pens can live for decades (Wilson et al., 2002; Neves et al., 2015a; Murillo et al., 2018a), In addition, they can be slow growing, with many species growing <1 cm/year (Sherwood and Edinger, 2009). Growth and age in cold-water corals can vary by taxa and are likely influenced by environmental factors such as depth, temperature, current, and food availability (Boutillier et al., 2010).

Sponges are sessile filter-feeding animals that pump water through their bodies to obtain food and oxygen, and to excrete waste. This seemingly simple mode of life plays a key role in the coupling of pelagic productivity and the benthos (Dayton et al., 1974; Kahn et al., 2015). Sponges use flagellated cells called choanocytes to create a unidirectional flow of water through their bodies, removing bacteria, organic particulates, and dissolved organic carbon from the water (Reiswig, 1975; Yahel et al., 2003; de Goeij et al., 2008; Kutti et al., 2013). Some sponges have lost the water pumping aquiferous system altogether and have developed a carnivorous feeding strategy, which allows them to compensate for limited particulate and dissolved organic carbon in their given habitat (Vacelet and Boury-Esnault, 1995). There are more than 9000 species of sponge described worldwide (Van Soest, 2020). This vast diversity of species results in marked differences in body form, reproductive method, preferred habitat, and lifespan (Hughes et al., 1992; Connell et al., 1997; Anthony, 1999; Hooper and Van Soest, 2002). Sponges are asymmetrical and grow in three dimensions, so accurately assessing growth rates of living individuals over a time series is difficult (Kahn et al., 2016). Efforts to age sponges through direct measurements over time (Kahn et al., 2016; Leys & Lauzon, 1998) and radiocarbon dating (Fallon et al., 2010) have estimated the age of some glass sponges to be several hundred years old, and growth rates range from 2.9 mm/year to 5.7 cm/year.

Sponges can have very elaborate and diverse body forms, with many variations in shape and appearance occurring within taxonomic groups (Hooper and Van Soest, 2002). Sponges occur as thin crusts, fan-like and digitate forms which are presumed to take advantage of ambient currents, or massive sponges which can occur in dense aggregations (Klitgaard and Tendal, 2004; Klitgaard, 1995), and some species can even form dense, multi-species reefs (Chu and Leys, 2010; Conway et al., 1991; Krautter et al., 2001; Leys et al., 2004; Maldonado et al., 2017). Variations in sponge shape are therefore quite common, leading to a multitude of sponge growth forms – even within individual species (Hooper and Van Soest, 2002). A given habitat can affect the growth form of a sponge, particularly where flow regimes can be exploited to obtain food. For instance, the cost of pumping can be high for some reef-forming glass sponges (Class Hexactinellida), so reefs in areas of high ambient flow allow sponges to have their food source replenished and to more effectively remove wastes (Leys et al., 2011).

1.2. PROTECTION AND MONITORING OF COLD-WATER CORALS AND SPONGES IN CANADA

By the end of the 2020, Canada, through its commitments to national and international marine conservation targets (MCT), has protected 13.81% of its marine and coastal areas through the establishment of marine protected areas (MPAs) and other effective area-based conservation measures (OECMs). Fisheries and Oceans Canada (DFO) developed guidance for identifying OECMs in 2016 (DFO, 2016). In 2018, the Convention on Biological Diversity (CBD) Conference of the Parties, which includes Canada, adopted voluntary international guidance for identifying OECMs. DFO is in the process of finalizing guidance and criteria based on the 2016 CSAS advice and 2018 CBD voluntary guidance for the establishment and management of OECMs (DFO 2016; CBD 2018). DFO has also taken steps to conserve benthic ecosystems through its “Policy to manage the impacts of fishing on sensitive benthic areas” (DFO, 2009). At the time of writing, Canada has 59 OECMs, 40 of which are established with the conservation objective to protect cold-water corals and/or sponge benthic ecosystems (Figure 1).

DFO created these 59 OECMs using authorities granted under the Fisheries Act to prohibit fishing activities that pose a risk to biodiversity. OECMs created by fisheries restrictions are termed “marine refuges” by DFO in order to distinguish them from other types of OECMs the department may have in the future (for example, historical shipwreck sites). In order to maintain continuity with past science advice and to mirror language used in the scientific literature, the coral and sponge conservation areas will be referred to as “OECMs” in this document.

Canada’s coral and/or sponge OECMs prohibit bottom-contact fishing activities in order to protect these often fragile, sometimes slow-growing species. An OECM provides biodiversity conservation benefits (BCBs), which are benefits for a habitat, species or other component of the ecosystem resulting from the implementation of the management measure. The goal is to create a net positive change in, or prevent the loss of, biodiversity within the OECM (DFO, 2016). BCBs include the focus of the conservation area, known as a direct BCB, and the “co-benefits” which can occur incidentally as a result of the area, known as an indirect BCB (DFO, 2016). For coral and sponge OECMs, the direct BCBs relate to the health of coral and sponge species and their habitats. Indirect BCBs for corals and sponges vary by location and type of coral and/or sponge habitat and will be explored further in this CSAS process.

DFO is responsible for the establishment and management of OECMs in Canada, and monitoring is essential to determine if these closures are effective at meeting their conservation objectives. The conservation objectives of these OECMs is, in general to protect cold-water corals and/or sponges. In order to develop monitoring plans for these areas, managers will need to identify appropriate ecological monitoring indicators. The proper selection of these is essential to properly assess the effectiveness of an OECM. Once indicators are selected, monitoring plans can be developed, including the tools and techniques needed to collect appropriate data, and data assessment methodologies.

1.3. MEETING OBJECTIVES

The Marine Planning and Conservation and Fisheries Resource Management programs requested national guidance on how to monitor coral and/or sponge OECMs to demonstrate that they achieve direct and indirect BCBs. More specifically, the objectives are to:

1. Characterize the corals and/or sponges in Canadian OECMs (for example, by group or habitat type), and detail the available baseline information and knowledge gaps;
2. Provide a review of the known and expected indirect BCBs of coral and/or sponge habitats, and where possible, link these to the groups of corals and/or sponges described in Objective 1;
3. Identify appropriate ecological indicators to monitor coral and/or sponge areas for direct and indirect BCBs along with the strengths and limitations of each indicator; and
4. Identify potential tools, techniques and/or methodologies for monitoring the direct and indirect BCBs of coral and/or sponge areas and provide advice on their strengths and limitations.

This work will support the development of a robust and consistent approach to developing ecological monitoring programs for Canada's OECMs[1].

2. CORAL AND SPONGE OECMS ACROSS CANADA

2.1. GROUPING CORALS AND SPONGES

Monitoring programs focus on the conservation objectives of an area to make sure they are being met. In the case of coral and sponge OECMs, the conservation objectives are to protect corals and sponges, with slight variations depending on region (see Table 1). Since coral and sponge OECMs are often inhabited by multiple, the creation of groups can facilitate monitoring. For instance, the term "functional groups" has been previously used in this context, defined by collections of organisms that share similar characteristics (physiological, behavioral, trophic level) or perform a similar ecological function, regardless of taxonomy (Bundy et al. 2017; Faille et al. 2019). Here we suggest the use of a classification based on readily identified groups of corals and sponges to help in indicator selection and monitoring development. In this context, groupings for monitoring purposes are based on previously described assemblages or taxonomic groups, which can be particularly helpful when data on individual species is limited.

The level of taxonomic identification of coral and sponge species is limited by the types of data available. Most corals and sponges found in Canadian waters are located in relatively deep waters (e.g., >100 m) which limits the availability of data on their distribution and biology for many species. Scientific trawl surveys and commercial fishing have been the main source of

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[1] The meeting took place in December of 2020, and some references which were in press at the time of the meeting were published after 2020. The document still reflects the state of knowledge in 2020.

data used by DFO to map their distribution in Canada (Gullage et al., 2017; Kenchington et al., 2016; Wareham and Edinger, 2007), although imagery data have also contributed to this knowledge (e.g., Baker et al., 2012; Beazley et al., 2019; Dinn, 2020; Du Preez et al., 2015). While these data have been useful to identify areas where corals and sponges are found, trawl data can be limited in the type of information provided. Firstly, fisheries observers on commercial trawlers and even some DFO personnel aboard scientific trawl surveys may not be fully trained to identify corals and sponges at lower taxonomic levels. For example, sea pens are known to exist in the northern Gulf of St. Lawrence (nGSL), but the northern shrimp fishery survey with 100% observer coverage reported no sea pen catches in the trawls (Colpron et al., 2010). Even in cases where personnel are trained, species-level identifications can be difficult or impossible in some cases. For example, inter-habitat morphological variation can occur in sponges (Hooper, 2003), and in most cases accurate species identification requires the use of microscopy tools and/or molecular methods. In addition, due to their fragility, coral and sponge specimens obtained through trawling are often collected as fragments that can be impossible to identify to species level. For instance, gorgonian corals can be quite brittle, and sponges are rarely brought up in a trawl intact (Jørgensen et al., 2016; Rideout, et al. 2024). Catchability of corals and sponges is also a caveat of the survey process. For example, sea pen catchability in the nGSL survey varies from one species to another and is biased towards larger species. This can lead to underestimations of abundance (Sainte-Marie, B., DFO, unpublished data; Kenchington et al. 2011). The bias between sea pen species is explained, among other things, by the ability to burrow in the sediments of certain small sea pens such as *P. aculeata* (Langton et al. 1990; Chimienti et al., 2018a). As a consequence, our knowledge of coral and sponge diversity based on trawl samples is limited by these facts.

For the purpose of this report, coral groups were defined based on phylogeny, morphology (e.g., body size, shape), life history traits, and habitat preferences, following the convention described in Gullage et al. (2022) (Table 2). Corals were classified into eight groups: large and small gorgonians, soft corals, sea pens, black corals, reef building corals, cup corals, and hydrocorals (Figure 2). First, corals in the class Hydrozoa (hydrocorals) were separated from those in the class Anthozoa, which holds a much higher diversity. Then monophyletic taxa within Anthozoa were grouped on their own: black corals (order Antipatharia) (Barrett et al., 2020), sea pens (order Pennatulacea) (McFadden et al., 2006), and stony corals (order Scleractinia) (Kitahara et al., 2010; Lin et al., 2016). Stony corals were further divided based on functional traits (e.g., colonial vs. solitary: reef-forming vs. cup corals) as these corals can form quite distinct communities on their own. The remaining corals in order Alcyonacea (non-monophyletic) were grouped based on general morphology and size (e.g., fan-shaped) into gorgonians (large and small) and soft corals. Descriptions of the known groups contained in each OECM are presented in Table 1, and these groupings and examples are presented in Table 2.

Sponges, on the other hand, are more difficult to separate into discrete groupings. Although many sponges perform similar biological functions across phylogenetic groupings, external morphology may vary, making monitoring through visual means difficult. With the exception of carnivorous species, sponges filter water to obtain food and often form important three-dimensional habitat. Sponges are also generally asymmetrical as adults, so growth is not always predictable even within a species. Across sponge classes there are varied morphologies which grow and take on forms which take advantage of environmental conditions, to the point where similar morphologies may arise in distantly related groups (Hooper and Van Soest, 2002; Dinn et al., 2020b). Sponge larval settlement substrate is also variable, ranging from sand to bedrock, and even marine debris (Dinn et al., 2020a; Santín et al., 2020). Sponges are therefore particularly difficult to identify from imagery surveys as morphologies are not always consistent and the same species can be found growing on a number of substrate types (Beazley and

Kenchington, 2015). For these reasons, sponges are more challenging to group either based on phylogenetic relationships or morphology.

Maldonado et al. (2017) provided a review of the major known sponge aggregations that can act as key marine habitats. Considering the challenges described above in grouping sponges, and the fact that several of the sponge aggregations described in Maldonado et al. are present in Canadian waters, we opted for selecting four of these groups for the purposes of this document: glass sponge reefs, astrophorid grounds, *Vazella pourtalesii* grounds, and a group composed of mixed sponge types described below (Figure 3). Glass sponge reefs are present on the west coast of Canada (Krautter et al., 2001), deep-sea astrophorid grounds are known to occur in Newfoundland waters and the eastern Canadian Arctic (Dinn et al., 2019; Fuller, 2011; Hestetun et al., 2017; Murillo et al., 2018b; Murillo et al., 2012; Verhoeven & Dufour, 2018) and monospecific hexactinellid grounds comprised of *Vazella pourtalesii* occur uniquely on the Scotian Shelf (Beazley et al., 2018). Additionally, we have defined a mixed-sponge group that includes sponges that have been systematically recorded during the trawl or camera surveys, but which are not unique or consistently formed by similar species, as are the other groups. Due to emerging new sponge records across Canada, it may be possible to expand and introduce additional groupings when more data become available and different monitoring plans for particular sponge aggregations are required. For the purposes of monitoring, the focus should be on the best methodologies to monitor a particular group of sponges, rather than the individual species of the area (except for monospecific grounds). Therefore, sponge groupings were based on common aggregations rather than on phylogenetic groups to more conveniently develop a monitoring plan.

The use of coral and sponge groups can also aid in our understanding of coral and sponge indirect biodiversity conservation benefits (BCBs). An ecological monitoring program involves monitoring not only the corals and sponges (direct BCBs), but also monitoring the co-benefits, or indirect BCBs, of the closures. As research on cold-water corals and sponges is challenging due to their often fragile and deep-dwelling nature, research on their ecological roles is an ongoing task. The use of groups can help researchers to develop monitoring plans that examine both direct and indirect BCBs, especially when there is limited species level information.

2.2. PACIFIC

The Strait of Georgia and Howe Sound Glass Sponge Reef (32.6 km^2, contributing <0.01% to MCT, Figure 4) OECMs cover 17 fisheries area closures that protect and conserve glass sponge reefs (Table 1) - a globally unique ecosystem that provides habitat for many species including those of economic importance (Dunham et al., 2018a). The glass sponge reefs also play a role in filtration, nutrient cycling, sequestering carbon and silica and recycling nitrogen thereby linking benthic and pelagic environments (Chu et al., 2011; Tréguer and De La Rocha, 2013; Kahn et al., 2015; DFO, 2018). In the Strait of Georgia and Howe Sound there are 17 glass sponge reef OECMs in which commercial, recreational or Indigenous Food, Social and Ceremonial bottom-contact fishing activities are prohibited. These OECMs contribute <0.01% to MCT (exclusive of overlap with existing protected areas in Howe Sound). Commonly found species include the cloud sponges *Aphrocallistes vastus* and *Heterochone calyx*.

Natural Resources Canada has used remote sensing data (i.e., multibeam, swath bathymetry and backscatter), collected by the Geological Survey of Canada and the Canadian Hydrographic Service, to identify locations of geological glass sponge reefs and to create a geological aerial footprint for each reef (Conway et al., 2007; Conway et al., 2005). Video and still imagery data from remotely operated vehicles (ROVs) and passive acoustic data from hydrophones have been collected during several DFO Science surveys (e.g., DFO 2018, DFO 2020a) to assess the condition of the reefs and their biological communities, and to support several scientific studies. These studies include analyses on ecological roles and functions of glass sponge reefs, reef soundscape, invertebrate settlement, and food web typology (Archer et al., 2018; Archer et al., 2020a; Archer et al., 2020b; Dunham et al., 2018a). Additional biological data on the Howe Sound reefs has been collected by the Marine Life Sanctuaries Society and other community partners using drop camera and SCUBA (Clayton and Dennison, 2017; McAuley, 2017).

Baseline information and knowledge gaps

Datasets resulting from the above-mentioned studies have been used in DFO assessments of the Strait of Georgia and Howe Sound glass sponge reefs (DFO, 2020a). The video and still imagery data collected by DFO represent only a very small portion of the known reef habitat (i.e., 0.2-0.8% in Strait of Georgia and Howe Sound (Dunham et al., 2018a); and 0.4-4.4% (DFO, 2018) and 0.8-5.1% (DFO, 2020a) in Howe Sound only) and there is still insufficient data to form a comprehensive ‘baseline’ (Dunham et al., 2018a). The low spatial coverage of imagery data across sponge reefs limits our ability to detect change and presents a challenge for monitoring, especially given the natural variability of sponge reefs and their associated communities.

The Offshore Pacific Seamounts and Vents Closure (82,530 km^2, contributing 1.44% to MCT, Figure 4) was initiated in 2017. This area is an Area Of Interest (AOI) slated to be established as a Marine Protected Area (MPA)[2]. The Fisheries Act closure includes 35 hydrothermal vent fields and 36 seamounts (both designated by the Canadian Government as Ecologically and Biologically Significant Areas and Vulnerable Marine Ecosystems; Ban et al. 2016; DFO 2019). The closure prohibits all bottom-contact commercial and recreational fishing activities to protect and conserve cold-water coral and sponge populations (taxa listed as ecological components of interest for the OECM; Du Preez and Norgard, 2022). While inactive vent fields host novel assemblages of cold-water corals and sponges, the depths of these geomorphic features are presently beyond the reach of bottom-contact fishing gear (vents between 1,850 to 3,000 m depth; Ban et al. 2016). Seamounts support regionally rare assemblages of cold-water corals and sponges but at depths accessible to fishing (some seamounts rise above 300 m depth; DFO 2019; Du Preez and Norgard, 2022). Prior to the closure, multiple shallow seamounts above ~1,500 m depth were fished for sablefish (*Anoplopoma fimbria*) and other groundfish using bottom-contact long-line trap and hook gear.

The Fisheries and Oceans Canada Deep Sea Ecology (DSE) program was established in 2017 to provide science advice related to—among other things—the management and monitoring of the offshore Pacific seamounts. The DSE program has used meta-analyses of bathymetric models to identify the location of seamounts and used ship-based and visual surveys to ground-truth these predictions, as well as to document the presence, condition, and ecology of cold-water corals and sponges as well as other ecological components of interest. The ongoing research aims to support the management and monitoring of the offshore Pacific seamounts. Studies completed to date include analyses on an ecological overview (DFO 2019), climate change and its impacts on cold-water corals and sponges (Ross et al. 2020), and anomalous events of productivity redistribution (Archer et al. 2018), and ecological categorization of seamounts and seamount areas (Du Preez and Norgard, 2022).

Baseline information and knowledge gaps

The following is a summary of the information provided in Du Preez and Norgard (2022). Most data on cold-water corals and sponges within the Offshore Pacific Seamounts and Vents

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[2] This area became the Tang.ɢwan – ḥačxwiqak – Tsig̱is Marine Protected Area in 2024.

Closure was collected between 2017 and 2019. During three offshore seamount expeditions, eight of the 36 seamounts were visually surveyed using a towed-camera or remotely operated vehicle. These surveys are invaluable snapshots of a discrete area, but of a relatively limited footprint compared to the size of a seamount. For example, Union is one of the most well-explored seamounts in the OCEM, with five benthic visual transects completed in 2017. The dives to 2,100 m depth (equipment limit) covered roughly 23.3 km or 0.09 km^2 (based on 4-m wide camera field of view), but Union Seamount starts at 3,239 m depth and covers 680 km^2, so while the survey successfully captured the top two-thirds of its height, it only covered 0.013% of its area.

These visual surveys also suffer a significant depth bias since the initial surveys targeted “shallow” seamounts impacted by fishing. Of the 36 OECM seamounts, 31 summit below one km depth but only three of these have been surveyed. Furthermore, there is little to no information on natural spatio-temporal variability.

Additional information is derived from specimen collections, DNA and eDNA samples, oceanographic profiles and samples, and ship-based mapping collected during the same expeditions. There is a general lack of high-resolution multibeam bathymetry mapping within the OECM, with only one seamount of the 36 mapped in its entirety (Dellwood Seamount). Some time-series data related to stressors and/or environmental conditions are available from the DFO long-term oceanographic line P program (>50 years of oceanography to depths of 2,500 m, including some locations over or adjacent to seamounts) and the new DFO glider program, underwater autonomous oceanographic mooring and hydrophones, and Ocean Networks Canada NEPTUNE Observatory nodes.

2.3. ARCTIC

As of July 2020, there are three OECMs established within the Canadian Arctic where the protection of corals and sponges is being specified within their conservation objectives and where bottom trawling is prohibited (Table 1). These OECMs are all located in the Eastern Canadian Arctic, and designated as: Hatton Basin Conservation Area, Davis Strait Conservation Area, and Disko Fan Conservation Area. Coral and sponge concentrations in the Eastern Canadian Arctic have been delineated based on a modelling process using scientific trawl surveys and historical commercial bottom trawl catch information (DFO, 2017a; Kenchington et al., 2010; Kenchington et al., 2016). The Hatton Basin Conservation Area is co-managed by DFO-Ontario & Prairie and DFO-NL regions.

Hatton Basin Conservation Area (42,459 km^2, contributing ~0.74% to MCT, Figure 5) is located on the border of Newfoundland-Labrador Shelves and Eastern Arctic Bioregions. It overlaps with the Hatton Basin/Labrador Sea/Davis Strait Ecologically and Biologically Significant Area (EBSA) and the Outer Shelf Saglek Bank EBSA. It contains significant concentrations of small gorgonians including *Acanella arbuscula*, *Radicipes gracilis*, *Anthothela grandiflora*, large gorgonians such as *Acanthogorgia armata*, *Paragorgia arborea*, *Primnoa resedaeformis*, and *Paramuricea* spp., as well as sponges (*Geodia barretti* and *Geodia phlegraei*) and non-aggregating species such as black coral (*Stauropathes arctica* and *Bathypates* spp.), cup corals (mostly *Flabellum alabastrum*) and hydrocorals (Wareham, 2009; Wareham et al., 2010; Knudby et al., 2013). This conservation area has particularly important concentrations of large gorgonians (e.g., *P. resedaeformis*), soft corals (e.g., *Duva florida*), and sponges as evidenced from both trawl bycatch samples and seafloor imagery (B. Neves, personal communication). Twelve sponge species morphotypes were quantified in the region based on ROV imagery and collections (Dinn et al., 2020a), and considering limitations in terms of taxonomic resolution of trawl samples (often identified as Porifera sp.), additional sponge species are expected to exist in the area.

Davis Strait Conservation Area (17,298 km^2, contributing ~0.30% to MCT, Figure 5) is located within the Eastern Arctic Bioregion, and within the Hatton Basin/Labrador Sea/Davis Strait EBSA. Species of regional importance in Davis Strait Conservation Area include small and large gorgonians such as *Acanella arbuscula*, *Radicipes gracilis*, *Paragorgia arborea, Primnoa resedaeformis,* and *Paramuricea* spp., although based on trawl data records of *R. gracilis* and large gorgonians are rare (unpublished data V. Wareham Hayes and B. M. Neves), black corals (V. Wareham Hayes and B. M. Neves, unpublished), sea pens (*Umbellula lindahli, Anthoptilum grandiflorum, Balticina finmarchica, Ptilella grandis = Pennatula grandis*) and sponges (*Geodia barretti and Geodia phlegraei*) (Kenchington et al., 2010; Kenchington et al., 2016; Knudby et al., 2013; Neves et al., 2015b; Wareham, 2009).

Disko Fan Conservation Area (7,485 km^2, contributing ~0.13% to MCT, Figure 5) is located within the Eastern Arctic Bioregion. It was identified as an EBSA in 2011, partly due to the presence of significant concentrations of large gorgonian corals (DFO, 2011; Hiltz et al., 2018) including large patches of globally unique, high-density bamboo corals – *Keratoisis* sp. (DFO, 2007; Neves et al., 2014; Wareham, 2009). At this location, *Keratoisis* sp. forms dense patches on soft bottoms, and can be ~55 m long and ~1 m tall at depths >900 m (Neves et al., 2014). Molecular analyses of two *Keratoisis* sp. samples from this location revealed it to be the 'kerD2d' haplotype (Neves et al., 2014), different from *Keratoisis ornata*, which had been previously reported by DFO at this location (DFO, 2007). This species has later been identified as *Keratoisis flexibilus* (Heestand Saucier, 2016). Other examples of large gorgonian corals found in this area include bubblegum corals (*Paragorgia arborea*) and *Primnoa resedaeformis* (bycatch data; Neves et al. 2014). The bamboo coral *Acanella arbuscula* and sea pens (e.g., *Anthoptilum* sp., *Ptilella grandis*, and *Umbellula encrinus*) can also be found within this conservation area (B. Neves, personal communication, Neves et al., 2018), and seven sponge species were identified among *Keratoisis* coral patches therein, which likely represents an underestimation of sponge diversity in the area (Dinn et al., 2020). Finally, black corals have also been caught as fishing bycatch north of this conservation area, just outside of its boundaries (V. Wareham Hayes and B. Neves, unpublished).

Baseline information and knowledge gaps

Most of the published information on coral and sponge taxonomic diversity and abundance in the Canadian Arctic has been obtained from bycatch data from annual multispecies trawl surveys and from the Northern Shrimp Survey Foundation (NSRF) surveys being conducted in this region. In addition, scientific trawl surveys and historical commercial catch information have been used in modelling processes to delineate coral and sponge concentrations within the Arctic OECMs (DFO, 2017a; Kenchington et al. 2010; Kenchington et al. 2016). In general, information available from the Eastern Arctic region including the established OECMs is still limited, and numerous knowledge gaps exist on species distributions, their role within the Arctic marine ecosystem on multiple regional and temporal scales, and impact of known and predicted stressors on their biodiversity and abundance.

2.4. GULF OF ST. LAWRENCE BIOREGION (QUÉBEC AND GULF REGIONS)

The Gulf of St. Lawrence bioregion (GSL) contains 11 coral and/or sponge OECMs (Table 1) that were implemented in December 2017 based on identified significant benthic areas (SBAs; DFO, 2017a; Kenchington et al., 2010; Kenchington et al., 2016). Of note, the areas for sponges were mostly established based on Porifera spp. biomass captures, as sponge species identifications were not routinely performed from trawl surveys until recently (Bourdages et al. 2020; Dinn 2020, Nozères et al. 2020). From collections in the bioregion, ten coral species (four sea pens, three soft corals (previously family Nephtheidae), *Clavularia* cf. *modesta*., *Heteropolypus* cf. *sol*., and the cup coral *Flabellum alabastrum*) and 46 sponge taxa (Dinn

2020) have been identified. Additional sponge taxa are being identified and additional species records are expected to be discovered. The use of bottom-contact gear such as bottom trawls, dredges, gillnets, bottom longlines, bottom seines and traps, for commercial and recreational purposes or for food, social and indigenous rituals, is prohibited in these OECMs. Five of these areas have specific conservation objectives for cold-water sponges, five aim to protect sea pen fields, and the remaining OECM aims to protect both coral and sponge habitat. Species that link to the conservation objectives of particular OECMs are described below. Several demersal species at risk such as Thorny skate, Smooth skate, Atlantic wolffish, and Spotted wolffish are also found in some of the OECMs.

Beaugé Bank Sponge Conservation Area (215 km^2 contributing <0.01% to MCT, Figure 6) is a shallow bank (<100 m) located in the northeastern Gulf of St. Lawrence with the objective to protect cold-water sponges. This conservation area contains a high concentration of sponges, including the large structure-providing *Hemigellius arcofer* and *Mycale lingua* (Kenchington et al., 2016; Nozères et al., 2020). The area also includes other biologically significant features, such as a high concentration of soft corals.

Jacques-Cartier Strait Sponge Conservation Area (246 km^2, contributing 0.01% to MCT, Figure 6) is located in the northern Gulf of St. Lawrence, just north of Anticosti Island. The conservation objective of this closure is to protect cold-water sponges. This conservation area features some of the highest sponge biomass (kg/tow) bycatch of all coral and sponge conservation areas established in this bioregion (Nozères et al. 2020). The area also includes other important biological features, such as a high concentration of *Gersemia rubiformis* soft corals, and the presence of the large structure-providing *Hemigellius arcofer* sponge.

East of Anticosti Island Sponge Conservation Area (939 km^2, contributing 0.02% to MCT, Figure 6) is located in the eastern Gulf of St. Lawrence east of Anticosti Island. The objective of this closure is cold-water sponge protection. The area also includes other biologically important features, such as a high concentration of *Duva florida* and *Gersemia rubiformis* soft corals and the presence of the large structure-providing *Hemigellius arcofer* and *Mycale lingua* sponges (Nozères et al. 2020; Dinn 2020).

South-East of Anticosti Island Sponge Conservation Area (845 km^2 contributing 0.01% to MCT, Figure 6) is located in the Laurentian Channel, a deeper refuge (>200 m) southeast of Anticosti Island. The objective of this area is cold-water sponge protection. Five species of sponge have been identified in this area, including the structure forming *Asconema foliatum* (Nozères et al. 2020; Dinn 2020). This area also includes other biologically important features, such as high concentrations of soft corals and *Anthoptilum grandiflorum* sea pens (Bourdages et al. 2020).

Parent Bank Sponge Conservation Area (530 km^2, contributing <0.01% to MCT, Figure 6) is located in the northwest Gulf of St. Lawrence. The conservation objective of this closure is to protect cold-water sponges. The area also includes other important biological features, such as a high concentration of *Pennatula aculeata* sea pens and soft corals (Bourdages et al. 2020). At least six sponge species have been identified in the area, such as large structure-providing *Hemigellius arcofer* and *Mycale lingua* (Nozères et al. 2020; Dinn 2020).

Eastern Honguedo Strait Coral and Sponge Conservation Area (2,338 km^2, contributing 0.04% to MCT, Figure 6) is located in the western Gulf of St. Lawrence east of the Gaspe peninsula. The objective of this closure is to protect cold-water corals and sponges. This conservation area is the only closure that includes high concentrations of all four sea pen species, *Balticina finmarchica, Anthoptilum grandiflorum, Ptilella grandis* and *Pennatula aculeata* (Bourdages et al. 2020), as well as a high sponge species richness with at least 10 taxa identified from trawls (Nozères et al. 2020). The area also includes other important biological features, such as a high concentration of *Duva florida* soft corals.

Central Gulf of St. Lawrence Coral Conservation Area (1,284 km^2 contributing 0.02% to MCT, Figure 6) is located in northwest of Cape Breton Island in the central Gulf of St. Lawrence. The objective of this area is cold-water coral protection. This conservation area features the highest known concentration of the *Anthoptilum grandiflorum* sea pen in the Estuary and Gulf of St. Lawrence bioregion (Bourdages et al. 2020). It is also the only OECM in the Gulf of St Lawrence region that includes *Flabellum alabastrum* cup corals, the only species of hard coral (Scleractinia) in the Gulf of St. Lawrence, which have a limited range in the Gulf's channels. This conservation area also includes other biologically important features, such as a high concentration of *Duva florida* soft corals and the presence of the large structure-providing *Asconema foliatum* sponge (Nozères et al. 2020; Dinn 2020).

Western Honguedo Strait Coral Conservation Area (496 km^2, contributes <0.01% to MCT, Figure 6) is located in the northwest Gulf of St. Lawrence. The conservation objective of this closure is to protect cold-water corals. This conservation area features high concentrations of three sea pen species*: Pennatula aculeata, Ptilella grandis* and *Anthoptilum grandiflorum* (Bourdages et al. 2020).

North of Bennett Bank Coral Conservation Area (821 km^2, contributing 0.01% to MCT, Figure 6) is located in the central gulf of St. Lawrence, south of Anticosti Island. The conservation objective of this closure is to protect cold-water corals. This conservation area features a high concentration of the sea pen *Anthoptilum grandiflorum* (Bourdages et al. 2020).

Slope of Magdalen Shallows Coral Conservation Area (335 km^2 contributing <0.01% to MCT, Figure 6) is located in the Laurentian Channel, northwest of Cape Breton Island. The objective of this area is cold-water coral protection. High concentrations of *Ptilella grandis* and *Anthoptilum grandiflorum* sea pens are present in this area (Bourdages et al. 2020).

The Eastern Gulf of St. Lawrence Coral Conservation Area (423 km^2 contributing <0.01% to MCT, Figure 6) is located in the Laurentian Channel, north of Cape Breton Island, NS and west of Port aux Basques, NL. The objective of this area is cold-water coral protection. Concentrations of *Ptilella grandis* sea pens within the Estuary and Gulf of St. Lawrence Bioregion are highest in this area. A high concentration *of Anthoptilum grandiflorum* sea pens is also present in this area (Bourdages et al. 2020).

Baseline information and knowledge gaps

Presently, the primary means for the collection of biodiversity data in these areas is through fisheries independent multispecies trawl surveys (Bourdages et al., 2020; Nozères, et al. 2015). These surveys have been occurring in the Southern Gulf of St. Lawrence (sGSL) since 1971 and the Estuary and Northern Gulf of St. Lawrence (nGSL) since 1984. Recently since 2003 (sGSL) and 2006 (nGSL, greatly improved in 2011) the data on benthic invertebrates has improved through identifications at lower taxonomic levels (Dinn 2020, Bourdages et al., 2020; Nozères, et al. 2015). Based on collected trawl data, kernel density analysis was used to identify likely concentrations of corals and sponges, and models of probability of occurrence using random forest machine learning were used to predict biomass of several invertebrate groups in the region (Kenchington et al. 2016, Murillo et al. 2016). Efforts to identify collected sponge specimens to lower taxonomic levels have occurred in recent years (Dinn, 2020; Nozères et al., 2020). Currently for the nGSL, annual post-survey reports list general sponge captures and names of some species, however additional species identifications are ongoing (Dinn, 2020; Nozères et al., 2020).

Baseline information was also gathered using different benthic imagery tools in three different OECMs of the Gulf of St. Lawrence. A survey was conducted in 2017 using the ROV ROPOS (Remotely Operated Vehicle for Oceans Sciences) in the Eastern Honguedo Strait Coral and Sponge Conservation Area and the Eastern Gulf of St. Lawrence Coral Conservation Area (Faille et al. 2019). This survey was exploratory in nature where video transects one km in length were accomplished at five locations. During the ROV survey, samples, especially sponges, were collected for identification. The Jacques-Cartier Strait Sponge Conservation Area was surveyed in the summer of 2019 over a systematic grid with a towed camera system (benthic sled, 20 × 300 m transects) and a drop camera (53 stations), and these data are currently being assessed.

Data gaps exist for each of the 11 OECMs in the GSL bioregion, especially concerning precise bathymetric information and bottom types present in each closure. General benthic habitat data is available on a 10 × 10 km grid (Dutil et al. 2011, with CD-ROM) that includes information on sediment types in the region (Loring and Nota 1973). Higher resolution data obtained with surveys using multibeam echosounders (MBES) and/or sidescan sonar (SS) would be needed to assist with seafloor mapping and aid in habitat classification.

Although trawl data is the main source of information available, trawl surveys are limited in OECMs within the bioregion. On average, one trawl set takes place each year in a given OECM, which represents a mean percentage of 0.06 % of swept area in total in the past 15 years (DFO 2020b). Furthermore, there remain some areas in sponge OECMs where trawl sets have not been successful or where trawls have not been attempted because of the roughness of bottom, mostly in the shallower areas (40-100 m). Especially in those areas, alternative sampling methods such as non-destructive underwater imagery will be needed to complement surveys in each of the GSL OECMs. Species identifications of sponges have only occurred in recent years and were based on limited trawl survey collections (Dinn, 2020; Nozères et al., 2020), therefore sponge data in catch databases is sparse. More effort is needed to fill those gaps.

2.5. MARITIMES

The Scotian Shelf Bioregion contains five coral and/or sponge OECMs (Table 1). In total they represent 16,580 km^2, contributing 0.17% to MCT. All commercial bottom contact fishing gear is prohibited in these areas and no human activities that are incompatible with the conservation of the ecological components of interest may occur or be foreseeable within each area.

Corsair and Georges Canyons Conservation Area (8,797 km^2, contributing ~0.15% to MCT, Figure 7) is located south of Georges Bank, near the Canadian-United States border. These canyons are part of a chain found along the continental slope of eastern North America. It was established in 2016 to protect the dense aggregations of the large gorgonian corals *Paragorgia arborea* and *Primnoa resedaeformis*, discovered in 2014 (Metaxas et al., 2019). In Corsair Canyon, *P. arborea* was very abundant at depths 484-856 m with some colonies larger than 2 m tall. Locally, these are the densest aggregations of *P. arborea* detected on the continental slope off Nova Scotia (Metaxas et al. 2019).

Jordan Basin Conservation Area (49 km^2, contributing <0.01% to MCT, Figure 7) captures two prominent bedrock ridges, including an outcrop called the “Rock Garden”, in the eastern portion of Jordan Basin, 100 km west of Nova Scotia. It was established in 2016 to protect the high densities of the large gorgonian coral *Primnoa resedaeformis* and other sensitive filter feeding invertebrate communities.

Lophelia Coral Conservation Area (15 km^2, contributing <0.01% to MCT, Figure 7) is located at the Stone Fence, southeast of Cape Breton, Nova Scotia. It was put in place in 2004 and closed a small area surrounding the entire reef building coral *Lophelia pertusa* to all bottom contact fisheries (Breeze and Fenton, 2007). This area was subject to intense bottom fishing between 1980 and 2000, and the corals and nearby seabed showed signs of extensive damage from fishing gear (Buhl-Mortensen et al., 2017a). In addition to *L. pertusa*, other corals taxa, including soft corals, large gorgonians and cup corals have been observed in the area (Buhl-Mortensen et al., 2017a). The objective of the closures is to protect the reef complex from further damage and allow for recovery. It represents the only known living cold-water coral reef in Canada. Through the analysis of benthic images collected in 2003, 2009, and 2015, Beazley et al. (2021) evaluated the effectiveness of the conservation area in terms of its success in facilitating the recolonization and recovery of its target species, *L. pertusa*, and in conserving local benthic biodiversity. They observed an increase in epibenthic megafaunal species density and abundance over time that was higher inside the closure than outside, suggesting that the conservation area has facilitated the recruitment and recovery of the benthic communities within its confines. Additionally, they discovered numerous undisturbed large mounds of live *L. pertusa* that establish a local recruitment source.

Northeast Channel Coral Conservation Area (391 km^2, contributing <0.01% to MCT, Figure 7), off southwestern Nova Scotia, was the first coral conservation closure in the Maritimes (Breeze and Fenton, 2007). It was established in 2002 with the objective of protecting high densities of large gorgonian corals (*Paragorgia arborea* and *Primnoa resedaeformis*). This conservation area also includes other coral groups such as soft corals, small gorgonians, sea pens, and cup corals (Cogswell et al. 2009).

Emerald Basin and Sambro Sponge Conservation Areas (260 km^2, contributing <0.01% to MCT, Figure 7) were designated in 2013 to protect the significant concentrations of the glass sponge *Vazella pourtalesii*, commonly known as “Russian Hat”, on the Scotian Shelf. It includes two closures: The Emerald Basin *Vazella* Conservation Area (197 km^2), and the Sambro Bank *Vazella* Conservation Area (62 km^2), which added a second significant concentration of *Vazella* in order to provide replication to aid population recovery (Beazley et al., 2018).

Baseline information and knowledge gaps

The primary means for data collection in these areas is through scientific surveys with underwater camera systems (e.g., ROPOS, Campod, etc.), complemented by data from the multispecies trawl surveys. Cogswell et al. (2009) provided an overview of coral knowledge distribution in the Maritimes Province. Posterior studies include Kenchington et al. (2016), Beazley et al. (2017); Beazley et al., 2021), Buhl-Mortensen et al. (2017a) Metaxas et al. (2019). Additionally, species distribution models (SDMs) have been applied to different coral and sponge taxa/groups to map their distribution in the Scotian Shelf (e.g., Bryan and Metaxas 2007; Beazley et al. 2016; Beazley et al. 2018). There is good knowledge of coral composition and distribution in the Scotian Shelf and upper Slope, and 39 coral taxa have been identified. However, the information from deeper areas (> 1,500 m) is still limited. Sponge composition research in this region is still underway. Most of the sponge knowledge comes from shallow areas and old surveys, although sponges collected from the 2017 summer RV surveys have been identified adding at least 14 taxa to the previous knowledge (Murillo unpublished data). A total of 79 sponge taxa have been identified from these surveys. However, RV surveys do not sample hard bottom, and information from these areas is limited. Hawkes et al. (2019) identified 32 sponge taxa from the *Vazella* sponge grounds, most being encrusting sponges that cannot be identified from images, whereas Beazley et al. (2018) identified 5 large -sized sponge morphospecies from the Eastern Scotian Slope from a benthic imagery survey. Sponge identification from *in situ* images is very challenging and most of these records remain at a low-resolution.

2.6. NEWFOUNDLAND & LABRADOR

The Newfoundland and Labrador region has three OECMs focused on corals and sponges (Table 1).

Division 3O Coral and Sponge Conservation Closure (10,422 km^2 within Canada's Exclusive Economic Zone (EEZ), contributing 0.18% to MCT**,** Figure 8) extends to outside of the Canadian EEZ, which is managed by the Northwest Atlantic Fisheries Organization (NAFO). The conservation objective of this OECM is to protect corals and sponges. It overlaps a significant portion of the Southwest Shelf Edge and Slope EBSA. The closure includes areas of sea pens and large and small gorgonians. All bottom fishing activities are prohibited in this OECM. Some portions of the closure are expected to be highly suitable for large and small gorgonians, and sea pens (Gullage et al., 2017). It partially overlaps several significant benthic areas (SiBAs): sea pens (1); large gorgonians (2, and slightly overlaps a 3rd one); and small gorgonians (3).

Hopedale Saddle Closure (15,411 km^2, contributing 0.27% to MCT, Figure 8) is conserved to protect corals and sponges and contribute to the long-term conservation of biodiversity. This area overlaps three EBSAs: Outer Shelf Nain Bank, Labrador Slope, and Hopedale Saddle, and it overlaps 46% of the Outer Shelf Nain Bank EBSA. All bottom-contact fishing activities are prohibited in this OECM. Some portions of the closure are expected to be highly suitable for large and small gorgonians, and even sea pens (Gullage et al., 2017). High biomass of the large gorgonian *Paragorgia arborea* has been reported inside of this closure. It partially overlaps several SiBAs: large gorgonians (1); small gorgonians (1); and sponges (4,1 entirely inside OECM).

Northeast Newfoundland Slope Closure (55.353 km^2, contributing 0.96% to MCT, Figure 8) is the largest Newfoundland OECM, and its conservation objective is to protect corals and sponges and contribute to the long-term conservation of biodiversity. This area also overlaps with 32% of the Orphan Spur, an EBSA that supports high diversity, including several depleted species (e.g., Roundnose Grenadier). All bottom-contact fishing activities are prohibited in this OECM. Some portions of this closure (shallow areas) are expected to be highly suitable for large and small gorgonians, and sea pens (Gullage et al., 2017). High biomass of the large gorgonian *Paragorgia arborea* has also been reported inside of this closure. It partially overlaps several SiBAs: sea pens (large SiBA), large gorgonians (3, and slightly overlaps a 4th one), small gorgonians (1), and sponges (1).

Baseline information and knowledge gaps

Similarly to other regions in Canada, most of the information on coral and sponge distribution and ecology in the NL Region has been obtained through DFO RV surveys (Wareham & Edinger, 2007). Additional targeted benthic surveys in the region have been conducted through imagery (e.g., ROVs) over the past decade (e.g., Baker et al., 2012; Devine et al., 2020; Meredyk et al., 2020). Coral composition and distribution are relatively well known in the Newfoundland and Labrador shelves, where over 70 species have been reported (Gullage et al. 2022, V. Wareham Hayes, personal communication). An overview of coral and sponge distribution in this region is provided in Gullage et al. (2022). Because most data have been collected during RV surveys, which are limited to 1500 m, knowledge on corals and sponges in deeper areas is more limited. Baker et al. (2012) reported coral diversity from ROPOS dives reaching depths of >2,200 m and reported that assemblages differed significantly with depth class and bottom type. Miles (2018) examined ROPOS imagery from the Flemish Cap, where at least 27 coral species were reported up to depths of 2,900 m.

Coral research in the Newfoundland region has been considerable in the past decade. Collaborations with academia have yielded a number of studies including coral reproduction

(Sun et al. 2009; Sun et al., 2010a; Baillon et al., 2014; Baillon et al., 2015; Hamel et al., 2020), growth rates and longevity (Sherwood et al., 2005; Sherwood and Edinger, 2009; Neves et al., 2015a; Neves et al., 2018b), associated diversity (Baillon et al., 2012; Baillon et al. 2016; Neves et al., 2020), coral diversity and distribution (Wareham and Edinger, 2007; Edinger et al., 2011; Baker et al., 2012), trophic ecology (Hamoutene et al., 2008; Sherwood et al., 2008; Salvo et al., 2018), etc. These data have also been used to input species distribution (Radice et al., 2016; Gullage et al., 2017) and density models for the region (e.g., Kenchington et al., 2016). Despite the range in studies, there are still considerable gaps in our knowledge regarding coral distribution and diversity at coastal and deep-water sites. Furthermore, considering the large number of species in the region, our current knowledge is generalized, and additional specific studies are still required in all aspects listed above.

Obtaining data on sponge composition and distribution has been more challenging. Sponge bycatch samples are often fragmented, which further challenges their already difficult taxonomy. In the NL region, sponges are generally not identified at sea during RV surveys, and only specimens brought back for further studies are identified to lower taxonomic levels. Over 100 sponge taxa are believed to exist in the NL region (compiled in Gullage et al. 2022). NL sponge biomass data have also been used to input density models in the region (e.g., Kenchington et al., 2016). There have been fewer ecological studies on sponges from this region compared to corals (e.g., Hayes et al., 2010).

2.7. CONCLUSION

- Based on conservation objective there are: seven coral, 19 sponge, and 14 coral and sponge-focused OECMs nationally.
- Several coral and sponge OECMs in Canada are globally unique.
- Coral and sponge OECMs may also have additional species of interest present, though they do not fall under the conservation objective, which is to protect the corals and/or sponges.
- Common data gaps in coral/sponge OECMs include difficulties associated with species identification and limited knowledge of species distributions.

3. INDIRECT BIODIVERSITY CONSERVATION BENEFITS

3.1. INTRODUCTION

Corals, sponges, and their assemblages provide many ecosystem functions and services in the marine environment. They contribute to vertical relief and increase the availability of microhabitats in areas that often have little three-dimensional structure (Tissot et al., 2006). Increasing complexity provides feeding opportunities for aggregating species, a hiding place from predators, shelter from high flow regimes, a nursery area for juveniles, fish spawning aggregation sites and attachment substrate for fish egg cases and sedentary invertebrates (Fosså et al., 2002; Reed, 2002; Etnoyer and Morgan, 2003; Etnoyer and Warrenchuk, 2007). In general, coral and sponge habitats in deep water represent biodiversity hotspots for invertebrates (Frederiksen et al., 1992; Freiwald et al. 2004; Mortensen & Buhl-Mortensen, 2005; Reed, 2002), and commonly support a high abundance of fish (Koenig, 2001; Husebø et al., 2002; Krieger and Wing, 2002; Costello et al., 2005; Tissot et al., 2006). Additionally, they represent a key link between benthic and pelagic ecosystems (Griffiths et al., 2017; Leys et al., 2018; Pham et al., 2019), facilitate nutrient cycling (Perea-Blázquez et al., 2012; Kutti et al., 2013; Maldonado et al., 2020), and modify biochemical regimes (Kaufmann and Smith, 1997; Soltwedel and Vopel, 2001). Corals and sponges are therefore important ecosystem engineers as they modify water quality and flow and create important biogenic habitat for countless other benthic organisms. In addition, deep-water corals and sponges are slow-growing and fragile organisms, which are often extremely slow to recover after trawling-induced damage (Althaus et al., 2009; Heifetz et al., 2009; Sherwood and Edinger, 2009). Sponges also have biochemical and genetic properties which make them a valuable source of chemical compounds for the pharmaceutical industry (Bell, 2008; Luna, 2015).

Conservation of corals and sponges ensures not only their biodiversity preservation but also protection of the ecosystem functions and services they provide. Several studies have shown the indirect biodiversity conservation benefits that these organisms generate in Canadian and surrounding waters, as described in the following sections, and a list of examples of indirect BCBs associated with coral and sponge groups found in Canadian OECMs is provided in Table 3.

3.2. BIOGEOCHEMICAL CYCLING

Benthic–pelagic coupling, defined as the exchange of energy, mass, or nutrients between benthic and pelagic habitats, plays a key role in aquatic ecosystems. This transfer of energy is essential for various environmental functions, from nutrient cycling to energy transfer in food webs (Griffiths et al., 2017). Benthic–pelagic coupling can occur through the deposition of non-living organic material to benthic habitats (e.g., Smetacek, 1985; Suess, 1980), through the release of inorganic nutrients from the sediments (e.g., Maldonado et al., 2020), or can be mediated by living organisms through bioresuspension and biodeposition (Graf and Rosenberg, 1997). Processes mediated by living organisms play a key role and include predation by pelagic organisms on benthic fauna, ontogenetic shifts in habitat use, reproductive (life cycle) fluxes, diel and seasonal migrations, nutrient-cycling effects of benthic bioturbation and bioirrigation, and filter-feeding by benthic organisms (Griffiths et al., 2017).

3.2.1. Nutrient cycling

Cold-water coral reefs are hotspots of carbon cycling along continental margins (Van Oevelen et al., 2009). Oxygen uptake rates over coral reefs can be up to 20 times higher than those of the surrounding soft sediments, indicating consumption of strong organic matter and a downward flux exported from the surface ocean (Cathalot et al., 2015). The Maritimes region hosts the only known living coral reef complex in Atlantic Canada, which is located at the shelf break in the Stone Fence, at the mouth of the Laurentian Channel (*Lophelia* Coral Conservation Area). It is formed by several mounds of the scleractinian *Lophelia pertusa* that occur as live colonies, dead blocks and skeletal rubble (Beazley et al., 2021; Buhl-Mortensen et al., 2017a). In 2021, an expedition led by DFO-Pacific identified the presence of *L. pertusa* reefs in Finlayson Channel, within the central coast waterways of British Columbia. The visual survey captured 350 m of healthy reef between 180 to 210 m. The entire coral reef complex is likely much larger, with evidence suggesting there are more reefs in the surrounding region. Before this discovery, coral reefs were believed to be regionally extinct in Pacific Canada—with observations of *L. pertusa* limited to debris fields and the rare live colony (Cherisse Du Preez, pers. comm).

In the Disko Fan Conservation Area in the Arctic, the presence of bamboo coral forests (*Keratoisis flexibilus*) has been shown to increase nutrient fluxes (nitrate, ammonium, silica) released from the sediment, likely resulting from higher organic matter deposition allowed by their erected structure, enhancing the efficiency of benthic ecosystem functioning in this area (Pierrejean et al., 2020).

The globally unique glass sponge reefs in the Canadian Pacific are potentially a carbon sink as they are dense and can efficiently capture bacteria, and quickly process large volumes of water. Therefore, they may act as buffers for ocean acidification and climate change, as carbon can be sequestered into their tissues (Kahn et al., 2015). Glass sponges filter as much as 1% of the total water volume of the Strait of Georgia and Howe Sound combined each day (Kahn et al., 2015; Dunham et al., 2018a), and can remove up to 95% of bacteria and heterotrophic protists from the water they filter (Yahel et al., 2007). By feeding on bacteria from a variety of trophic subsidies (e.g., terrestrial and oceanic sources, potentially sediment-borne bacteria resuspended by tidal currents) sponges link the microbial loop with the benthic community (Kahn et al., 2018). Additionally, glass sponge reefs are substantial sinks of biological silicon, which can circulate back up to surface waters, linking these deep waters to diatom primary productivity at the surface (Chu et al., 2011). Glass sponge reefs also excrete ammonia waste, which acts as an important source of nitrogen for phytoplankton growth contributing to primary production (Kahn et al., 2015).

Dense aggregations of the glass sponge *Vazella pourtalesii* in the central Scotian Shelf (Sponge Conservation Area) also play a major role in the biogeochemical cycling of silicon, by accumulating large biogenic silica stocks both in the living population and in the sediments. A good portion of this biogenic silica deposited to the bottom after sponge death recycles as silicic acid before being permanently buried (Maldonado et al., 2020). This silicon, in the form of soluble silicic acid, is a key inorganic nutrient in the ocean, which modulates ocean primary productivity (Nelson et al., 1995; Tréguer et al., 1995) and the CO_2 exchange with the atmosphere (Mackenzie and Garrels, 1966; Tréguer and Pondaven, 2000).

In the Flemish Cap area, sponge grounds dominated by astrophorids form large-scale benthic habitats and have persisted for at least 17,000 years (Murillo et al., 2016). Based on modelling estimates (Pham et al., 2019), these sponge grounds revealed 231,136 t of wet biomass in an area of 135 057 km^2. Because these sponges filter 56,143 ± 15,047 million liters of seawater daily, consume 63.11 ± 11.83 t of organic carbon through respiration, and affect the turnover of several nitrogen nutrients, their removal would likely affect the delicate ecological equilibrium of the deep-sea benthic ecosystem (Pham et al., 2019). The economic value associated with seawater filtration by the sponges on Flemish Cap has been estimated to nearly double the market value of the fish catch (Pham et al., 2019). The sponge grounds found in the Flemish Cap (Murillo et al., 2012) have similar species composition and biomass as those found in the Newfoundland slope (Fuller, 2011) and eastern Arctic (Kenchington et al., 2016; Murillo et al., 2018b) and therefore, similar ecological functions are expected in these regions.

3.2.2. Bioturbation

The biogenic modification of sediments through particle reworking and burrow ventilation, defined as bioturbation, is a key mediator of many important geochemical processes in marine systems (Queirós et al., 2013). Sea pens and some soft corals (e.g., *Gersemia antarctica*, Slattery et al., 1997) can be important agents of bioturbation in soft sediments enhancing the mixing of sediment and particulate materials during their foraging and feeding. Corals and sponges can also influence macrofaunal diversity (described in section 3.4), which might have impacts on bioturbation activities performed by infaunal organisms. For instance, Pierrejean et al. (2020) found a higher proportion of infaunal surficial modifiers in soft sediments where dense aggregations of the sponge *Iophon koltuni* was present, in comparison to bare sediment areas. Infauna community bioturbators associated with the presence of bamboo corals in Baffin Bay were also found to influence benthic remineralization efficiency (Pierrejean et al. 2020).

3.3. PREDATOR-PREY INTERACTIONS

Glass sponges may represent an important food source for predators. Food webs have recently been studied in 19 glass sponge reef complexes on the Pacific coast using empirical data and published trophic interactions (Archer et al., 2020b). Sponges were found to be consumed by all species examined (e.g., crabs, nudibranchs, sea stars, squat lobsters and spot prawns) and contributed significantly to their diets. Further, community structure exhibited a threshold response to reef-building sponge cover – below a threshold of 8-13% live sponge cover, food webs are more clustered and less connected. This has implications for conservation and management, as reefs that are below this threshold will support different communities than reefs with higher percentages of live sponge cover (Archer et al., 2020b).

Several specific findings further suggest a trophic link between sponges and their associated community. The chiton *Hanleya hanleyi* was observed in European habitats displaying selective feeding and homing behavior on sponge species which also occur in eastern Canadian sponge grounds (Todt et al., 2009). In another case, stomach contents of the sea star *Ceramaster granularis* contained sponge spicules from the reef forming species the sea star lives on. Other sea stars are known predators of sea pens and soft corals (Birkeland, 1974; Weightman and Arsenault, 2002; Gale et al., 2013). Nudibranch have also been shown to prey on sea pens (Wyeth and Willows, 2006).

There is enough information available to indicate trophic links between cold-water corals, sponges and their surrounding communities. However, the paucity of studies on the dynamics of such interactions in most Canadian OECM suggest the need for further regional studies, considering the potential for modifications in the structure of interactions following closures. For instance, an increase in the abundance of predators is expected to have an impact on prey abundance (further commented in section 4.3.1).

3.4. HABITAT PROVISION AND INCREASED DIVERSITY

The *Lophelia* reef complex in the *Lophelia* Coral Conservation Area was subjected to heavy otter trawling for Redfish between 1980 and 2000 (Buhl-Mortensen et al., 2017a). In other areas of the North Atlantic, *L. pertusa* reefs have been shown to represent biodiversity hotspots for invertebrates (Frederiksen et al., 1992; Fosså et al., 2002; Henry and Roberts, 2007) and commonly support a high abundance of fish (Husebø et al., 2002; Costello et al., 2005) and different fish assemblages (Milligan et al. 2016).

Large gorgonians offer a variety of microhabitats for other organisms. Up to 114 invertebrate species were found associated with two common species (*Paragorgia arborea* and *Primnoa resedaeformis*) from Atlantic Canada (Mortensen & Buhl-Mortensen, 2005). De Clippele et al. (2015) reported several species associated with gorgonians and sea pens identified from *in situ* imagery. For instance, shrimps and the basket star *Gorgonocephalus* sp. were frequently observed living on gorgonians. Bamboo coral forests in the Disko Fan Conservation Area provide habitat for other invertebrates including sponges, anemones, crinoids, suspension filter feeders, as well as fish, including the Greenland halibut. These associations emphasize the importance of the presence of such structured habitat for other organisms in an otherwise low-relief, homogeneous environment (Neves et al., 2015b; Pierrejean et al., 2020). At least seven sponge species were also observed on and/or near bamboo coral forests during ROV surveys in this area (Dinn et al., 2020a), and Pierrejean et al. (2020) identified a greater diversity of infauna in sediment associated with bamboo corals, in comparison to bare-sediment at this location.

Similarly to the bamboo corals in Disko Fan, sea pen fields provide important structure in low-relief sand and mud habitats where there is little physical habitat complexity. Individual sea pen colonies as well as sea pen fields can provide refuge for small planktonic and benthic invertebrates (Birkeland, 1974; Baillon et al. 2015; De Clippele et al., 2015; Hamel et al., 2015), which in turn may be preyed upon by fish (Krieger, 1993). Redfish larvae have been found on sea pens, raising the possibility that sea pen fields are important nursery areas for commercial fish species (Baillon et al., 2012). A recent study on soft corals (previously in family Nephtheidae) has shown their importance providing habitat for juvenile basket stars (*Gorgonocephalus* sp.), which are conspicuous components of benthic communities in cold-water environments (Neves et al., 2020).

Glass sponge reefs provide valuable habitat for invertebrates and fish in Canada's Pacific Region. High species richness of epibenthic organisms associated with glass sponge reefs was noted in the Salish Sea as a result of ROV surveys (Dunham et al., 2018b) in Howe Sound and Hecate Strait (Cook et. al., 2005; Archer et. al., 2020a) and in Galiano Reef (Chu and Leys, 2010). Mobile fauna, such as squat lobsters and lithodid crabs can hide in and around live and dead oscula and feed on the surface of the reefs. Rockfish (*Sebastes* spp.) use the reefs for refuge either from predators or from the high water flow above the reefs (Chu and Leys, 2010). When comparing undamaged reef with a close-by damaged reef in the Strait of Georgia, it was observed that the undamaged reef had a higher concentration and diversity of juvenile rockfish, indicating that the reefs are foundational for many species (Cook et al., 2008). Rockfish also appear to use sponge gardens (i.e., aggregations of individual sponges) and reefs at different life cycle stages, the former supporting new recruits and the later juveniles and adults (Marliave et al., 2009). Acoustic recordings on and off the Outer Gulf Island sponge reef indicate the increased presence of fish and invertebrates on the reef compared to the off-reef recorder (Archer et al. 2018). The patterns of fish calls and observations of fish presence support the conclusion that megafauna abundance is higher on the reefs (Archer et al. 2018). The skeletons of dead glass sponges can act as a settlement surface for the larvae of other sponges (Guillas et al., 2019). Glass sponges can also have endosymbiotic relationships with other benthic organisms such as hydrozoan and zoanthids (Schuchert & Reiswig, 2006; Sanamyan et. al., 2012).

On the Scotian Shelf, populations of the glass sponge *Vazella pourtalesii* enhance species density and abundance of the associated epibenthic community (Hawkes et al., 2019). The megafaunal assemblage associated with these grounds was significantly different in composition and higher in species density and abundance compared to locations without *V. pourtalesii*.

In the Flemish Cap area, sediment samples with sponge spicule mats were associated with higher macrofaunal abundance and diversity and different community structure (Barrio Frojan et al., 2012; Ashford et al., 2019). This positive influence in the macrofauna was attributed to a local enhancement of organic matter and reduction in predation pressure (Bett and Rice, 1992; Ashford et al., 2019). Similarly, Beazley et al. (2013) found enhanced diversity and abundance of megafauna in areas with mixed-species sponge and similar results were found in the *Geodia* dominated sponge grounds (Beazley & Kenchington, 2015; Murillo et al., 2020a). Additionally, Klitgaard (1995) found 242 species of both epi- and infauna associated with sponges characteristic of the North Atlantic astrophorid sponge grounds, indicating the large diversity that these grounds support.

3.5. CONCLUSION

Cold-water corals and sponges provide several indirect BCBs. They aid in nutrient cycling, both by filtering the water for food and chemicals, which are used in building their structures. The corals and sponges themselves can also be a food source for fish and invertebrate species. Some corals (e.g., sea pens) enhance the cycling or mixing of sediment and particle materials.

Both corals and sponges also form important habitats in areas where there is little structure that could otherwise provide habitat for fish and invertebrates, increasing local species diversity. It is important to note that research on the indirect BCBs associated with corals and sponges is still limited. As monitoring activities and research on Canadian corals and sponges increase and improve additional benefits will likely emerge. It is likely that all the indirect BCBs mentioned in this section exist in some fashion in each of Canada's OECMs. The development of monitoring plans should consider this to be true even in the absence of data confirming localized indirect BCBs.

Summary:

- Indirect biodiversity conservation benefits (BCBs) are provided by both corals and sponges.
- Corals and sponges contribute to biogeochemical and nutrient cycling, enriching surrounding habitat.
- Corals and sponges can also be a source of food for other benthic organisms.
- Corals and sponges provide habitat for other species, which increases overall biodiversity. Increased diversity of bioturbators has also been associated with sponge aggregations.

4. ECOLOGICAL MONITORING INDICATORS

4.1. INTRODUCTION

The identification of ecological indicators is one of the most important steps in planning monitoring, as indicators will provide information on whether closed areas are effective. In the Northeast Atlantic, the Convention for the Protection of the Marine Environment of the North-East Atlantic (OSPAR) Commission provides advice on the selection of such indicators, and defines marine biodiversity indicators as 'any measurable feature or condition of the marine environment that is relevant to the stability and integrity of habitats and communities, the sustainability of ecosystem goods and services (e.g., primary productivity, maintenance of food chains, nutrient cycling, biodiversity), the quality and safety of seafood, and the status of amenities of socio-economic importance.'

Two main types of indicators have been identified by OSPAR (2012) which will be adopted here: state types and pressure types. State indicators refer to environmental conditions within a given geographical area, and include species, assemblage characteristics, and biotic functional groups, in addition to habitat characteristics, and physico-chemical properties (e.g., hydrodynamic parameters, nutrient levels) (Alexander et al., 2014). Pressure indicators refer to the anthropogenic pressures to which an area is exposed and may be used indirectly to infer the environmental condition (Noble-James et al., 2018). Various categories of indicators have been identified in DFO documents. Kenchington et al. (2012) used the OSPAR terminology (i.e., state and stressor indicators), while Lewis et al. (2016) suggested four main categories: biodiversity indicators, direct indicators, indirect indicators, and anthropogenic stressor indicators. Potential indicators for the monitoring of glass sponge reefs include sponge reef-based and sponge community-based indicators (Dunham et al., 2018a). In the case of the St. Anns Bank MPA, the monitoring framework was focused on four groups of indicators: background indicators (to provide information on the natural drivers), indicators of anthropogenic pressures, effectiveness indicators (provide information on the effectiveness of management actions), and socio-economic indicators (provide information on the socio-economic effects of the closed area) (DFO, 2014).

Noble-James et al. (2018) highlighted how the selection of state indicators is a very difficult task, and that indicator robustness is crucial for the success of a monitoring program. The selection of inadequate indicators may undermine the monitoring objectives and lead to erroneous conclusions. The selection of indicators should consider the available data on ecological parameters that might be suitable for monitoring.

DFO (2013) has provided guidance on the identification and prioritization of indicators for Operational Conservation Objectives to assess MPA and MPA network performance. That advice defines an indicator as a variable or pointer around which its fluctuations reflect key elements of the system of interest. For an indicator to be useful in determining change it must have a reference point, which could include a desired target state, a limit, or a risk tolerance value past which management action is required. The selection of indicators should be carefully considered by those with the appropriate expertise (DFO 2013).

According to that DFO guidance, the selection of indicators should be performed in seven steps:

1. Identify conservation objectives;
2. Identify suitable indicators;
3. Identify selection criteria;
4. Evaluate indicators;
5. Assess whether there is redundancy;
6. Agree on the final suite of indicators; and
7. Establish reference levels.

Although the advice was focused on MPA and MPA networks, it will also be used as guidance for OECMs in this document. However, not all steps are possible when considering available information on corals and sponges and baseline data for Canadian coral and sponge OECMs. This DFO advice suggested using the following eight criteria when assessing the appropriateness of indicators:

1. **Theoretical basis**: The indicator is based on concepts that are consistent with established theory;
2. **Measurement**: Data used to estimate indicators should be easily and accurately measured;
3. **Historical data**: Data from earlier time periods should be available, ideally with a time series of at least 10-20 years;
4. **Sensitivity**: The amount of change in indicator value corresponds to a change in the pressure (e.g., fishing, pollution);
5. **Responsiveness**: The type of response (linear, non-linear, random) of the indicator to the pressure, the timelines of the response, and the signal to noise ratio, (i.e., the data used to estimate the indicators should be measurable accurately enough that any change or trend in the indicator is greater than the variance in its measurement) can be reasonably measured;
6. **Specificity**: Indicators may be influenced by more than one pressure (e.g., fishing and temperature). How specific is the indicator to the pressure of concern? Can it be disentangled from other pressures (i.e., it is critical to know why an indicator is changing?)
7. **Public awareness**: The indicator should be easily understandable by non-scientists, and can be clearly communicated; and

8. **Cost-Effectiveness**: Sampling, measuring, processing, analyzing indicator data, and reporting assessment outcomes, should be feasible and within existing financial resources.

In this document we identify ecological indicators appropriate to monitor coral and/or sponge conservation areas. A list of indicators was compiled based on current literature on cold-water corals and sponges, and we describe each indicator using the criteria suggested in the DFO (2013) framework. The selection of monitoring indicators is a critical step in the management of conservation areas, and many factors need to be taken into consideration. We present the list of indicators after following steps 1-4 of the eight steps suggested in DFO (2013). Steps 5-8 will not be assessed in this document. Redundancy analyses (step 5) have been performed for indicators related to fish (Bundy et al., 2017), but such analyses could not yet be performed for corals and sponges due to data limitations. Furthermore, steps 5-8 are better performed at the regional level during the development of a monitoring plan, as they are specific to the conservation area, types and quality of data available. The final selection of suitable indicators should be conducted on a case-by-case basis, and the purpose of this document is to identify indicators that could be further considered during the development of local monitoring plans.

4.2. SELECTING INDICATORS

4.2.1. Step 1. Identify conservation objectives

Conservation objectives for each OECM are described in Table 1, but they can be summarized as follows:

1. Cold-water coral protection;
2. Cold-water sponge protection;
3. To conserve coral concentrations;
4. To protect corals and sponges;
5. To conserve sensitive benthic areas;
6. To protect globally unique concentration of the glass sponge *Vazella pourtalesii*;
7. To protect corals and sponges and contribute to the long-term conservation of biodiversity;
8. To protect the only known living *Lophelia pertusa* coral reef in Canada’s Atlantic waters;
9. To protect glass sponge reefs; and
10. To protect seamounts, hydrothermal vents and the ecosystems they support.

These objectives may be enhanced as the OECM program advances, potentially to include indirect BCBs.

It is important to acknowledge that as currently described, these conservation objectives are not operational because they are neither specific nor measurable. Past work by DFO has defined operational conservation objectives as those that deal with specific ecological outcomes, and that describe the desired states of key ecosystem components (DFO 2020c). The objective should also have a specific timeframe in which the outcome should be achieved in order to ensure a healthy ecosystem (DFO 2020c). The conservation objectives, as currently described for coral and sponge OECMs, are broad and will pose challenges to the selection of operational monitoring indicators. For the purposes of this document, we have focused on the OECM conservation objectives as they have been originally defined. However, we identified indicators in a more specific and measurable way, based on ways of monitoring metrics that could aid in our understanding of coral, sponge, and overall ecosystem trends in these OECMs. Therefore, setting SMART (Specific, Measurable, Achievable, Realistic, and Timebound) operational objectives will be an important part of this process (e.g., Wood, 2011).

4.2.2. Step 2. Identify suitable indicators

We started the selection of ecological indicators by reviewing suggested indicators related to coral and sponge conservation objectives and glass sponge and coral (*Lophelia*) reefs (step 1). While OECMs whose conservation objectives are focused on corals and sponges might share most indicators, coral and glass sponge reefs will have specific indicators that will not apply to other coral and sponge OECMs. It should be emphasized that a lack of international consensus on which variables to monitor has been identified as an important obstacle to the delivery of information on biodiversity change (Pereira et al. 2013). Choosing indicators for monitoring will be driven by site-specific factors and the realities of conducting monitoring.

Both quantitative, nominal, and qualitative indicators have been suggested in the literature. Smith and Hugues (2008) provided a list of potential indicators for the monitoring of cold-water coral and sponge communities in the UK. Quantitative indicators suggested by these authors, which could be relevant in a Canadian context, include:

1. Coral and sponge reef extent, density and biology,
2. Seamount diversity,
3. Extent/ density/ biology of deep-sea sponge aggregations (hexactinellid and demosponge aggregations) and octocorals, as well as
4. Community parameters (e.g., abundance, biomass, diversity, composition) (Smith and Hughes, 2008).

DFO (2010b) suggested four main indicators related to coral and sponges for the Gully MPA: coral distribution, density and size structure by species, proportions of live and dead corals, proportion of live corals with zoanthid over-growths, and the extent of over-growth in affected colonies. Kenchington et al. (2012) suggested 12 state indicators for the monitoring of corals and sponges in the Eastern Canadian Arctic, most of which would also be appropriate for other regions in Canada: abundance, biomass, distribution, diversity indices, size structure, live:dead ratio, percent zoanthid cover, patch area, patch density, patch isolation/proximity, patch connectivity, and patch dispersion.

Pereira et al. (2013) listed a series of candidate Essential Biodiversity Variables (EBVs) and EBV classes which could form “the basis of monitoring programs worldwide”. These include allelic diversity, abundances and distributions, phenology, taxonomic diversity, habitat structure, nutrient retention and other metrics within those.

For the Laurentian Channel MPA, potential indicators identified to monitor sea pens include four main classes:

1. Biodiversity indicators (community, ecosystem and productivity),
2. Direct indicators,
3. Indirect indicators (habitat characterization and oceanographic), and
4. Anthropogenic stressor indicators (Lewis et al., 2016).

Community indicators include species richness and diversity, while ecosystem indicators include population structure and abundance of other key species (not sea pens), benthic community structure/composition (infauna and epifauna), biomass of predator/prey species. Productivity indicators include chlorophyll and zooplankton variability. In terms of direct indicators, taxonomic diversity and richness, taxonomic abundance and density, and biomass by species were identified. Indirect indicators included habitat characterization, oceanography, and anthropogenic stressor indicators.

A suite of more specific indices that could potentially be used as indicators for the monitoring of glass sponge reefs in Pacific Canada include:

1. Reef-building-sponge-based (e.g., live sponge abundance, distribution, condition, and recovery potential), and
2. Community-based (e.g., community structure, indicator taxa of dense live and live reef, visible reef structure, no visible reef) (Dunham et al., 2018a).

The evaluation of these indices as potential indicators still requires the definition of clear conservation objectives and further knowledge of reef ecology and their response to stressors (Dunham et al., 2018a). The monitoring of broader ecosystem indicators related to glass sponge reefs has also been suggested in DFO (Dunham et al., 2018a) (e.g., oceanographic time series data, and sponge-specific data on nutrients, bacteria, and silicate).

Bell et al. (2017) considered that conventional diversity measures (e.g., abundance, richness) can provide important indicators of change but are less useful in terms of chronic impacts. Considering that, these authors identified several potential indicators for the monitoring of sponges, which in some cases could also be applied to other organisms. Among these indicators, those that would make a useful addition to the list suggested above and that are relevant to Canadian systems include: macro and micro morphological variation, competitive ability, predation, mortality, disease; genetic diversity, bioerosion rates, growth, reproduction, respiration, feeding, regeneration, pumping, mucus production, biochemical compounds/ secondary metabolites, associated microorganisms, and gene expression relating to long-term stress.

Non-quantitative indicators suggested by Smith and Hugues (2008) include nominal indicators, such as the ranking of ecological status (e.g., 0 = completely destroyed, 5 = pristine), and qualitative indicators such as skilled ‘eye’ appraisal associated with the same metric evaluated using the ranking indicator.

Stressor indicators suggested by Kenchington et al. (2012) include: distribution of fishing activities, aggregation of fishing activities, areas not impacted by mobile gears, timing and duration of anomalous events, timing of phytoplankton bloom, timing, duration and path of sea ice melt, and biomarkers. The stressor indicators identified in Lewis (2016) related to sea pens were: distribution of commercial fishing effort and compliance inside the MPA. An additional stressor indicator that could be considered in the context of Canadian coral and/or sponge OECMs is anthropogenic sedimentation related to anthropogenic activities (mostly trawling) outside of closed areas. Grant et al. (2019) showed that sediment plumes from bottom trawling happening >2 km outside of a closed area had an impact on glass sponges found within the adjacent closed area. Other potential stressor indicators to be considered are those related to oil and gas activities (distribution and chemicals), seabed litter, and activities related to submarine cables.

It should be mentioned that the identification of indicators does not imply that all of these indicators should be ultimately selected and/or used, but rather re-evaluated in the context of each individual OECM and their conservation objectives (Pomeroy et al. 2004). Furthermore, there is a difference between research being conducted for monitoring purposes versus other types of scientific research. While some indicators might not be selected for monitoring of an area, they might still be suitable to increase our general knowledge about the areas and species that inhabit. In some cases, data on certain indicators might be collected opportunistically, as part of other surveys without a focus on monitoring (e.g., RV surveys). Although these data might generate useful information about the system, their potential utility for monitoring will need to be assessed on a case by case basis.

4.2.3. Step 3. Identify selection criteria

The eight criteria for determining the appropriateness of indicators identified in DFO (2013) were described in section 4.1, and are: theoretical basis, measurement, historical data, sensitivity, responsiveness, specificity, public awareness, and cost-effectiveness. A table listing these criteria in relation to both state and stressor indicators is presented (Table 4). All pre-selected indicators for Canadian coral and sponge OECMs in general were evaluated using six of the eight criteria, excluding public awareness and cost-effectiveness. Public awareness was not assessed because all the indicators have the potential to meet it, depending on how the results are presented. Assessing general indicators not linked to a specific area did not allow us to evaluate the cost-effectiveness criterion, which can vary greatly depending on the location, size, and depth of an OECM.

The assessment of criteria suitability for each indicator is not a straightforward task and, in many cases, there are several exceptions that make it difficult. For instance, historical data is difficult to assess at a national level, but for some specific OECMs there might be good historical data available.

In assessing whether a potential indicator has a good theoretical basis (i.e., concepts are consistent with established theory), our approach was to identify whether an indicator has a good basis in the specific context of cold-water corals and sponges. Wherever we agreed that there is limited data regarding the theory behind the development of an indicator, we considered that there was a weak theoretical basis.

Where possible, we assessed indicators including thoughts on their cost. Although the DFO (2013) framework states that the “sampling, measuring, processing, analyzing indicator data, and reporting assessment outcomes should be feasible and within existing financial resources”, there is no doubt that there will be extra costs associated with monitoring. Furthermore, conservation objectives of coral and sponge OECMs across Canada are variable, OECM sizes are quite different, and they will require different strategies and tools, which will likely lead to some OECMs costing more to monitor than others.

4.2.4. Step 4. Evaluate indicators

From the assessment of potential indicator performance in relation to these criteria, we developed a list of indicators that we consider most suitable for the monitoring of corals and sponges and their indirect BCBs (Table 5). Among the pre-selected **state indicators**, we found that most had a good theoretical basis and public awareness qualities for the monitoring of corals and sponges. We found that availability of historical data was difficult to generalize, as it is variable across regions. Cost-effectiveness was evaluated in terms of whether collecting data for a given indicator can also provide data regarding other indicators. It should be highlighted that while cost effectiveness is an important criterion for the selection of an indicator, some indicators will be more costly than others to monitor. Monitoring plans might include both routine ‘inexpensive’ and targeted, more infrequent and experimental ‘expensive’ sampling efforts, which might vary in their frequencies. Particularly in the case of the latter, collaborations within and outside of DFO might facilitate data collection in a more cost-effective way.

Given that there is limited time, resources, and funding, and that many of these areas are difficult to study, we need to carefully consider how we design a monitoring program. A useful approach is the application of systematic conservation decision-making (CDM), which considers various monitoring scenarios with estimations of their benefits and drawbacks in order to determine the best approach under real-world constraints (e.g, Possingham 2001, Margules & Pressey 2000). CDM can help choose indicators that can detect real environmental change, be representative of other species in the environment, and cost-effective (Tulloch 2015). Schwartz et al. (2013) analyzed a number of CDM frameworks for their use in conservation planning. Bower et al. (2018) provides guidance on the use of three prominent frameworks (structure decision-making, systematic conservation prioritization, and systematic reviews) to determine the best method for various conservation problems, including advice where time and data are limited. The use of decision-making frameworks can help guide indicator selection and monitoring for corals and sponges while considering the realities that these areas are often data-poor and difficult and expensive to access.

While most indicators described here focus on direct BCBs, in several cases the same indicator can also be used in the context of indirect BCBs. To include potential indirect BCBs not explicitly associated with any of the described indicators, we created the indicator **indirect BCBs**.

4.3. STATE INDICATORS

4.3.1. Numerical abundance

In the context of this document, numerical abundance represents the counts of coral, sponges, and other benthic organisms. It is a well-established diversity metric (further referred here as abundance). Abundance is commonly used in calculations of diversity indices, and density (abundance per unit area) provides a measure of patchiness. It is one of the variables considered as a candidate EBV (Essential Biodiversity Variable), since it provides information on population trends (Pereira et al. 2013), and as changes in species abundance can lead to extinction and/or the shift or loss of ecological traits and functions (e.g., sediment bioturbation) (Keil & Jetz 2015; Solan et al. 2004). Areas with high densities of corals and sponges (and other taxa) often constitute significant benthic areas (SiBAs) or vulnerable marine ecosystems (VMEs).

Corals and sponges are sensitive to mechanical contact. Their removal from fishing influences population abundance, and it is expected that changes in abundance are likely to be a result of changes in fishing pressure. There is currently limited information regarding the impacts of oil and gas activities on coral and sponge abundance.

Historical coral and sponge abundance data are limited in many OECMs. For instance, most of the coral and sponge data available for the NL Region is based on DFO RV trawl surveys. Invertebrate abundance, including corals and sponges, is not one of the metrics consistently determined at sea. For some coral groups (e.g., large gorgonians, sponges), specimens from trawls are often too fragmented to be counted. Furthermore, trawl catchability is unknown or very low for most benthic taxa. For instance, sea pen catchability (Campelen trawl) in the Laurentian Channel has been estimated at only 5.2% (Kenchington et al., 2011). Therefore, there are clear limitations regarding the use of trawl data to determine coral and sponge abundance (historical or not). Recent studies have shown that imagery technologies can generate more accurate abundance data when compared with trawls and dredges (Ayma et al., 2016; Chimienti et al., 2018b). Abundance of corals and sponges from imagery has also been successfully quantified in Canadian waters (e.g., Leys et al., 2004, Neves et al., 2014, Devine et al., 2019, Dinn et al. 2020a; de Mendonca et al. 2021).

Abundance data should be fairly responsive to changes, as long as comparable gear is used to monitor over time. The use of downward-looking imagery might be important to depict the presence and accurate counts of juveniles in flat-bottom areas. Similarly, in the case of indirect

BCBs, the use of the same mesh size is important to accurately compare infauna abundances over time. Having a sampling design (section 5) that considers the inclusion of replicates might increase responsiveness. In fact, appropriate sampling design will be particularly crucial for this indicator, considering the patchy nature of corals and sponges, and the impact of sampling these vulnerable species.

Although coral and sponge abundance are expected to increase given the removal of benthic fishing pressure inside OECMs, they could still change as a function of bottom fishing taking place outside of OECMs, for example as a response to increased sedimentation (described as a separate indicator). Abundance could also vary as a response to changes in population dynamics resulting from the area being closed. For instance, sea pens can be predated upon by sea stars (e.g., Gale et al. 2013) and nudibranchs (Wyeth and Willows, 2006). Therefore, monitoring the abundance of other megafauna will also be important.

In terms of costs, abundance data should be obtained using imagery tools (see section 5), since trawls are not the most appropriate gear to assess coral or sponge abundance. The cost of using imagery tools includes the equipment and deployment in the field, and also for post-processing after the survey is finished (i.e., video/photo annotation). If resources for imagery surveys are not available in a particular year, it might be wise to delay data collection rather than to use less suitable gear (e.g., bottom trawls). If scientific trawl survey data are available (i.e., scientific surveys continue inside of OECMs), coral and sponge abundance data can be partially obtained, but caveats in terms of catchability and state of sample (e.g., damaged specimens, fragmented colonies, etc.) will need to be considered.

4.3.2. Biomass

Biomass can provide information on the flows of carbon and energy in the environment (Benoist et al. 2019). It is one of the variables considered a candidate EBV (Pereira et al. 2013) that, combined with specific biological traits, can be used to map different ecological functions (Murillo et al. 2020b). Body mass can be a proxy for other metrics such as numerical abundance and population structure (age/size class), and it can be used in secondary production models (Dolbeth et al. 2012).

Data from DFO RV trawl surveys have been used to identify areas with significant coral and sponge biomass concentrations, and the application of Kernel Density Estimation (KDE) has provided useful information regarding their distribution in eastern Canada (Kenchington et al., 2012). However, as for abundance data, there are also issues with interpreting biomass data collected using trawl gear, mostly due to unknown or low taxa catchability (Kenchington et al. 2011).

Specimen biomass, usually by species, is routinely determined at sea during most DFO trawl surveys (e.g., Rideout et al. 2024). Issues with using trawl biomass data have been highlighted above, and most regions do not identify specimens to the species level. To obtain biomass from imagery data, researchers need to use methods such as conversion of size into weight or Length-Weight relationships (LWRs). This requires considerable work and might not be feasible at the scale of Canadian OECMs because of factors such as the potential regional variations in LWR relationships for different taxa. Relationships between coral and sponge metrics like size and wet weight can be obtained based on trawl samples (e.g., Murillo et al. 2018a), but the application/input of imagery data into these equations will require considerable work due to the high species diversity and potential regional variability. As a successful example, Metaxas and Giffin (2004) were able to calculate ophiuroid biomass from imagery data based on metrics of ROV-collected samples from the same area. In another recent study, Benoist et al. (2019) described the generalized volumetric method (GVM) for calculating biovolume as a predictor of biomass, which does not depend on the availability of physical specimens, uses surface area from imagery, and was shown to have better predictive power than LWRs. The potential of GVM might be more limited for corals and sponges than other benthos due to their 3-D structure.

Biomass data from trawl surveys would likely to continue to be collected only if trawl surveys continue inside of OECMs. If smaller trawl gear is used to collect samples for biomass assessments (e.g., Agassiz trawl, section 5), the new data would not be directly comparable to data previously collected using RV trawls, although new studies could be developed to assess comparability. If biomass data continues to be obtained from trawl surveys, then there would be no extra costs associated with those collections and sample processing. Imagery data collected for abundance would also be used for measurement of size and potential conversions to biomass.

Like numerical abundance data, coral and sponge biomass would be expected to increase following the closure of areas to fishing, due to removal of the fishing pressure. Biomass would increase from either an increase in abundance and/or from specimens potentially growing larger. Therefore, biomass is sensitive to changes in fishing pressure. If the biomass data are to be used as a result of conversions from relationships between metrics obtained from imagery (e.g., size vs weight), then responsiveness of biomass data might be low as there might be considerable variation in these measurements.

Changes in biomass would likely be a response to the removal of fishing pressure. However, if there are changes in abundance following other factors as described above (e.g., predation), then changes in biomass would follow.

4.3.3. Distribution

Distribution data can provide information on ecosystem resilience, ecosystem function, and genetic diversity (Kenchington et al., 2012). Coral and sponge distribution data in Canadian and surrounding waters have mostly been based on DFO RV trawl survey presence data (Wareham and Edinger, 2007; Murillo et al., 2012; Kenchington et al., 2016; Gullage et al., 2017), although imagery surveys have also contributed information on their distribution (e.g., Baker et al. 2012; Neves et al. 2014; Beazley and Kenchington 2015; Du Preez et al. 2015; Dinn et al. 2020a; Law et al. 2020). It is one of the variables considered as a candidate essential biodiversity variable (EBV) (Pereira et al. 2013).

Determining coral and sponge distribution within an OECM can be relatively straightforward, as it represents one of the results from the collection of several other metrics (i.e., abundance data requires taxonomic identification associated with the samples). The main challenge associated with distribution data lies in the level or precision of taxonomic identifications. For instance, physical samples are required for identification at low taxonomic levels. Taxonomic identifications can still be determined from imagery, but usually at higher levels (discussed in section on imagery). Additionally, distribution data from multispecies surveys may be subject to contamination from one trawl set to the next one. Different invertebrate organisms such as sponges and some types of corals can remain hooked to the trawl net or in other parts of the vessel and appear posteriorly in the sorting process of the next trawl set catches (Kenchington et al. 2016).

Distribution data are somewhat sensitive to changes in pressures. Since baseline data on coral and sponge distribution in Canadian waters are mostly based on trawl survey presence data, absences in future surveys might not represent true absences. Gear selectivity and type are important issues to be considered in this regard. For instance, imagery data have shown the presence of certain sea pens rarely collected in trawls due to either low gear catchability or sea pen behavior (e.g., *Kophobelemnon* sp. and *Protoptilum* sp. in the Laurentian Channel MPA).

On the other hand, new presence data and new distribution records might be an indication of taxa expanding their known distribution potentially due to protection measures that remove fishing pressure. Even if distribution ranges do not change at large spatial scales, taxa not detected in an area previously might be detected as a result of removing the fishing pressure. We expect that an increase in the use of imagery will also lead to spatial and bathymetric range extensions in taxa distribution. Physical samples will likely be required to confirm such extensions.

The responsiveness and specificity of distribution data are not very high. Since coral and sponge distribution data are still lacking for areas of all OECMs, identification of changes in species distribution might be confounded by our lack of baseline information in the first place. However, removal of pressures might allow larval settlement and development of taxa not previously found in an area. Habitats suitable for cold-water corals are also expected to shift as a result of climate change (Morato et al., 2020). Distribution data can be collected along with the collection of data for other indicators, so cost-effectiveness is high.

4.3.4. Diversity indices

Diversity indices can provide information on biodiversity, which is related to ecosystem resilience and ecosystem function (e.g., Solan et al. 2004; Gamfeldt et al. 2015; Strong et al. 2015). The calculation of diversity indices will follow the collection of data on abundance and taxonomic richness.

As mentioned earlier, it is expected that in some cases, taxa richness could increase with the increased use of imagery data. This is particularly true if collections are used to validate taxa identification from imagery. On the other hand, the collection of imagery data without the collection of associated specimens might lead to a decrease in taxa richness due to the challenges associated with taxa identification from imagery (i.e., using high taxonomic levels which represent multiple species).

If changes in diversity are mostly a response to changes in pressure, there would likely be an increase in overall diversity of an area, if assessed using comparable gear types. Trawl-collected diversity data should not be compared to imagery data. Similarly, imagery data collected using the same gear type but at different resolutions (e.g., camera, spatial) and distance from the seafloor will not be directly comparable.

4.3.5. Size structure

The size of corals and sponges can provide information on their reproductive stage (Baillon et al. 2015), age (Hamel et al., 2010; Neves et al., 2015a; Murillo et al., 2018a; Neves et al., 2018), and population structure (D'Onghia et al., 2010; Gori et al., 2013). Body size is also often related to abundance (White et al. 2007) but these relationships have not yet been explored in the context of coral and sponge height/length in Canadian OECMs. This indicator would focus on individuals or individual colonies, which could mean that it may not be cost effective over large spatial areas.

Obtaining cold-water coral or sponge size data is not straightforward. As mentioned earlier, data from multispecies trawl surveys do not capture all sizes of corals and sponges, as catchability is unknown/low and mesh size of the cod-end is large enough for small specimens to be lost. Another factor is that coral and sponge size data are not routinely collected during these surveys, meaning that there is no available baseline data on coral and sponge size for most taxa and regions. Size metrics have been obtained for a few coral taxa in specific regions, but these might not represent data from specific OECMs (e.g., Baillon et al. 2016; Murillo et al. 2018a).

Finally, measuring specimens from imagery data can be complicated by the three-dimensional nature of many of these organisms. While measuring specimens from downward-looking cameras would not be suitable in most cases (although suitable for cup corals, some sponges), measuring specimens from forward-looking cameras depends on laser points being close enough to the specimens to allow for accurate measurements. While precise measurements might be difficult to obtain, colony/individual size can still be determined qualitatively using general size classes. For instance, Bennecke and Metaxas (2017) mentioned that no colonies <20 cm in height were found in their study area, indicating that no recruitment was taking place. In the case of some sea pen taxa, the number of polyps (or polyp leaves) can be used as a proxy for colony size, if colonies can be seen in detail from imagery (Chimienti et al., 2019).

Colony/individual size is generally a proxy for age. Therefore, the presence of colonies of specific size classes can indicate recent recruitment events or an established/pristine population. Changes in values for coral and sponge size would likely represent changes in recruitment associated with changes in pressure. However, our knowledge regarding population dynamics and fitness of most cold-water coral and sponge taxa in their early life stages is rudimentary.

As mentioned above, obtaining the size of corals and/or sponges from imagery is generally time-consuming, which is a challenge in the context of monitoring large areas. The size-structure indicator should therefore be developed to focus on measurable taxa using available gear guided by validated methods in the literature. For example, if only downward-looking imagery are available, then measuring the heights of three-dimensional corals and sponges might not be the best use of such imagery, but measuring other metrics as proxy for size might be possible (e.g., sea pen polyp leaves by Chimienti et al. 2019, osculum diameter in some sponge species (Chu & Leys 2010, Kahn et al., 2016). The size-structure of cup corals, which lay relatively flat on the seafloor, might allow measurements of width, and measuring sponge oscula might also be possible from downward-looking imagery (Maldonado et al. 2020). In addition, considering size classes (e.g., small, medium, large), as opposed to detailed measurements, might be a more practical option to be considered in monitoring efforts.

4.3.6. Proportion of live and dead corals (live:dead ratio) and condition – also considered a stressor indicator

This indicator has been suggested for monitoring of the Gully MPA (DFO, 2010b) and by Kenchington et al. (2012). While the identification of dead reef-forming corals might be relatively straightforward due to their long taphonomic degradation times and massive, easily visible structures, non-scleractinian corals such as sea pens and some gorgonians might degrade much faster, leaving no trace on the seafloor at the scales of monitoring (e.g., annually or <10 years). Dead colonies of certain gorgonians like *Primnoa* spp., which have a mixed skeleton composed of solid calcite and protein, can remain on the seafloor for centuries or even thousands of years after their death (Edinger and Sherwood, 2012). On the other hand, other large gorgonians such as *Paragorgia* spp. decompose much faster (e.g., 14 months), due to their porous skeletons (Edinger and Sherwood, 2012). Graveyards of the solitary cup coral *Desmophyllum dianthus* have been identified in the Orphan Knoll, Northwest Atlantic, some as old as 180 ka (e.g., Edinger and Sherwood 2012, Maccali et al. 2020). Potential markers to assess condition in Canadian coral and sponge species have not yet been defined.

Degradation rates of most cold-water coral taxa (non-Scleractinian) and individual sponges in Canada are largely unknown. Sea pen skeletons are not commonly seen or reported on the seafloor. Documenting coral skeletons on the seafloor in most cases requires observation within months after the impact (Edinger and Sherwood, 2012). The identification of recently dead colonies (i.e., still partially bearing tissue) is also a variable of interest. The proportion of live and dead corals based on trawl survey specimens can only be determined based on skeletal evidence (e.g., presence of colonies completely lacking tissue, taphonomic evidence of bioerosion and boring organisms). However, this information is usually not collected during DFO trawl surveys, and therefore no consistent baseline data are available.

In terms of the reef-forming coral *Lophelia pertusa*, quantifying the proportions of live and dead coral is considered essential to long-term monitoring programs (Vad et al., 2017). Using a remotely operated vehicle (ROV), Vad et al. (2017) measured *L. pertusa* colony size (N = 18) and dead/living layer size. These authors suggest that a clear visual contrast between living and dead portions *L. pertusa* colonies can facilitate visual monitoring.

In the case of glass sponge reefs, exposed dead glass sponge skeletons can be an indicator of a well-functioning sponge reef ecosystem. The proportion of live and dead glass sponges is also a variable of interest, and it can be quantified from imagery (Dunham, et al. 2018a). Reef building sponge-based and community-based suites of indicators have been proposed for monitoring glass sponge reefs (Dunham, et al. 2018a).

This indicator might therefore be focused on specific taxa such as large gorgonians and scleractinians with long-lasting skeletons. Noting the presence of any dead corals and sponges, including *Paragorgia* spp. and sea pens, during imagery annotation can provide additional information about mortality in the system and assist with the development of this indicator over time.

4.3.7. Percent of coral colonies colonized by zoanthids – also considered a stressor indicator

This indicator refers to the proportion of live corals that show zoanthid over-growths, and the extent of over-growth in affected colonies (Kenchington et al., 2012). Zoanthids are benthic cnidarians similar to sea anemones in their gross morphology. In some cases, zoanthids can parasitize corals, covering up their bodies and/or skeletons. It is considered an indicator of health status, since zoanthid growth directly implies coral tissue death. However, it is difficult to determine whether their presence necessarily leads to coral death or whether the coral can remain alive and healthy despite the presence of the zoanthid. The zoanthid *Epizoanthus* sp. has been reported on the gorgonian *Primnoa resedaeformis* (Buhl-Mortensen and Buhl-Mortensen, 2004) in the NE Channel (NW Atlantic), while *Epizoanthus norvegicus* has been observed on both *P. resedaeformis* and *Paragorgia arborea* on the Norwegian coast (Carriero-Silva et al., 2011). An unidentified zoanthid has also been observed from ROV imagery covering colonies of the bamboo coral *Keratoisis flexibilus* in Disko Fan, Baffin Bay (Disko Fan Conservation Area; B. Neves, personal communication). Mortensen et al. (2005) revealed that zoanthid cover per colony (in infected colonies) averaged 60% and that they were more common on intact colonies (in relation to damaged ones).

This indicator might not be relevant for all coral groups or sponges. However, parasitic zoanthids have been reported covering black corals (Suarez et al., 2015) and zoanthids have been reported in association with sponges (e.g., Bulh-Mortensen et al., 2017b), although the nature of the relationship in the latter has not been determined (i.e., parasitism, commensalism). Accurate measurement of the proportion of colonies with parasitic zoanthids is possible from imagery, particularly for taxa such as *P. resedaeformis* and *P. arborea* that have a single point of attachment to the substrate (i.e., hard). In the case of *K. flexibilus* in Disko Fan, colonies are so densely distributed that identification of individual colonies is generally not possible (Neves et al., 2015b). However, zoanthid presence should always be reported if observed, as it could be a sign of declining health/increased stress of the corals.

4.3.8. Patch area and density (for sponge grounds, sea pens, large and small gorgonian corals)

Patches are considered as areas of concentration of a taxa over a small scale and provide information about fragmentation of the distribution of a taxa. Assessing changes in patch metrics (e.g., area and density) can provide information on the status of a population. Corals and sponges can have a patchy distribution. This is particularly clear in the case of those groups that can be found in high densities over small (or large) areas (e.g., sea pens). In terrestrial ecosystems, patch area has been strongly correlated with species richness and the occurrence and structure of some species (Maseko et al. 2020 and references therein). Some anthropogenic activities, such as bottom-contact fisheries, can decrease patch area by removing the corals and sponges. This patch size decrease will lead to a loss of the indirect BCBs associated to these organisms and can increase susceptibility to predation and competition from an invading alien species, as have been shown in terrestrial ecosystems (e.g., McIntyre, 1995; Gaublomme et al. 2008).

Kenchington et al. (2012) suggested patch perimeter or area and patch density (number of patches per unit area) as good indicators, as they generally have low variance, making statistical analyses more robust. However, high inter-annual variation was identified for patch area. This variable could still be used, but at higher temporal scales (e.g., re-assessed every three years, rather than annually), or with enough data to be able to use inter-annual variation as a model parameter (Kenchington et al., 2012).

In the Northwest Atlantic and Eastern Canadian Arctic, areas of significant concentrations of corals and sponges have been identified based on trawl survey biomass data thresholds (Kenchington et al., 2016). For instance, the most recent assessment by NAFO led to the identification of 100 kg of sponges as the threshold to consider a significant concentration of sponges in the NAFO Regulatory Area (NRA) (Kenchington et al., 2019a). Thresholds are revised as new biomass data become available from annual scientific surveys in those areas. The definition of a patch, however, is critical for the use of these metrics.

We considered that the theoretical basis for patch metrics is not yet well-developed. The definition of significant concentrations in the NW Atlantic and Eastern Canadian Arctic is based on trawl biomass data, for which there are issues (highlighted earlier). Furthermore, those thresholds were calculated at the functional group level and should not be generalized to all regions and depths, as taxonomic composition can vary, influencing metrics of interest (e.g., the weight of whip-like vs. keel-like sea pen differs). The definition of a patch based on abundance/density data still requires investigation. Preliminary patch size thresholds for abundance data calculated from biomass data have been suggested in the context of oil and gas exploration activities in NL (DFO 2020d). Similar exercises will need to be performed to identify further abundance thresholds that can be used to define a patch from imagery.

Once a patch has been defined, calculations can be performed using GIS methods. The patch metrics calculated by Kenchington et al. (2012) were based on trawl data. Patch calculations can also be performed from imagery data, likely same imagery used for abundance. The sensitivity, responsiveness and specificity qualities for patchy area and density would be similar to those for abundance data. However, ecological dynamics might differ among patches due to their different sizes, shapes, exposure, and physical environment, for example.

Extensive sampling may be needed for some coral and sponge groups that form large patches, particularly in areas for which very limited data are available (e.g., deep-water portions of some OECMs).

4.3.9. Patch isolation/proximity

This indicator refers to the tendency for patches to be relatively isolated in space from one another (Kenchington et al., 2012). The proximity index (PX) as described by Gustafson and Parker (1994), can be used to quantify the spatial context of a habitat patch in relation to its neighbors. Large values of PX are indicative of patches that are large and close together, whereas a small PX can indicate a more fragmented space of small patches. Sensitivity, responsiveness and specificity qualities would be similar to those for patch area and density.

4.3.10. Patch connectivity

This indicator refers to larval connectivity between patches (i.e., populations). Understanding population and landscape connectivity is an important consideration in the development of place-based conservation measures (e.g., Gallego et al., 2017; Van Wyngaarden et al., 2017). Larval dispersal can be predicted using biological parameters combined with physical oceanographic models of the direction and speeds of currents (Young et al., 2012), although information regarding early reproductive biology for most cold-water corals and sponges is still lacking. For instance, Metaxas et al. (2019) used the Finite-Volume Community Ocean Model (University of Massachusetts-Dartmouth) to assess hydrodynamic connectivity between canyons in the Scotian Slope with known occurrences of *Paragorgia arborea* and *Primnoa resedaeformis*. Whereas, Lagrangian particle tracking has been used in the Flemish Cap area to study the connectivity among areas closed to protect vulnerable marine ecosystems (Kenchington et al. 2019b; Wang et al. 2019). Assessing spatial autocorrelation is also a way to investigate whether patches are related/connected. For instance, spatial correlation of glass sponge reef patches has been assessed using underwater imagery transects (Chu & Leys, 2010). In some cases, patchiness may encourage overall diversity of organisms by creating diversity of habitat types between patches. However, this might not be the same in the context of unwarranted habitat fragmentation, where low diversity, between-patch areas might represent remnants of once flourishing habitats.

4.3.11. Patch contagion index

This indicator refers to the patch dispersion index suggested by Kenchington et al. (2012). It is the tendency for patches to be regularly or contagiously (i.e., clusters) distributed with respect to one another (Kenchington et al., 2012). High contagion results from areas with a few large, contiguous patches, while lower values generally characterize areas with many small patches. There are available formulas to calculate contagion index (Ritters et al., 1996). Sensitivity, responsiveness and specificity qualities would be similar to those for patch area and density.

4.3.12. *Lophelia* reef extent

The *Lophelia* Coral Conservation Area is the only site known to host live colonies of *Lophelia pertusa* in Atlantic Canada, where colonies are found forming small mounds on the seafloor (< 3 m high) (Buhl-Mortensen et al. 2017a; Beazley et al. 2021). Research suggests that these mounds should be detectable in multibeam data with a 5 m horizontal resolution (Roberts et al. 2005; Buhl-Mortensen et al. 2017a). The spatial extent of live *L. pertusa* cover has been assessed by Huvenne et al. (2016) using ROV imagery and acoustic technologies (MBES; revisited in section 5). In Southeast Greenland, colonies of *L. pertusa* are found forming clusters on a vertical wall, in such a way that identification and quantification of individual colonies might be possible from imagery (Jalim, 2020).

4.3.13. Glass Sponge-reef indicators

Several glass sponge reef-specific indicators have already been assessed and suggested in the context of monitoring of glass sponge reefs (DFO, 2017b). These include direct and indirect BCBs: live sponge abundance, distribution, and condition, recovery potential, community structure, indicator taxa of live sponge reef and reef structure habitat categories (no visible reef, dead reef, mixed reef, live reef). Details on these specific indicators can be found in DFO (2017b).

4.3.14. Indirect BCBs indicators

Many of the state indicators detailed above can apply to indirect BCBs. For example, abundance, biomass, distribution, diversity indices, and size structure could also be indicators applied to indirect BCBs. Examples of additional potential indicators to measure indirect BCBs include species associations (e.g., Redfish and sea pens), infauna diversity, sponge oscula density and area (i.e., sponge filtration rate proxies), contribution to biogeochemical cycles. Regions will need to choose appropriate indicators for indirect BCBs from known or expected species in the area. Information in Objective 2 could be utilized in the development of indirect indicators specific to each OECM.

4.3.15. Environmental indicators

Collecting environmental data is crucial to understand trends and changes in the area, which may or not be related to their protection. Similarly, changes in some of the ecological indicators could be a response to environmental changes. These are broader indicators, focused on environmental and physical data, and can include: habitat-related parameters, oceanographic data (e.g., CTD parameters), oxygen, chlorophyll *a* concentration and zooplankton diversity and abundance. In the case of sponge focused OECMs, bacteria and silicate are also variables of interest. Climate change-related indicators are discussed in the next sections.

4.4. STRESSOR INDICATORS

Stressor indicators discussed here were mostly obtained from the list suggested by Kenchington et al. (2012), with the addition of four new indicators. Stressor indicators suggested by Kenchington et al. (2012) are: distribution of fishing activities, aggregation of fishing activities, areas not impacted by bottom-contact gears, timing and duration of anomalous events, timing of phytoplankton bloom, and timing, duration and path of sea ice melt. To those we added: distribution of oil and gas activities, anthropogenic sedimentation deposition, chemical impact related to oil and gas activities and seabed litter.

Assessing sensitivity and responsiveness for stressor indicators can be less applicable than for state indicators. The sensitivity criteria examines whether the amount of change in indicator value corresponds to a change in the pressure. In the case of the stressor indicator “distribution of fishing activities” for example, assessing sensitivity would mean assessing whether “changes in the distribution of fishing activities that result from protected areas correspond to a change in the distribution of fishing activities”. While distribution of commercial bottom fishing activities inside the conservation areas are expected to be non-existent, activities outside of those areas continue, and perhaps more intensively as fishing grounds have been displaced. Furthermore, the possibility of non-compliance should not be excluded, and fishing might still take place inside of conservation areas. While investigating compliance is not part of the ecological monitoring, keeping track of stressor indicators can be useful as part of the latter.

4.4.1. Distribution and aggregation of fishing activities

As mentioned earlier, although coral and sponge OECMs do not permit commercial bottom fishing, this type of fishing exists outside of closed areas. Trawling outside of closed areas can still influence nearby areas (e.g., Grant et al., 2019), and the possibility of non-compliance should not be neglected.

There is a good theoretical basis for the use of this indicator. Commercial bottom fishing activities can be confidently identified from Vessel Monitoring System (VMS) and logbook data. These two sources of data are more powerful when combined, as there can be limitations within both data types (e.g., vessel speed not always available from VMS data). Commercial vessel speeds are an indication of fishing activity status such as fishing or transiting (e.g., Koen-Alonso et al. 2018). Automatic Identification System (AIS) is another way to obtain vessel positioning data, and it is now mandatory in the European Union for vessels >15 m to assist with maritime surveillance and safety (Natale et al., 2015). While VMS provides point-to-point satellite communication between the ship and the ground-based centers, AIS communications are broadly broadcasted and can be received by other ships in the area (Natale et al., 2015). Both technologies have been advocated and used in the context of tracking fishing effort (Lee et al., 2010; Gerritsen and Lordan, 2011; Chang and Yuan, 2014; Mazzarella et al., 2014; Natale et al., 2015; Le Guyader et al., 2017; Dunn et al., 2018; Koen-Alonso et al., 2018; Guiet et al., 2019).

The distribution of fishing activities is an indicator of the spatial extent of fishing activity. To produce fishing effort maps, both VMS and/or logbook data can be used (see details in Koen-Alonso et al. 2018). It would be based on the total area of grids (1 km × 1 km) with fishing activity based on the VMS records, at specific temporal resolutions (e.g., annual or monthly data) This indicator would be reported in conjunction with the indicator for ‘Aggregation of fishing activities’.

The aggregation of fishing activities is an indicator of the fishing effort intensity and would be reported in conjunction with the indicator for ‘Distribution of fishing activities’. It would be based on the total area of grids (1 km × 1 km) within which the top 20th percentile bin of fishing activity from the VMS records (e.g., annual data). This represents the area where the most intense fishing has occurred (Koen-Alonso et al., 2018).

Measuring distribution and aggregation of fishing activities is relatively straightforward using GIS tools, but obtaining commercial fishing data (i.e., VMS, AIS, and/or logbook) at the scale of Canadian OECMs might be challenging. Changes in the distribution of fishing activities outside of the closed area could be related to fishing grounds being displaced. Responsiveness of this variable would likely be low, as changes in distribution of fishing could be a response to external factors that are not related to the conservation area. On the other hand, conservation benefits of conservation areas could have an impact on the environment outside of the area (e.g., increase in fish abundance), which could have an impact on fishing activity outside of the area.

It should also be noted that scientific fishing (e.g., trawling and longline) by DFO and their partners might continue inside of OECMs, which takes place at much smaller scales than commercial fishing (e.g., DFO, 2020b). Therefore, the information described above in terms of tracking fishing activities should also apply in these cases. For example, in the Newfoundland Region, the potential authorization of scientific bottom-contact activities such as fishing inside of an OECM requires the development of a detailed activity plan by the proponent, for evaluation by DFO’s Marine Planning and Conservation (MPC) branch. Information generated from these scientific surveys on fishing effort, distribution, and invertebrate catch (particularly of corals and sponges) should be utilized to track these activities in the OECM of interest.

4.4.2. Areas not impacted by bottom-contact gears

This indicator provides information on the area of the seabed that has not been impacted by bottom fishing gear (Kenchington et al., 2012; Piet et al., 2012). It was listed as a stressor indicator by Kenchington et al. (2012) but suggested by Piet et al. (2012) primarily as a state indicator, because it might take a long time to detect changes in habitat/seafloor integrity resulting from the absence of bottom-contact gear impact. In this document we list it as a stressor indicator to keep in line with Kenchington et al. (2012) but acknowledging that it might not be possible to report on this indicator at high temporal frequencies usually done for other stressor indicators (e.g., annually). Kenchington et al. (2012) suggest that if fishing activity is not intense in the area, annual reporting might not be necessary.

As per the distribution and aggregation of fishing activities indicator described above, there is a good theoretical basis for the use of this indicator, since seafloor integrity can be directly impacted by bottom-contact gears. Furthermore, trawl mark intensity can be obtained from acoustic data (e.g., Huvenne et al., 2018), proving to be an additional method to obtain information on the bottom fishing status of an area. Sediment profile imagery (SPI) has also been suggested as a potential technique to allow identification of trawl marks and other seafloor disturbances (Germano et al., 2011; Rosenberg et al., 2003; Smith et al., 2003). SPI uses an imaging device in an inverted periscope (optical prism) to obtain images of upper sediment layers from which physical, chemical, and biological characterization is possible, usually in combination with macrofaunal and geochemical data (reviewed by Germano et al., 2011). Imagery data have also been used to identify trawl marks (e.g., Roberts et al., 2000).

The use of VMS, logbook, acoustic data, and SPI can be a powerful combination to monitor this indicator. However, if seafloor acoustic data or imagery have not been obtained before areas have been closed to fisheries, then use of an acoustic method to detect trawl marks will likely require additional data (e.g., VMS) to validate the interpretation from the acoustics, because trawl marks on the seafloor fade over time. Studies have shown that trawl mark persistency on the seafloor can vary between 6 and 18 months, and depend on sediment grain size and hydrodynamics (e.g., greater longevity in finer-grained sediments and low-energy environments) (Schwinghamer et al., 1998; Tuck et al., 1998). Furthermore, if scientific trawl surveys (i.e., DFO RV surveys) continue inside of closed areas, presence of trawl marks from such surveys could be confounded with those from commercial fishing, despite differences in towed distance, trawl metrics (e.g., door spread, weight), and trawl mark depth into the sediment.

The amount of change in an area of the seabed not impacted by bottom-contact gear is expected to be a direct response to the removal of commercial fishing from closed areas; hence sensitivity and specificity of this indicator are both high. Responsiveness will likely depend on the type of data being analyzed, as well as quality and amount of baseline available data. For instance, the use of acoustic data alone to identify trawl marks might generate considerable noise. On the other hand, the use of VMS and logbook data have been shown to be particularly useful to identify fished areas and fishing effort (Lee et al., 2010; Gerritsen and Lordan, 2011; Chang and Yuan, 2014; Koen-Alonso et al., 2018).

4.4.3. Distribution of oil, gas, and seabed mining activities

Oil, gas, and seabed mining activities are not explicitly prohibited in Canadian OECMs and will be evaluated on a case-by-case basis for each potential site within an OECM to determine if they are consistent with the conservation objectives of a specific area. Although no active licenses exist for oil and gas production activities inside Canadian OECMs, as of 2020 there were 30 active exploration licenses within the Newfoundland region alone, with some of these falling within the Northeast Slope Marine Refuge, as well as NAFO fishing closures outside of the Canadian EEZ (Gullage et al. 2022). Exploration licenses allow industry to perform exploratory drilling in areas where seismic surveys have indicated potential for hydrocarbon extraction (Cordes et al., 2016). Exploration activities might or not lead to oil and gas production; yet, they have the potential to impact cold-water corals and sponges through several steps of drilling activities (Cordes et al., 2016; Gullage et al., 2022).

The distribution of oil and gas activities is an indicator of their spatial extent (e.g., abundance, distance to features of interest) in relation to OECMs. It would be based on the total area of grids within which exploratory or production wells are known to occur, each year.

The measurement of the distribution of oil and gas activities (e.g., spatial data for wells, pipelines, etc.) would be relatively straightforward to perform (i.e., smaller spatial scale than commercial fishing). Even in the case of oil and gas activities taking place outside of closed areas, knowing their location in relation to the closed area is important, as activities might still have the potential to impact organisms kilometers away from the source (e.g., drilling cuts). Sensitivity, responsiveness, and specificity do not need to be assessed against this indicator. The distribution of oil and gas activity sites would not be expected to change as a direct response to changes (decrease) in fishing pressure (while the opposite might be more common, see Rouse et al., 2020). However, the potential for physical interaction between activities from these two industries does exist (e.g., Rouse et al. 2018, 2020) and it should not be dismissed.

Other emerging industries such as seabed mining can also have an impact on corals and sponges. Seabed mining usually takes place in areas where polymetallic nodules, sulphides/vents, and cobalt-rich crusts have been identified (Miller et al. 2018). While no such areas have been identified for most of Atlantic Canada’s and Eastern Canadian Arctic’s EEZ, areas with polymetallic sulphides/vents and cobalt-rich crusts can be found in Pacific Canada (Miller et al. 2018). Although mining might take place in areas beyond national jurisdiction, the plumes generated from these activities can still impact benthic organisms such as corals and sponges (Gollner et al. 2017) hundreds of kilometers away from the source (Christiansen et al. 2020, Drazen et al. 2020).

4.4.4. Anthropogenic sediment deposition

This indicator refers to sediment deposits produced or transported as a result of anthropogenic activities (e.g., trawling, oil and gas exploration, mining). The sediment itself might be natural (e.g., sediment resuspended during bottom trawling activities) or man-made (e.g., drill muds produced during oil and gas exploration). It can have an autochthonous (generated at the source of impact) or allochthonous (transported) origin.

Corals and sponges can be sensitive to resuspended sediment (natural and anthropogenic), as it can cause smothering of coral polyps and tissue necrosis (Erftemeijer et al., 2012), changes in food intake, adult behavior, polyp loss (Liefmann et al., 2018), coral larval mortality and behavior changes (Järnegren et al., 2017; Järnegren et al., 2020), and physiological arrests (e.g., Grant et al. 2019). Liefmann et al. (2018) showed that sharp particles produced during seabed mining can be particularly deleterious in cold-water soft corals exposed to mine tailing particles.

Considering that commercial bottom trawling is not allowed inside any of the coral and sponge OECMs, this indicator would be mostly examined to evaluate potential impacts of **outside** trawling activities on **inside** corals and sponges. However, if DFO scientific trawl surveys continue inside of OECMs, the potential impacts of sediment disturbance from these surveys should be evaluated. Grant et al. (2019) suggested that sediment plumes from bottom trawling activities >2 km outside of a protected area could increase the amount of suspended sediment within the protected area to a level that would cause glass sponges to stop filtering water.

Furthermore, sediment associated with exploratory drilling (e.g., drill cuttings) can be deposited within 1 km from the site of exploration (Roberts et al., 2006). Measuring the deposition of anthropogenic-related sediments inside of OECMs could be achieved through the deployment of sediment traps (i.e., associated with moorings or benthic landers) inside of those areas. Traps should stay underwater for a period that varies between months and one to a few years (longer if there are issues with equipment recovery). They would provide information on sediment deposition, rate of deposition, and composition. The distribution of sediments produced as a result of oil and gas activities and potentially transported to OECMs could be traced through the targeted sampling of sediment from near corals and sponges, and analyzed for biomarkers (e.g., fatty acids, discussed below). Targeted sediment sampling be more accurate using ROV push-cores (discussed in section 5). Although sediment traps can be an excellent tool for research, their utility as part of a coral and sponge monitoring program has not been determined yet for Canadian OECMs. Sediment profile imagery (SPI) could also likely be explored as a tool to assess "new" sediment accumulation on the seafloor.

Sediment dispersion/transport models can be generated to assess the potential spread of sediment particles in an area (e.g., Grant et al., 2019). Dispersion models are used by industry (e.g., oil and gas, aquaculture) to assess the spread of sediment and other particles resulting from their activities (e.g., Crawford et al., 2002; Cromey et al., 2002). Although having baseline data on undisturbed conditions is crucial to understand the impacts of disturbance, disturbance simulations could be performed to produce acceptable baseline data. For instance, Grant et al. (2019) used the ROV ROPOS to carry out sediment resuspension experiments where the ROV was used to generate sediment plumes and assess their impact on glass sponge physiological activities, while also assessing these activities under pre-disturbed scenarios. However, in some locations (e.g., Newfoundland shelf), sediment dispersion (transport) models can be difficult to generate due to complex oceanography and logistics associated with obtaining bottom current data over meaningful spatial and temporal scales.

Sediment plumes produced during bottom trawling can be several meters high and dozens to hundreds of meters wide (e.g., Durrieu De Madron et al. 2005). Smaller bottom contact gear can also generate sediment plumes, but with a much smaller footprint (e.g., 1 m high and 1.5 m wide for a benthic sled, O'Neill and Summerbell 2011). The characteristics of sediment plumes generated during DFO trawl surveys (and other comparable scientific surveys) are currently unknown. Therefore, if these surveys continue inside of closed areas, then potential effects of **outside** commercial and scientific trawling versus **inside** scientific trawling could be confounded. However, considering that commercial trawling has a much larger footprint than scientific surveys (e.g., Rideout et al. 2024), it is possible that changes in the characteristics of sediment deposition inside of OECMs correspond to a direct change in fishing pressure (high sensitivity). Regional studies will be crucial to investigate the dispersion potential of sediment plumes generated during anthropogenic activities occurring inside and outside of OECMs, and their potential to impact the corals and sponges found therein.

It is still difficult to assess the responsiveness of this indicator, considering our limitations in baseline data availability. However, assuming baseline data is available, detectable changes in sediment deposition inside of OECMs should be large enough to indicate a response to anthropogenic pressures rather than natural changes, unless specific consistent animal behavior or geological activities are known to affect sediment dynamics in a given area. This also highlights the importance of high-resolution seabed mapping and development of substrate maps for these areas (discussed in section 5). In terms of specificity, a distinction between sediments generated from trawling and oil and gas activities could likely be obtained based on sediment analysis (i.e., composition, grain size, chemistry), given their different nature (e.g., drilling muds are associated with specific chemicals). Data acquisition for this indicator would likely be associated with the collection of oceanographic data and benthic surveys (e.g., ROV, grab sampling).

4.4.5. Chemical impacts related to oil and gas activities

Concentrations of drilling chemicals have been detected in surface sediments up to 4 km away from drilling sites (Lepland and Mortensen, 2008). These chemical contaminants can have toxic effects on corals and sponges, which can be studied using biomarkers (Müller et al., 2000; Kenchington et al., 2012). The high concentration of hydrocarbons on a soft coral sample collected from a natural cold seep in Baffin Bay indicated that lipid analyses can be used to identify the presence of hydrocarbons (Cramm et al., 2021). There is some baseline data on lipid and fatty acid composition of cold-water corals in the Flemish Cap, Northwest Atlantic (Salvo et al., 2018). Sponge fatty acid composition has been described by Bergquist et al. (1984), and more recent specific studies are also available (e.g., Schreiber et al. 2006; Blumenberg and Michaelis 2007; Parzanini et al. 2018).

Detecting the presence of drill muds and cuts and other wastes from oil and gas exploration activities inside and outside of OECMs could potentially be achieved through the targeted collection of sediment cores. Undisturbed sediment would provide better data in terms of deposition of such materials. In this case, ROVs with push-coring capabilities are the tool of choice (as described above for the sediment deposition indicator).

This indicator can be both specific and sensitive, as detecting the presence of chemicals related to oil and gas activities can be a direct response to the presence of such chemicals. Responsiveness will depend on the availability of baseline data, the dispersal power of such chemicals, and the frequency of impact and monitoring, as levels might change from the moment of the impact to the time when sampling takes place.

4.4.6. Timing and duration of anomalous events

Anomalous climatic events related to ocean-atmosphere interactions (Josey et al., 2018) or sporadic earthquakes that can induce mass movements and gravity flows (Tripsanas et al. 2008), and can have an effect in the distribution, abundance, and biomass of cold-water corals and sponges, as a result of their physiological limits (Kenchington et al., 2012). However, these events are not well known and information on this indicator may be scarce or nonexistent for most OECMs considered here. Furthermore, most of the Canadian OECMs are not located in seismically active zones. Documenting the effects of anomalous events should be part of monitoring protocols, but drawing meaningful patterns based on sporadic data might not be possible.

4.4.7. Timing, duration and magnitude of phytoplankton blooms

This was suggested as a stressor indicator by Kenchington et al. (2012), but it could also be included in the environmental indicators. Phytoplankton blooms are thought to be a controlling factor in the reproductive cycles of deep-sea corals (Sun et al., 2010a; Sun et al, 2010b) and sponges (Spetland et al., 2007) and therefore may influence recruitment success and productivity. This indicator would be based on surface chlorophyll *a* concentration measured from satellite data. In regions where significant and widespread sub-surface phytoplankton blooms exist, such as in the Arctic, measurement of this indicator should also include vertical profiles of chlorophyll *a* concentration (fluorescence-based techniques). Sensitivity, specificity, and responsiveness of this indicator are being assessed here as low, as changes in timing, duration, and magnitude of phytoplankton blooms are not expected to directly correspond to changes in fishing.

However, measurements of this indicator can, in some regions, greatly contribute to discriminating the effects of ceased demersal fishing pressures from climate-change related changes. For example, the Canadian Arctic is undergoing profound changes in sea ice conditions affecting the timing of phytoplankton bloom, and in water circulation pathways affecting nutrient availability and distribution and, thus, the magnitude of the phytoplankton bloom. Impacts of these changes on corals and sponges will be overlaid on changes caused by protection measures. If this indicator is not being monitored, effectiveness of closures cannot be properly assessed in regions where significant environmental changes at the primary producer level are occurring, which are translated to coral and sponge productivity and diversity via benthic-pelagic coupling mechanisms.

4.4.8. Timing and duration of sea ice cover

The seasonal cycle of sea ice is an important driver of the marine environment productivity in Arctic communities (Clark et al., 2015). Sea ice is a major control of the phytoplankton bloom onset through light availability, but it also acts as habitat for ice algae which, in some Arctic settings may contribute a significant fraction to the overall ecosystem primary productivity at the beginning of the productive season. This indicator could be based on ice thickness and distribution data, which can be obtained from the Canadian Ice Service (CIS). Like the phytoplankton bloom indicator, sensitivity and specificity of this indicator in terms of bottom fishing pressure are low, as changes in the timing and duration of sea ice cover are not expected to directly correspond to changes (removal) in this pressure. However, increased fishing in expected in the Canadian Arctic as a result of decreasing ice extent (Tai et al. 2019), which can have an influence in Canadian OECMs influenced by ice (e.g., Disko Fan Conservation Area). Responsiveness of this indicator can be considered low, due to natural inter-annual and spatial variations in ice dynamics.

Climate change, on the other hand, has a direct impact on sea ice dynamics and thermodynamics. Both sensitivity, specificity, and responsiveness of this indicator to climate change can be considered high. Changes in ice dynamics and thermodynamics can have a direct impact on benthic processes, as a response to changes in pelagic and ice-associated productivity and benthic-pelagic coupling (Wassmann and Reigstad, 2011). Therefore, monitoring this indicator is crucial to understand potential changes in benthic diversity (e.g., abundance, biomass, growth rates) and hence coral and sponge dynamics in Canadian OECMs influenced by ice (e.g., Disko Fan Conservation Area).

4.4.9. Seabed litter presence

This indicator would be represented by any seabed litter, but particularly the presence of ghost fishing gear in coral and sponge OECMs. Seabed littler has been assessed from trawl samples in the Flemish Cap (García-Alegre et al., 2020). However, in the case of closed areas, assessing seabed litter would be performed based on imagery data, likely as part of other assessments using imagery data (e.g., faunal assessments). Seabed litter might provide an indication of non-compliance depending on the state of the gear on the seafloor, but not necessarily as it could also drift into these areas.

In most cases, sensitivity and specificity of this indicator is high, as the presence of marine debris is directly associated with anthropogenic activities. Particularly, the presence of ghost fishing gear is a direct response from fishing activities. While the identification of gear of known age can inform compliance, in most cases knowing when the gear fell on the seafloor might not be possible. Responsiveness of this indicator might be low. While the number of bottom trawlers fishing in the OECMs is expected to be null (assuming compliance), vessel traffic is still permitted, and litter associated with human presence (e.g., cans, plastic bottles) is still expected in these areas. Furthermore, the presence of microplastics is also expected in these areas and their source (e.g., trawlers or other vessels) would be challenging if not impossible to determine.

4.5. ACTIVITIES RELATED TO SUBMARINE CABLES

This indicator would be represented by activities related to the installation, maintenance, and/or decommissioning of underwater cables (e.g., power transmission). Underwater cables are widely distributed around the globe, and they have the potential to impact benthic communities, including cold-water corals and sponges (Taormina et al., 2018). In a study on the impacts of a submarine power transmission cable installed in a glass sponge reef area, Dunham et al. (2015) identified a 100% mortality for sponges under the cable footprint, as well as a lower cover of live sponge along cable transects in relation to control sites. Underwater cables can be found in several coral and sponge OECMs, and their presence and potential impacts on these communities needs to be acknowledged and investigated. Measurement of the presence/extent of submarine cables in OECMs might be straightforward to be performed using GIS techniques, if data are made available.

4.6. OTHER ACTIVITIES

4.6.1. Offshore wind energy

As for oil and gas and mining activities, offshore wind farming is not explicitly prohibited in Canadian OECMs and will be evaluated in a risk assessment for each potential site. While as of December 2020 no offshore wind farms exist in Canada, several projects have been proposed, including the NaiKun Project in Hecate Strait, B.C. and five projects for Atlantic Canada: two off the coast of Newfoundland and Labrador, and one each off Nova Scotia, Prince Edward Island, and New Brunswick (CER, 2017). In Newfoundland there is interest on the possibility of powering offshore oil platforms through wind energy, similar to recently developed in Norway.

Like the other activities described in this section, developments of offshore wind energy projects also have the potential to generate impacts on benthic communities during different phases of the activity (e.g., Schröder et al. 2006, Wilson et al. 2010). Offshore wind energy farms are composed of wind turbines and their associated infrastructure, which includes substations and subsea cables (Wilson et al. 2010). To date, there are no studies on the potential impacts of offshore wind activities specific to corals and sponges, since this is a relatively new industry. However, some of the steps in the installation of offshore wind turbines and their associated infrastructure can be compared to those seen in the oil and gas industry, such as the anchoring of a large structure on the seafloor (i.e., the turbine) and in some cases the drilling of bedrock areas with the generation of drill bits, although the volume of drill cuttings disposed from the turbines are likely to be much smaller than those generated during oil and gas drilling activities (Wilson et al. 2010). While it is not our objective to discuss in detail the potential impacts of offshore wind farms on the benthos, these can include change in sediment structure and flow patterns, increased turbulence (which modifies the substrate), changes in species abundance, richness, and diversity (Wilson et al. 2010).

Since offshore wind energy is a new industry sector, there are no historical data available for Canadian waters. The creation of farms inside or near OECMs should be tracked as part of the OEMC's monitoring plan. Their spatial extent in terms of abundance of turbines and distance to features of interest (e.g., OECM, VME, SiBA, or specific coral and sponge communities) should be considered over time, which would be relatively straightforward to perform if the specific locations of such farms are available. Sensitivity, responsiveness, and specificity do not need to be assessed against this indicator. The distribution of offshore wind activity sites will depend on the objectives of the farms (e.g., if associated with oil platforms or not). Since turbines might serve as artificial substrate for the settling of sessile organisms including corals and sponges, having access to data on fauna colonization on these turbines over time would be of interest.

4.6.2. Climate change

Global climate change caused by anthropogenic greenhouse gas emissions is bringing significant changes in the physical and chemical properties of the oceans, which have profound implications for marine ecosystems (e.g., Harley et al., 2006; Pörtner and Farrell, 2008). Climate-change induced stressors include ocean warming, ocean acidification, and deoxygenation.

Warming ocean temperature can negatively impact the performance and survival of marine organisms (McWilliams et al. 2005); can cause shifts in geographic distribution (e.g., Perry et al., 2005; Campana et al. 2020); and can produce changes in the seasonal timing of biological events (e.g., Edwards and Richardson, 2004; Philippart et al. 2003). Rising temperatures can also change food availability and food web structure causing changes at the community level (e.g., Schiel et al. 2004; Schultz and Cloutier, 2016). Increase in surface temperature is also predicted to enhance stratification of the water column, which will increase the degree of decoupling between the surface and deeper waters (Capotondi et al. 2012) and can cause a shift in phytoplankton assemblages (Bopp et al., 2005). This shift is likely to reduce carbon flux to the seafloor, as well as decrease efficiency of the biological pump (Steinacher et al., 2010; Moran et al. 2015).

Warming also affects dissolved oxygen in the ocean. Direct observations suggest that warming is accelerating ocean deoxygenation and the global ocean oxygen inventory is decreasing (Oschlies et al. 2018), whereas the oxygen minimum zone is expanding in some areas (Ross et al. 2020). Warming affects ocean oxygen inventory through two mechanisms: directly, via solubility effects (the warmer the water, the less gas that can dissolve in it); and indirectly, via changes in global ocean circulation, mixing and oxygen respiration (Oschlies et al. 2018). Oxygen depletion will diminish the potential of colonization of the habitat due to avoidance by larvae of organisms with sessile or sedentary adult phase (Lagos et al., 2015), and can decrease regional diversity and cause local extinctions (Ross et al. 2020).

The increase of ocean acidity has profound implications for physiological process in marine organisms, especially in marine invertebrates that build carbonate structures such as corals. Decreased calcification rates in response to ocean acidification have been shown in reef-building scleractinian corals (Kleypas et al. 1999; Gómez et al. 2019) and in gorgonians (Cerrano et al. 2013). Additionally, ocean acidification has been shown to have substantial adverse effects on the pumping capacity, tissue withdrawal, and structural integrity of the glass sponge *Aphrocallistes vastus* (Stevenson et al. 2020). The decrease in skeletal structures needed for their support and protection may result in structural collapse and increased mortality (Cerrano et al. 2013; Büscher et al., 2019; Stevenson et al. 2020).

Recent projections suggest that by the year 2100 abyssal depths (3000–6000 m) could experience temperature increases in excess of 1°C, whereas bathyal depths (200–3000 m) worldwide will be exposed to significant reductions in pH (0.29 to 0.37 pH units) and O_2 concentrations decreases up to 3.7% or more in some oceans (Sweetman et al., 2017). Recent habitat suitability models under different climate projections suggested that the reef coral *Lophelia pertusa* and the large gorgonian *Paragorgia arborea* would suffer a reduction on habitat in the northwest Atlantic (Morato et al. 2020).While all indicators described in this section can be selected for the monitoring of Canadian OECMs, climate change indicators will also need to be incorporated in monitoring plans. The Government of Canada developed the “Federal Adaptation Policy Framework for climate change” to guide priorities and action

(Government of Canada 2011). Currently there is no national framework or guidance on incorporating climate change in Canadian MPAs and their networks or OECMs. However, DFO has been working on initiatives for the creation and development of tools to allow accommodating climate change in marine research, including the monitoring of protected areas.

4.7. CONCLUSION

We presented here a number of indicators that can be used to monitor coral and sponge status in Canadian OECMs. The conservation objectives of each area and the availability and appropriateness of data will need to be considered when choosing indicators to include in a monitoring plan. In addition, as our knowledge of cold-water corals and sponges increases with technological advances and increased monitoring, indicator selection will need to be reviewed for completeness. This is especially true in areas where little research has occurred previously, and in frontier areas. Indicators are not static; their selection and analysis will need to be iterative and responsive to advancements in knowledge.

5. TOOLS, TECHNIQUES, AND METHODOLOGIES

5.1. INTRODUCTION

The identification of appropriate tools, techniques, and/or methodologies for monitoring is essential for assessing the effectiveness of conservation areas and needs to be directly linked to the conservation objectives (Figure 9). The choice of tools will influence the resolution of data. For instance, with imagery tools, detectability of certain taxa might differ greatly between high-resolution still images and standard definition video (Althaus et al., 2015; Dunham et al., 2018a). Also, the use of different gear can yield significant differences in terms of species richness, densities or species composition (Sheehan et al., 2014). Therefore, careful consideration and trials, when possible, should be undertaken to make sure that the selected tools and the data they provide are well-aligned with conservation objectives and monitoring indicators.

Some of the challenges associated with the monitoring of the 40 coral and sponge Canadian OECMs include that they are mostly offshore and in deep water (i.e., > 200 m and up to 4,700 m; Table 1). These factors impose logistical constraints and require the use of specific tools and platforms (i.e., gear, vessels) (Table 6). Furthermore, the large size of several of these OECMs (e.g., half of them is > 800 km^2 and seven of them are > 7,000 km^2, and up to 55,350 km^2), represents a challenge in terms of baseline and ongoing data collection as well as implementation of appropriate monitoring measures. These challenges and limitations will need to be considered during the development of monitoring designs, which will need to follow best practices to ensure that enough and the right type of data is collected to allow for meaningful statistical analyses. Robust monitoring programs are essential for optimizing available resources and allowing for the detection of change in the benthic environment, as well as to inform which management measures have been successful (Foster et al., 2018; Noble-James et al., 2018).

5.2. TOOLS AND TECHNIQUES

In this section we describe potential tools and techniques currently used in the survey of benthic organisms, with a focus on imagery technologies and bottom contact gear. We also briefly discuss the use of acoustic techniques for habitat mapping and environmental DNA (eDNA) in monitoring, offer a comparison of tools and methods, and suggest best practices. It should be emphasized that depth and bottom type are important variables to be considered when planning benthic surveys and selecting suitable tools. Potential monitoring indicators have been listed in the previous section, and we will focus this section on tools to monitor those indicators.

5.2.1. Imagery technologies

Imagery technologies have several advantages for the monitoring of benthic taxa. In the early 2000s, Gordon Jr. et al. (2000) discussed the application of imagery tools developed by Canadian scientists and engineers for the study of the marine benthos: towed camera; Campod; and videograb. Since then, the range of tools and environments surveyed in Canadian waters has expanded. In this section we describe some of the most current imagery technologies used for the study of the marine benthic environment, with the potential to be used for monitoring purposes in Canadian coral and/or sponge OECMs. We also provide a comparison of tools, with limitations and benefits of each technology, and discuss challenges associated with imagery data collection, storage, processing, and analysis.

This document is not meant to provide a full review of these tools or to provide specific protocols, but rather to list and briefly describe them as potential tools to be used in the monitoring of Canadian OECMs. We focus on tools recently used in Canadian surveys and factors to consider in the usability of each. Standards and protocols for the use of imagery technologies in benthic research have been developed by several authors and should be considered on a case-by-case basis (e.g., Coggan et al. 2007; Larocque and Thorne 2012; Jamieson et al. 2013; Mallet and Pelletier 2014; Brandt et al. 2016; Moore et al., 2019).

5.2.1.1. Remotely Operated Vehicles (ROVs)

ROVs are tethered unmanned underwater vehicles that can be equipped with sensors, cameras, oceanographic instruments (e.g., CTD, Niskin bottles, multibeam echosounder) and other equipment (e.g., manipulators, sampling boxes, sediment push-cores). ROVs navigate at a certain distance from the seafloor and provide a live-feed view to pilots and scientists aboard the vessel. They are therefore considered more suitable for the investigation of areas hosting vulnerable benthic fauna such as corals and sponges, particularly in comparison to invasive bottom-contact gear such as bottom trawls. ROVs are also a useful tool to survey rough terrain areas (boulder fields, cliffs, etc.) where trawls and other conventional benthic gear cannot be safely and efficiently operated. They are one of the least invasive ways to conduct biological surveys of deep-water habitats, which is particularly appropriate when surveying conservation areas, many of which support sensitive and/or vulnerable species.

ROVs have successfully collected imagery data that have been used to investigate quantitative monitoring indicators (Table 5, Table 6), including benthic taxa abundance, distribution, and size (Chimienti et al., 2018b; Dinn et al., 2020a). In fact, ROV imagery data have been shown to yield more accurate abundance data for benthic taxa in comparison to bottom trawls (e.g., Ayma et al. 2016; Chimienti et al. 2018b). In some cases, these vehicles are the most appropriate tool to survey an area. For instance, the monitoring of glass sponge reefs and hydrothermal vents might require the collection of both imagery data and associated physical samples in a very targeted, precise way. ROVs equipped with cameras positioned at different angles (e.g., forward and downward-looking) can provide imagery data at different spatial scales. Downward-looking cameras might better allow the detection of juveniles (e.g., sea pens) as they provide a focused and more detailed view of the seafloor. Forward-looking cameras allow a better estimation of population density, as they provide a wider field of view. Having different cameras pointing at different angles is not an ROV exclusivity, being possible in other types of imagery gear as well (e.g., drop cameras and towed video systems discussed later in this section).

Each ROV has a different set of associated tools and capabilities, and not all ROVs are suitable for all kinds of surveys. For instance, some do not have sampling capabilities or even high-resolution cameras. Operating depths can also vary. The type of camera system and lighting set-up will directly influence the quality of imagery, which will also be directly affected by ROV speed and distance from the seafloor. In most cases, ROV speed should not exceed 0.5 knots in transect mode, and the vehicle should navigate at <2 m from the seafloor (camera position), as both variables directly influence imagery quality (e.g., for species identifications). The issue of altitude is further discussed in the section on towed underwater vehicles, where this variable might be more difficult to control. ROV altitude can be controlled if the vehicle has an altimeter or another system capable of measuring distance from the seafloor (e.g., Doppler Velocity Log, DVL). However, pilot experience is paramount to maintain an ROV at a relatively constant distance from the seafloor during a transect because high current can have a significant effect on ROV behavior. If the ROV does not have an instrument to allow measuring altitude, a pair of laser points - frequently present in ROVs - can be used as a proxy. Variation in pixel distance between laser points can be used as a measure of variation in vehicle altitude. Having a pair (or more) of laser points available in the imagery is one of the most basic requirements associated with current imagery technologies. Lasers have the main objective of providing a scale for size estimates. Not all ROVs have associated lasers, and alternative methods might be necessary to obtain object size estimates (e.g., Neves et al. 2015b used a metal ruler). Therefore, having clear survey objectives will guide the choice of ROV depending on its capabilities (technical and operational) and history (previous users' experience), as well as the availability of capable ships.

In Canada, ROPOS (Remotely Operated Platform for Ocean Sciences) is currently the leading science-oriented ROV, which is internationally recognized for producing high quality imagery data and for sampling objects, sediment, and water precisely (e.g., Baker et al., 2012; Bennecke & Metaxas, 2017; Campanyà-Llovet et al., 2018), and employing highly qualified personnel. ROPOS is managed and operated by the Canadian Scientific Submersible Facility (CSSF). A board of directors consisting of nine Canadian marine scientists and business leaders, and a user committee made up of Canadian and international scientists provide CSSF guidance (ROPOS 2020). Some Canadian institutions and industries have their own ROVs and associated technical personnel, but most of these ROVs have limited capabilities in comparison to ROPOS. Amundsen Science (Laval University) has recently acquired a new science oriented ROV to replace their old SuMo ROV. The new ROV (ASTRID) has HD cameras and sampling capabilities (7-function arm and sampling storage boxes), including the ability to sample sediment push-cores and seawater through attached Niskin bottles.

Science ROVs are usually large enough that they need to be deployed from a vessel with access to a capable crane. ROPOS has its own launch and recovery system (i.e., LARS), which greatly facilitates its deployment, operation, heave compensation, and recovery. Where a system such as LARS is not available, ROVs can be deployed directly from a crane, but the process can be less straightforward. Where a vessel dynamic positioning (DP) system is not available, the vessel crew might encounter challenges in station-keeping, which can influence ROV survey efficiency and data quality. In Canada, ROPOS has been deployed from Canadian Coast Guard (CCG) ships multiple times including: CCGS *Tully*, *Vector*, *Martha L. Black*, *Hudson*, some of which do not have DP; and yet, operations have been successful. Amundsen Science's SuMo and the ASTRID ROVs have only been deployed from CCGS *Amundsen*, but there are plans to allow deployment of ASTRID from other vessels (A. Forest, personal communication). ROV deployment and recovery can be time consuming, therefore a survey planning that maximizes time on bottom should be considered to optimize data collection (e.g., favor fewer but longer transects to several short transects).

A recent trend in the ROV realm is the use of mini ROVs, which are small, light, and can be hand-deployed and operated using a simple tablet (e.g., Buscher et al. 2020; Raoult et al.

2020). These machines are particularly useful for seafloor inspection, and they have been successfully used in the collection of video transect data (e.g., Buscher et al. 2020; Raoult et al. 2020). However, they are usually limited to shallow-water operations (e.g., <100 m) (e.g., Buscher et al., 2020) and require some user experience to properly deploy. Considering that most of the Canadian coral and sponge OECMs are found in waters >200 m, the use of these ROVs would be limited for the needs described here. Yet, they are affordable (Table 6) and might be particularly useful for the collection of baseline data and/or in the exploration of frontier areas in the lack of more capable equipment.

Science ROVs are complex tools, the logistics associated with their operation are not negligible. As such, they can be considerably more costly than more conventional benthic gear (e.g., trawls, drop cameras, Table 6). However, in many cases they certainly fit conservation purposes better due to their low-invasive nature. It is important to emphasize that when attempting to reduce costs, there might be a trade-off between data quality certainty and uncertainty (i.e., reliable versus unreliable or less tested gear). On the other hand, certain metrics can be efficiently collected using alternative imagery gear (described in the next subsections). Despite the elevated cost and need for advanced planning, ROV expeditions are usually of interest to a range of scientists (e.g., government, academia, and NGOs) who are often willing to collaborate and share associated costs. Engagement across multiple research partnerships and interests is therefore crucial. Although availability of resources clearly needs to be considered, the use of efficient and adequate equipment is paramount for the collection of suitable data.

Key points to consider:

- The selection of ROVs and their tools/capabilities depends on the different types of work and habitats;
- The quality of imagery and resulting data can vary greatly depending on ROV and survey design, and need to be linked to survey objectives and data requirements for monitoring;
- ROVs are versatile and can access locations where other traditional benthic gear cannot;
- ROVs can be the most efficient tool to collect both imagery and physical samples from targeted locations; and
- Despite advantages, the elevated costs of using ROVs likely limit their frequent use in monitoring, except where collection of data on selected indicators requires or favors their use (e.g., glass sponge reefs, *Lophelia* reefs).

5.2.1.2. Human-occupied submersibles and underwater cabled observatories

Human-occupied submersibles are untethered underwater vehicles that allow the presence of a pilot(s) and a scientist(s) aboard (Smith and Rumohr, 2013). There are several submersibles around the world (Smith and Rumohr, 2013), with a few based in Canada (e.g., Aquarius, Deep Rover, and Curasub, Nuytco Research - British Columbia). The Aquarius submersible has HD cameras and a seven-function manipulator, but it is limited to a depth of 300 m (Nuytco, 2020). The submersible Pisces IV (originally Canadian) was deployed around Newfoundland and Baffin Bay (e.g., Syvitski et al. 1983; Grant et al. 1986; Hughes-Clarke et al. 1989; Haedrich and Gagnon 1991) and British Columbia (Richards, 1986) for the study of the marine benthos and seafloor geology in the 1980s. Submersibles can be similar to ROVs in their sampling capabilities and navigation precision, but they are unlikely to be consistently used in the monitoring of Canadian OECMs (Table 6) and generally have limited bottom time (e.g., 4 hours, but some can stay underwater longer - Smith and Rumohr 2013).

Underwater cabled observatories are another class of advanced imagery technology. There are several underwater observatories in Canadian waters, two of the most prominent being the deep-water VENUS and NEPTUNE (both in the West coast), managed by Oceans Network Canada (ONC). These types of observatories allow for long-term oceanographic and imagery data collection, and observations of faunal behavior and seasonal patterns (Aguzzi et al., 2015; Juniper et al., 2019). The NEPTUNE observatory is located within the deep-water MPA Endeavor Hydrothermal Vents. The observatory has fixed subsea sensors (e.g., temperature) and is frequently (i.e., several expeditions per year) surveyed by ROVs for the maintenance of the platform, allowing the simultaneous collection of scientific data (Juniper et al., 2019).

Key points to consider:

- Similarly to ROVs, submersibles are great tools but with high associated costs for monitoring purposes. Therefore, it is unlikely that submersibles would be selected as a frequent monitoring tool for Canadian OECMs;
- Submersibles can still be useful for the collection of baseline data and for research surveys; and
- Wherever available, underwater cabled observatories can provide an opportunity for data collection at high temporal resolutions.

5.2.1.3. Automated underwater vehicles (AUVs)

AUVs are untethered underwater vehicles with autonomous navigation capabilities (Smith and Rumohr, 2013). These vehicles follow a pre-programed track and can be equipped with cameras, oceanographic instruments (e.g., CTD) and sonar systems (Huvenne et al., 2018). Like ROVs and human-occupied submersibles, AUVs have the advantage of being non-invasive tools, as they usually do not have any contact with the seafloor. Most AUVs can be programmed to navigate at a few meters from the seafloor to avoid potential collisions with objects. Unlike ROVs, AUVs generally do not allow for a live feed of the survey and do not have sampling capabilities, although Nishida et al. (2019) described an AUV equipped with a suction sampler. If mounted with high-resolution cameras and good lighting, they can also be used to collect high quality downward-looking imagery data that can be used to determine taxa diversity, abundance, distribution, seabed classification and have been used for monitoring purposes (Williams et al., 2012; Du Preez et al., 2015; Ferrari et al., 2018; Meyer et al., 2019). AUVs are also a very powerful tool in seafloor mapping (described in subsection on acoustic methods).

As with ROVs, AUV speed for the collection of imagery during benthic surveys should be slow enough to avoid compromising image quality (i.e., <0.5 knots). AUVs can usually be left in the water longer than ROVs, and hence cover more ground (Smith and Rumohr, 2013; Huvenne et al., 2018). Although in theory ROVs can also remain underwater for days in a row (e.g., ROPOS record is >99 hours, P. Lockhart – CSSF, personal communication), in practice this rarely occurs, as other vessel operations (i.e., other gear deployment) are usually paused while an ROV dive is ongoing. AUVs, on the other hand, can be left operating underwater while other activities can take place.

AUV altitude during benthic surveys can be variable. Surveys have been conducted anywhere from altitudes of 2 m (Ferrari et al., 2018) to over 5 m above the seafloor (Meyer et al., 2019). However, AUV altitude can be problematic when exceeding 2 m, as object detection and taxa identification might be compromised, especially if not associated with physical sampling, or at sites where fauna diversity is not well known. Although surveys at higher altitudes can still be performed successfully (e.g., Meyer et al., 2019), high altitudes are certainly a limitation of certain AUV surveys. But most importantly, AUV altitude (and for all other imagery gear) across monitoring surveys needs to be comparable. Some AUVs have been designed for near-bottom surveys (e.g., ABE - WHOI, United States, and IAUV - MBARI, United States, IMOS AUV, Australia) (Smith and Rumohr, 2013).

Key points to consider:

- AUVs can collect high quality imagery data, but generally cannot collect samples;
- Live feed is generally not available; and
- They can cover large areas and be deployed while other operations take place.

5.2.1.4. Drop camera systems

These systems are generally composed of a camera(s) and lights mounted to a frame, which is tethered. Drop cameras can be used in a wide range of depth and type of bottom and are an especially good tool in more rocky areas. Cameras can be positioned at different angles on the frame, providing imagery data in different spatial resolutions (Larocque and Thorne, 2012; Beazley et al., 2015; L. Buhl-Mortensen et al., 2017a). As with other imagery technologies, drop cameras can be used to assess several monitoring indicators (Table 5, Table 6, Table 7). Some drop camera systems, such as Campod (Gordon Jr. et al. 2000), can yield a real-time view of the seafloor during deployment and therefore act both as drop camera and towed underwater video systems (TUVS discussed in next section). Campod has an oblique video camera that provides general reconnaissance information, whereas detailed imagery can be obtained from the downward looking high resolution video camera and still photographs. Campod has proven to be an excellent tool for obtaining high-resolution video and photographic imagery of benthic habitat and epibenthic organisms and has been recently deployed to ~1000 m depth in the Maritimes Region (Beazley et al. 2019). When cameras are deployed “blindly”, with no real-time view, imagery collection might be based on a “yo-yo” system, in which the camera is lowered down until it has touched the seafloor. The camera is then hauled back as the ship moves forward, until it is lowered down again, and the process is repeated for a pre-set distance or duration. In these cases, video might be continuously recorded. In other cases, the system might have a mechanism that allows the camera to be triggered only once it touches the seafloor, when a still image will be taken, but with no continuous video being collected between “drops”.

Imagery quality based on blind camera deployments will depend on several factors, including water depth, winch operator experience, and bottom type. Water depth can have a significant impact on the winch operators’ ability to identify changes in the cable’s tension; but this is further complicated at depths over 800 m (Y. Gagnon, CCGS boatswain, personal communication). If a positioning device (e.g., acoustic beacon) is attached to a “blind” system, better precision for timing when the camera touches the seafloor can be achieved. Sea state (e.g., heave and swell) are also important factors in image quality. Blind gear deployment can be problematic in soft bottom areas, where gear contact with the seafloor might create a cloud of suspended sediment that can substantially impair visibility. While “approaching the seafloor” imagery might still be usable, sediment clouds might extend between drop locations. As it is not possible to assess video quality during these blind deployments, real-time adjustments of the technique are not possible, and the resulting imagery might not be suitable. It is therefore important to thoroughly test the gear in different settings and to establish deployment protocols. For example, waiting between 2-3 minutes before relocating to the next point might be necessary. Although uncommon, blind deployments might also impact the benthic fauna if the gear is dragged rather than lifted between drop locations, which could happen under high seas conditions.

Drop cameras can have high position accuracy if deployed with an acoustic beacon attached to the gear or cable. If such a positioning system is not available, positional accuracy might be limited to the ship's position rather than gear location on the seafloor. For monitoring purposes, being able to revisit sites with a high level of positioning precision is important. Camera use will differ region to region and can be explored in more detail depending on location and equipment.

Key points to consider:

- Drop camera systems can be small and affordable (i.e., in comparison to ROVs and AUVs), facilitating their deployment from small-size vessels;
- Since most drop cameras touch the seafloor upon deployment and have a downward-looking camera, imagery are always taken at the same distance from the seafloor (i.e., covering the same field of view), which facilitates collection of precise quantitative data;
- Drop cameras with no imagery live feed ("blind cameras") can be more challenging to operate, but once a good protocol is developed, they can be very efficient tools;
- These systems can be deployed in diverse habitats (e.g., soft, hard), although still limited in steep areas or boulder-dense areas.

5.2.1.5. Towed underwater video systems (TUVS)

TUVS are a group of towed underwater systems that usually have continuous contact with the seafloor and provide video recording along a transect (Larocque and Thorne, 2012; Smith and Rumohr, 2013; Sheehan et al., 2016; Foveau et al., 2017; Prado et al., 2020). As for drop camera systems, TUVS can be used to determine fauna abundance, diversity, and distribution (Shortis et al., 2007; Fields et al., 2019). These systems include benthic sleds (also called sledges) and are generally composed of cameras and lights attached to a metal frame that is towed over the seafloor. Some benthic sleds also have sampling capabilities (e.g., KC Denmark A/S). They mostly differ from drop cameras in that they provide an uninterrupted view of the seafloor over a distance, rather than a snapshot in one location. Benthic sleds can be efficient and low-cost tools for benthic surveys (Rooper, 2008; Long et al., 2020). The "Aberdeen-type" sled is an example of TUVS routinely used around Europe (Shand and Priestley, 1999; Smith and Rumohr, 2013). One disadvantage of TUVS is that while they might work well in flat bottom environments, their use is limited or not possible in areas of hard bottom or complex/steep terrain.

Sheehan et al. (2010, 2016) have also proposed the use of alternative "benthic-tending" TUVS, which are suspended versions of TUVS that only contact the seafloor via a small ground chain. SubC technologies (Clarenville, Newfoundland) is currently working on the development of a benthic-tending sled based on Sheehan et al. (2016)'s model and expected to be deployed in relatively shallow-water environments (i.e., < 250 m) (Chad Collet, SubC technologies, personal communication). Benthic-tending TUVS have the advantage of being less limited by bottom type (e.g., could be deployed in relatively rocky areas) and have less impact on the seafloor than regular TUVS. Some TUVS can hover above the seafloor. This type might navigate at variable altitudes and should have scaling lasers to determine the distance from the seafloor (discussed in ROV section). The benthic-tending TUVS described by Sheehan et al. (2016) were tested in shallow-water, and the possibility of deploying these types of TUVS in deep-water settings should be explored. Both papers by Sheehan et al. (2010, 2016) offer excellent descriptions of the gear and method.

TUVS can be deployed in real-time, which allows necessary adjustments in direction and obstacle avoidance, and possibly better speed control. If deployed blindly, TUVS hovering above the seafloor might yield potentially challenging imagery data if distance from the seafloor cannot be controlled. A list of some well-established TUVS used in institutions across the world is provided in Jamieson et al. (2013).

Key points to consider:

- TUVS allow for the collection of continuous video during transects, and downward-looking photos;
- TUVS towed directly in contact with the seafloor can have a minimal impact on the habitat, and are more limited in terms of suitable terrain; however, they have the advantage of providing a fixed field of view (fixed distance from seafloor); and
- TUVS hovering above the seafloor are more flexible in the types of suitable terrain than TUVS touching the bottom and have little impact on habitat (i.e., limited to chain); however, quality of imagery might be reduced if too far from the seafloor or varying field of view.

5.2.1.6. Baited Remote Underwater Video Stations (BRUVS)

Baited Remote Underwater Video Stations (BRUVS) are stationary platforms equipped with a camera(s), lights, and some type of bait, commonly used to assess demersal fish or mobile invertebrates' distribution and/or behavior (Bailey et al., 2007; Przeslwaski and Foster, 2020). Once deployed through free-fall, BRUVS are left on the seafloor for a number of hours while collecting continuous video data. Due to their stationary nature, BRUVS are generally used to study mobile fauna, although in a few cases they have been used to assess benthic invertebrate diversity (Unsworth et al., 2014; Devine et al., 2019). It is relatively straightforward to deploy BRUVS, and they can be used as a time-lapse system to collect data on faunal behaviors (e.g., sea pen predation by sea stars), which could be useful in monitoring of Canadian OECMs.

Key points to consider:

- BRUVS are baited camera platforms mostly used to measure fish diversity, distribution and behavior; and
- These can be useful to measure the mobile species, fishes and invertebrates, that associate with corals and sponges.

Considerations on the use of imagery technologies

The choice of imagery tools and their specifications will depend on conservation objectives, monitoring indicators, survey objectives, habitat and depths being surveyed, local environmental conditions (e.g., clarity of water), as well as available resources. However, minimum specifications should be considered when working with imagery gear (e.g., gear speed, distance from the seafloor, lighting specifications, camera resolution, presence of laser points for size estimation, etc.), and these should be captured in protocols for imagery data collection. As previously mentioned, several protocols are available in the literature, and these could be used to guide the development of regional protocols.

Imagery data collection needs to be planned with specific and predetermined survey objectives. Video collected for one objective is often not useful for other objectives (Ahmadia et al., 2015). For instance, measuring vertical structures like sea pens would not be possible using a downward-looking camera. In fact, measuring vertical structures from imagery is always challenging, and alternatives (proxies for size) or method development will be necessary to confidently measure this variable, since size structure is one of the potential indicators for the monitoring of coral and sponge OECMs.

One aspect common to all imagery gear is the camera angle and resulting field of view. In the case of downward-looking cameras, field of view can be calculated if lasers or other scale are visible in the images. In the case of imagery obtained from forward-looking cameras, the camera angle needs to be considered for calculations of field of view area (Wakefield and Genin, 1987; Nakajima et al., 2014; Dias et al., 2015; Long et al., 2020). If for any reason information on camera angle is not available, calculations of the width of the field of view can still be possible based on lasers, allowing for estimations of total transect surveyed area if the surveyed distance is known. Photo-mosaic techniques can also be applied to calculate the area of images collected using a camera positioned at an oblique angle (Estomata et al., 2012). Precise field of view area is essential to allow accurate calculations of quantitative data such as the density of an organism.

A crucial consideration when planning on collecting and using imagery data is data management (Gomes-Pereira et al., 2016). The use of HD and/or 4K resolution yields large files. As an example, video files for an 8-hour long transect obtained using the ROV ROPOS can total >500 GB in size. At the end of a single expedition, there might be a need to store several terabytes of data. This is true for any HD system, and even more significant in the case of 4K or 8K imagery. While external hard drives are useful for data storage in the field, after several expeditions the management of hard drives (original and copies) might become challenging. Furthermore, physical hard drives can break, fail or be lost unexpectedly. Data storage systems which use RAID technology to pool disk drives together, are a good option, as these units can store >60 TB of data and have enhanced protection against drive failures. Cloud-based storage is also becoming a common approach to safely store and use imagery data. DFO is currently making the MS Azure cloud platform available (at a cost), which can be used with web-based software such as BIIGLE (B. Neves, personal communication). Storage cost, management, and cyber security are all factors that need to be considered when using cloud storage.

Imagery analysis is a time-consuming task. The collection of imagery data will therefore require the availability of in-house trained personnel or available funds to outsource qualified personnel. Imagery annotation and analyses have been and could continue to be conducted in-house at DFO. However, considering the government of Canada’s marine conservation targets, the scale at which monitoring activities will be conducted will require an increase in capacity through funding, access to ship, qualified personnel, and training. This capacity increase should come with stability for staff, as retention of qualified staff is important for maintaining data quality, and to ensure that it can be collected and processed in a timely manner.

Imagery annotation

Imagery data collection is only the initial step. Imagery annotation during and after surveys is a crucial factor to consider. During surveys, imagery annotation will depend on gear capabilities and level of gear sophistication. ROPOS users have access to the Integrated Real-Time Logging System (IRLS), which allows annotations of events of interest during the dive, and access to that information after the surveys. An “in the field” version of the open-source video annotation software BIIGLE is being developed (BIIGLE2Go) (Zurowietz et al., 2019). Real-time imagery annotation during surveys facilitates the posterior identification of events of interest (e.g., observation of rare taxon during a survey), which can be challenging when several hours of imagery data are available. If no specific on-survey annotation software is available, the use of a logging spreadsheet can also facilitate post-survey imagery processing. On-survey annotation is a consideration only when a live-feed of the seafloor is available.

It is the post-survey imagery annotation, however, that will contribute the most significant portion of the imagery data analysis. Imagery annotation can be performed in several ways, and it has been facilitated by the recent development of several software (reviewed in Gomes-Pereira et al. 2016). The use of specific software is not necessarily required, but it can reduce human error, accelerate the completion of tasks, and facilitate exchange between collaborators. For instance, the open-source web-based platform BIIGLE allows the annotation of photos and videos, the uptake of taxa names directly from WoRMS, comments from specific people, and measurements (Langenkämper et al., 2017), and has been used in the Pacific Region for several years (Nephin et al., 2020). Other DFO regions have started to use or have shown interest in using BIIGLE for imagery annotation (B. Neves, personal communication). Scientists at DFO-Maritimes Region have developed a MS Access database form (e.g., Beazley and Kenchington 2015) for imagery annotation in a more automated way than simply using MS Excel spreadsheets. Other specific open-source software include PAPARA(ZZ)I (Marcon and Purser, 2017) and MBARI's VARS (Video Annotation and Reference System). The software ClassAct Mapper (developed by Robert Benjamin of the Bedford Institute of Oceanography) has been used for processing and annotation of ROPOS videos for many years (e.g., Sameoto et al. 2008; Baker et al. 2012, Beazley et al., 2021), but it is no longer available to new users (R. Benjamin personal communication). In fact, software discontinuation is one of the concerns regarding the use of specific software for imagery analysis. Therefore, development of imagery annotation software – such as ClassAct Mapper - that can be maintained and used by DFO personnel in the long term should be considered.

Automated and semi-automated imagery annotation is also a current and future trend. In semi-automated annotation, there is input from a human annotator combined with artificial intelligence (AI) (Beijbom et al., 2015). AI techniques (e.g., machine learning, computer vision) can perform certain tasks faster and more efficiently. Automated identification of benthic taxa (particularly at low taxonomic levels) still requires important efforts and calibration (discussed in Gaston and O'Neill 2004); yet taxa identification at coarser taxonomic levels is certainly possible (Beijbom et al., 2015, 2016; Richards et al., 2019). Certain tasks, such as measuring objects and the distance between them, measuring image field of view, counting objects, etc, might be performed in an automated or semi-automated fashion with relatively high confidence (Beuchel et al., 2010; Aguzzi et al., 2011; Schoening et al., 2012; Beijbom et al., 2016; Piechaud et al., 2019; Richards et al., 2019). Other time-consuming tasks such as laser point distance measurement can also be performed using programming language software including Python and R or using plugins in ImageJ (Schneider et al., 2012), which might require less familiarity with more advanced programming techniques. Notably, image quality is of utmost importance in the case of automated tasks. For instance, automated measurement of distance between laser points requires lasers to be clearly visible and distinguishable from their surroundings. In the case of monitoring OECMs, automated technologies could be used to enhance efficiency of imagery annotation. However, it should be again emphasized that since these technologies are fast evolving, consistency across years needs to be maintained (e.g., level of taxa identification). Training of AI models depends on high-quality annotated training imagery performed by qualified professionals. Therefore, the use of AI for the monitoring of coral and/or sponge OECMs will require the development of robust human-made imagery annotation datasets representative of the study region. The success of applying these techniques for imagery annotation will be facilitated through the strengthening of collaborations between scientists with different backgrounds (e.g., ecological and computer sciences).

Long-term use of imagery technologies should also consider potential variation in data annotation between imagery analysts (i.e., annotators). Imagery annotation protocols need to be developed to maximize precision of annotation tasks and reduce inter-analyst variation (e.g., Beijbom et al. 2015; Matabos et al. 2017; Dunham et al. 2018a). The name of the analyst (or a code) should always be associated with annotations to allow checking for bias and/or to potentially include "analyst" as a random variable in statistical models. Software like BIIGLE offer the possibility of inviting guests to review and comment on annotations.

The development of regional guides and field keys for taxonomic identification from imagery should be considered in the context of monitoring coral and/or sponge OECMs. The development of these guides requires dedicated funding and a method for inter-regional collaboration. While guides for the Pacific region should be developed locally, the large overlap in coral and sponge diversity in East Canada would allow for the development of a single regional guide, which will depend on collaborative efforts across these DFO regions. The identification of both corals and sponges from imagery is challenging, and better efforts to create baseline catalogs of preserved vs. live samples are paramount. Therefore, efforts to collect imagery data and associated physical samples for the development of such guides are highly recommended. This is a good example of where the use of ROVs can be quite useful:

1. Collection of high-resolution imagery of a specimen *in situ*,
2. Collection of specimen,
3. Species identification in the laboratory,
4. *In situ* image-species id association made.

The use of open nomenclature is encouraged (e.g., *Genus* sp.) but it should be more consistently used and accompanied by the proper supplement (e.g., *Genus* sp. indet., rather than *Genus* sp.) (Sigovini et al., 2016). In a long-term context, the use of nomenclature supplements can assist different annotators to understand the level of confidence of a previous annotator or issues with annotation. The use of social networks such as iNaturalist can facilitate taxa identification from imagery, and taxa identification workshops should also be considered at regional levels. For instance, iNaturalist has been used in the Pacific Region to develop guides that are used in conjunction with the annotation software (Nephin et al., 2020). Marine imagery data annotation should be performed using a standardized vocabulary of defined terms, such as the classification scheme proposed by Sameoto et al. (2008) or the CATAMI Australian classification scheme. These data should also be collected through standardized, compatible sampling platforms (e.g., camera resolution) and even data processing hardware should be considered (e.g., monitor settings) to allow for better comparisons across space and time (Althaus et al., 2015; Dunham et al., 2018a).

Bowden et al. (2020) have recently published a comprehensive document on best practices in imagery annotation, which covers most of the points highlighted here, and which can be used to guide the development of protocols in a Canadian context.

5.2.2. Bottom-contact gear (non-imagery)

5.2.2.1. Bottom trawls, sleds, and dredges

Bottom trawls are an efficient method to sample fish. Although trawls can also sample some epibenthic fauna, they are considered more a qualitative method for the study of some benthic species (Jamieson et al., 2013). Trawl gear has low catchability for corals and sponges. As mentioned earlier, sea pen catchability for the Campelen shrimp trawl used by DFO-NL during multispecies surveys has been estimated to be only 5.2% (Kenchington et al., 2011). Another issue with using bottom trawls is that the coral and sponge catch often contains broken and damaged specimens, which in some cases make species identification challenging and skew measurements of biomass and abundance (e.g., fragments cannot be counted). Imagery data can provide better estimates of abundance in relation to trawl gear (Williams et al., 2015; Ayma et al., 2016; Chimienti et al., 2018b; De Mendonça and Metaxas, 2021; Uzmann et al., 1977). Yet, trawls can collect physical samples and might be used complement to imagery surveys.

Bottom trawls are limited to “trawlable” bottom (i.e., relatively soft bottom areas), as large rocks can damage the equipment and skew diversity estimates (Walsh et al., 2009). Areas of vertical walls and convoluted terrain also cannot be sampled using trawl gear. While commercial fishing trawls can be large and are usually towed for long periods, trawls deployed with scientific objectives are usually deployed for shorter periods of time. For instance, a Campelen trawl (16.8 m wide - door spread and trawl width) is usually deployed for 15 minutes (Rideout and Ings, 2018), in contrast to commercial fishing where trawls can be deployed for hours in a row (Cogswell et al., 2011). Alternative, smaller trawls such as the Agassiz and Beam trawls, have been used in benthic surveys for sample collection and megabenthos diversity assessments (Nephin et al., 2014; Roy et al., 2015; Sswat et al., 2015; Dinn et al. 2020a; Fredriksen et al., 2020). Agassiz trawls are usually 2-3 m wide (Jamieson et al., 2013) and therefore leave a much smaller footprint on the seafloor, in comparison to trawls like the Campelen. The mesh size of the Agassiz trawl cod-end is usually about 10 mm (Jamieson et al., 2013), which is more appropriate for the retention of small specimens (see discussion on the size structure indicator). The Beam trawl is larger than the Agassiz trawl (6 m wide), but the cod-end liner mesh is comparable. A notable difference with imagery gears is that bottom trawls usually cover much more area for a similar time allowing to get better widespread data over a large area. Despite issues with catchability, these trawls can still be used for diversity assessments, as long as towing speeds, cable payout (amount of cable released), and distance fished are equivalent between deployments (Jamieson et al., 2013).

Dredges are another type of seabed sampling gear capable of collecting epifauna and rocks, due to their sturdier nature in comparison to trawls. There are different types of dredges and their variations, two of the most common ones being the Naturalist's dredge and the anchor dredge (Jamieson et al., 2013). Dredges are generally heavier than small trawls and sleds, and usually have an outer protective bag, which allows them to be deployed in hard bottom areas without compromising the equipment. Since they are heavy, dredges can also be deployed at depths usually not accessible using smaller trawls. As with trawls, bottom type, depth, and cable payout are important factors to consider when deploying dredges, as variations in these factors can lead to data that is non-comparable. Despite the advantages of these tools to collect benthic samples, they are an invasive method that damages the habitat and brings samples that are often damaged and difficult to identify, and alternatives should be used whenever possible. An example of sled footprint was provided in section 4. These types of gear are not recommended for surveys of coral or sponge reef areas.

Key points to consider:

- Benthic trawls can collect samples of corals and sponges. However, they have low coral and sponge catchability;
- Dredges are another method that can collect coral and sponge samples, but they are similar to benthic trawls in that they are invasive and not ideal for monitoring of sensitive areas;
- Both benthic trawls and dredges can be used to collect coral and sponge specimens, however they often result in broken samples that can be difficult to identify and quantify.

5.2.2.2. Sediment samplers (grabs, corers)

Sediment samplers can be used for the collection of macrofauna/ meiofauna samples and allow for sediment physico-chemical characterization. Infauna diversity can be considered an indirect BCB associated with coral and sponge diversity, as a recent study has shown different diversity estimates in sediments associated with cold-water corals, versus bare sediment (Pierrejean et al., 2020).

Grabs, box-corers and push-corers can collect sediment samples, but each gear should be used with specific goals in mind as they are slightly different (Jamieson et al., 2013). Sediment samples collected using different equipment in the same area can still be comparable, as long as the sieve mesh size is the same (e.g., 0.5 mm for macrofauna) and sampled area/volume is considered in calculations of abundance. Jamieson et al. (2013) offer a comprehensive discussion and comparison of the different types of sediment samplers, so these will not be further discussed here.

Sediment samplers can be deployed with an acoustic beacon (if available), providing precise seafloor positioning information, which can be useful for long-term sampling. They can also be deployed with an associated camera (e.g., video grab, Gordon Jr. et al., 2000) which can allow quick observations of the piece of seafloor being sampled. It could assist in identifying a patch of corals and or sponges nearby, and other observations of interest.

Creating reference taxonomic (morphology), DNA, and eDNA libraries associated with infauna samples from Canadian OECMs can facilitate species identifications. For instance, the Atlantic Reference Center (ARC, St. Andrews NB) holds a collection of macrofauna samples collected from the Laurentian Channel MPA, including associated photographs. Finally, taxonomic identification and quantification of infauna samples (e.g., abundance and biomass) can be costly (hundreds of dollars per sample), and specific budgets will need to be allocated for that. As for other types of sampling, the number of sediment sampler replicates necessary in order to detect change should be assessed during the experimental design (e.g., power analysis – discussed in 5.3.2.2), as a considerably high number of samples is needed in some cases (e.g., Rogers et al. 2008).

Key points to consider:

- Sediment samplers can be used to monitor coral and sponge areas by collecting samples of the associated macrofauna/ meiofauna, as well as to characterize the sediments;
- Deploying sediment samplers with an associated camera can allow samples to be paired with observations of the seafloor.

5.2.3. Other tools

5.2.3.1. Acoustic techniques

Acoustic techniques such as singlebeam and multibeam echosounders (MBES) and sidescan sonar (SS) have been considerably useful in benthic habitat mapping (Kostylev et al., 2001; Brown and Blondel, 2009), and have been employed to directly identify and study cold-water coral and glass sponge reefs (e.g., Conway et al. 2001, 2005; Buhl-Mortensen et al. 2017a; Conti et al. 2019). Seamounts have been identified through both sonar and satellite altimetry and bathymetry data (Wessel et al., 2010; Yesson et al., 2011). The use of benthic habitat maps has also been advocated for use in the management of protected areas, including baseline data collection and monitoring (e.g., Brown et al. 2012; Copeland et al. 2013; Lacharité and Brown 2019; Baker and Harris 2020). Seafloor mapping resulting from MBES and benthic sampling (e.g., grabs and imagery) surveys led to the identification of benthoscapes in the Laurentian Channel MPA (Lacharité et al., 2020). These benthoscapes represent areas of the seafloor with specific geomorphological features and associated benthic fauna and can be useful in the identification of monitoring areas. High-resolution seafloor mapping of most Canadian OECMs has not been conducted.

These techniques have also been used beyond the identification of features and have been considered for expanded use in a monitoring context. For instance, sonar technology was used to assess distribution and density of trawl marks in a conservation area, providing indication of whether violation of a fishery closure has occurred (e.g., Huvenne et al. 2016). Further, these authors were capable of identifying *Lophelia pertusa* (framework-forming coral) growth using a combination of AUV-collected high-resolution sidescan sonar data and ROV video footage. The possibility of mounting MBES and SS directly on ROVs and AUVs allows for increased spatial data resolution and can facilitate the survey of vertical walls, generally inaccessible through other types of gear (e.g., Robert et al. 2017).

Although it is possible to use acoustic techniques alone to identify large structures such as reefs and seamounts, ground-truthing via imagery or other seabed surveys (e.g., towed gear, sediment cores) is still generally required to confirm interpretations from acoustic data. Furthermore, identification of individual corals and sponges or even areas with significant concentrations (e.g., sea pen fields) is still not possible using acoustic data alone. Advances in acoustic technologies, such as Kraken's Katfish with Synthetic Aperture Sonar (SAS), which promises centimeter-scale resolutions (Steele et al., 2019), might make this possible in the future. The use of acoustic technologies not related to seafloor mapping include an interesting case of sonar systems used to monitor hydrothermal flow in hydrothermal vents (Rona and Light, 2011; Bemis et al., 2015).

Passive acoustics instruments (e.g., hydrophones) also have the potential to be used for the monitoring of indirect BCBs. For instance, Archer et al. (2018) found that soundscapes in glass sponge reefs differ from soft sediment areas (non-reef) and identified fish sounds within the reef area. Furthermore, Lin et al. (2019) suggested that soundscapes might be important settlement cues for deep-sea benthic larvae (i.e., to detect their habitats). Therefore, the use of soundscapes has been argued as a potential new conservation tool, particularly useful in deep-sea habitats (Lin et al., 2019).

Key points to consider:

- Acoustic techniques are extremely useful in seabed mapping and identifying coral and sponge reef areas;
- They can also assist in the surveillance of a protected site, as they can identify trawl marks;
- Currently, acoustic techniques need to be ground-truthed with imagery and/or sediment data; and
- Passive acoustic data collection has the potential to assist in monitoring and should be a tool further explored for monitoring coral and sponge areas.

5.2.3.2. eDNA

Environmental DNA (eDNA) is a molecular tool that allows identification of multiple taxonomic groups from a specific area based on a sample of the environment where those organisms lived (Loeza-Quintana et al., 2020). In the case of marine taxa, seawater and sediment samples are usually targeted for eDNA research. The use of eDNA is particularly useful for the identification of cryptic or rare species, which would often not be visualized or sampled using imagery or physical sampling tools (e.g., trawls). The range of eDNA applications has been increasing over the past years, due to ease of sample collection, relative low sampling cost, and refinements of the technique (Garlapati et al., 2019; Loeza-Quintana et al., 2020). Samples for eDNA assessment can be collected from Niskin bottles and sediment samplers, which are usually straightforward to deploy and do not require complex field logistics.

The use of eDNA for the identification of cold-water corals and sponges is still in its infancy (Bell et al., 2017; Everett and Park, 2018; Kutti et al., 2020); however, it has been shown to yield good data on general megafauna diversity, which can be useful in monitoring (Mächler et al., 2014; DiBattista et al., 2020; Sevellec et al., 2020). eDNA is also proving to be reliable as a source of infauna data based on water samples (Leduc et al., 2019). eDNA can be paired with other taxonomic information gathered from other types of sampling such as trawls and sediment sampling to create reference libraries of species present in an area.

Key points to consider:

- eDNA is a useful tool for collecting data on species presence without physical sampling methods or visual tools;
- Using eDNA to detect corals and sponges is under development and is a promising tool for scientific monitoring.

5.2.3.3. Oceanographic moorings and benthic landers

Oceanographic moorings consist of a vertical chain of rope, buoys, and oceanographic instruments (e.g., CTD, current meters, sensors, sediment traps) anchored to the seafloor by a bottom weight (> 1 ton). Moorings are left underwater for a set period of time (often limited by the battery life of instruments), and usually recovered using an acoustic release (or a grapple if release fails). Data collected from moorings provide information on spatio-temporal variation in oceanographic parameters such as temperature, salinity and currents, which is important for monitoring purposes. However, moorings are large, heavy, and therefore have important logistics associated with their deployment and recovery. Depending on which instruments will be attached to it, it might also be expensive. Moorings can provide oceanographic information to an overall monitoring plan for coral and sponge areas and could provide useful ecological context for any changes noted during monitoring. Furthermore, sediment traps attached to moorings could be useful to collect baseline data on sediment resuspension and presence/absence of benthic organisms displaced as a result of bottom trawling (e.g., Pilskaln et al., 1998).

Benthic landers are a type of mooring that hosts benthic chambers usually used for incubation experiments, in which biogeochemical measurements (e.g., oxygen, nutrients) can be performed *in situ* at relatively long temporal scales (e.g., one year or longer). Landers are generally deployed through free fall methods, but they are equipped with buoyancy and ballast release systems, which allow them to sink slowly to the seabed, causing minimal seafloor disturbance before the chambers contact the sediment (Bagley et al., 2004; Gage and Bett, 2005). Landers can also carry cameras, other instruments, and sensors (e.g., current meter, CTD), sediment traps and settlement plates (for larval settlement studies) and SPIs (Sediment Profiler System, mentioned earlier in this document - can also be deployed from ROVs).

Key points to consider:

- Oceanographic moorings can provide information on oceanographic parameters over the time of deployment and can also be outfitted with several instruments and tools. They can be expensive and difficult to use in certain conditions, but are useful for longer-term passive data collection;
- Benthic landers are used to perform long-term *in situ* experiments and can also collect data using cameras, sensors, and other instruments.

Tool comparison

A comparison of the tools described in this section is shown in Table 7. Due to variation in capabilities of a same type of tool (e.g., ROVs), this table should be seen as a general overview, but acknowledging that there are exceptions within each gear class. Similarly, the outcomes for the cost category (which does not consider ship time) can be variable depending on the need for initial gear acquisition and repeated use over time, versus the need to invest in the gear every time that it is needed. For instance, a drop camera system might cost more than a Van Veen grab, but there should be no considerable costs after the initial investment. Applicability, advantages, and disadvantages of broad- and fine-scale survey methods for biogenic habitats were also compared by Bailey et al. (2007) and Loh et al. (2019). While each tool has their capabilities, issues related to cost and accessibility are the ultimate factors defining the selection of tools for monitoring surveys. Specific protocols for the use of tools for the monitoring of OECMs could be produced as a technical report for guidance at the national or regional levels.

5.3. METHODOLOGIES - MONITORING DESIGN

In addition to clearly defining monitoring objectives, selecting indicators, and choosing adequate tools as described in the previous section, a monitoring design needs to be developed (Figure 9). This section aims to describe some methodology considerations and best practices for designing monitoring programs in benthic habitats such as coral and sponge OECMs. The intent is not to provide recommendations on specific sampling design for each Canadian OECM, which would require a case-by-case review. The monitoring design needs to be statistically robust so it can allow conclusions to be drawn concerning the observed change, and thus effectively be used to inform management of conservation areas.

In practice, monitoring design will be directly linked to availability of resources (e.g., funding, access to tools and vessels) and must be adapted to specific habitat characteristics (Figure 9). Canadian coral and sponge OECMs have notable differences in size varying from 15 km^2 to 55,000 km^2, depths ranging from 50 m to over 3000 m (Table 1), and location, with some areas being more costal and others more difficult to reach offshore (Table 2). Heterogeneity of habitats inside one OECM also must be considered, especially when defining the best sampling design to use. While some Canadian OECMs are more homogeneous, most cover a wide bathymetric and environmental gradient supporting various benthic communities (Table 2). Therefore, the monitoring design of each OECM will certainly have to be adjusted to those differences. The Joint Nature Conservation Committee (JNCC, UK) has published a comprehensive report on monitoring guidance for marine benthic habitats, which details critical steps to be considered while developing a monitoring design, such as statistical considerations, frequency of sampling, size of sites, etc. (Noble-James et al., 2018). Australian scientists also recently published a field manual for marine sampling with a specific chapter on statistical considerations for monitoring and sampling (Przeslwaski and Foster 2020; Foster et al. 2020). Loh et al. (2019) performed a similar review (Figure 9) and described a case study on Pacific glass sponge reefs (Dunham et al., 2018a; Loh et al., 2019), which is particularly relevant to the present document. Based on those publications some of the primary questions that should be asked in order to have a complete and robust monitoring design include:

- **What** baseline data are available and how can we use them?
- **How** much should we sample?
- **When** and **how often** should we sample?
- **Where** should we sample? (sampling design)

This section reviews recommended best practices and general information that should be considered to answer these questions, but specific regional processes (e.g., workshops, CSAS meetings) will need to be conducted to develop adequate monitoring design for specific OECMs.

5.3.1. Baseline data

For most conservation areas, there is some baseline data available which can provide information to guide the development of a monitoring design. In the case of Canadian OECMs, baseline data gaps exist to varying degrees which will need to be addressed. Precise bathymetry and bottom types are incomplete and poorly known in most OECMs. Information on the distribution and diversity of the specific taxa we wish to protect in the OECMs is often not well known. For instance, DFO trawl survey strata are limited to depths of 1,500 m, which excludes the deep-water portions of several OECMs. Information on these areas is therefore lacking or limited. It was not in the scope of this document to assess in detail the state of baseline data for Canadian OECMs, but regional investigations on what kind of baseline data is available is an important step of the monitoring design.

Before using any existing data in a monitoring program, the data should be carefully evaluated to ensure that they are suitable for the purpose. JNCC has developed a list of considerations and questions by type of data (e.g., grab data, imagery data, trawl data, etc.) that can be useful for achieving this task (Noble-James et al., 2018). Data comparability is a key aspect when considering temporal data (i.e., data collected before and after the area has been closed).

When monitoring benthic conservation areas such as coral and/or sponge OECMs, the availability of bathymetric data and bottom type, which can be used to describe habitats, is a key factor for the development and implementation of a good sampling design (Noble-James et al. 2018, Przeslwaski and Foster 2018). As described in the previous section, multibeam sonar and other acoustic tools can provide high resolution bathymetry and backscatter data that can be used to map the seafloor and allow benthic habitat classification (in combination with ground-truth sampling). At present, these are major data gaps for most Canadian OECMs, which will be challenging to overcome.

Suggestions:

- When using existing data as the first point in a monitoring time-series, current monitoring practices should be aligned wherever possible with the existing data (e.g., in terms of survey timing, operational methods, equipment, processing and analysis techniques).
- Acoustic seabed characterization surveys (multibeam bathymetry and backscatter) should be conducted prior to the development of a sampling design, if resources and logistical limitations (i.e., area to cover) allow.

5.3.2. Sampling and statistical considerations

5.3.2.1. Size and replication

The size and type of the sampling unit (Table 8) should be related to the density and expected distribution of the taxa targeted by an indicator. For example, areas where coral and sponge density are sparse will be monitored more efficiently with larger relative sampling units (e.g., video transects) than with sediment grab samples at discrete points (Figure 10) (Noble-James et al., 2018). Appropriate size of sampling units increases the likelihood that rare or sparsely distributed taxa will be captured within the sample (Table 8).

Benthic organisms are often patchy in their distribution and abundance and thus form a system with naturally high spatial variation (Underwood and Chapman, 2005). Patchiness is an important factor to be considered in the collection of sediment samples for infauna characterization. The collection of replicates can balance the potential infaunal patchiness and is essential to consider when planning surveys. Sampling units should be replicated at the scale at which variability can be meaningfully measured. The level at which one replicates allows the determination of variation. Replication can be nested to assess different sources of variability and can be used to improve precision (in which case you would pool replicates) or accuracy (in which case you include the individual replicates in your analyses or pattern description) (Table 8). When resources are limited, collecting samples from a wider range of sampling stations should be prioritized over within-station replicates (Holtrop and Brewer, 2013). Loh et al. (2019) have highlighted some important points to be considered for the monitoring of heterogeneous environments such as smaller, more numerous sampling units, which are better at estimating cover (density permitting) or detecting change within a sampling stratum than a few large sampling units.

Suggestions:

- Replication of sampling units should be conducted where resources allow (e.g., multiple grab samples per sampling station) to reduce the effects of environmental ‘noise’ or random variation, and to provide a more accurate and precise estimate.
- Replication should occur at the level at which one needs to understand variability.

5.3.2.2. Sample size, power and significance

The optimal sample size is directly linked to the environment and indicators that are being monitored, as well as the type of statistical analysis that will be required. For statistical inference (e.g., hypothesis testing), robustness relies on accurate measurement of the selected indicator through the acquisition of a sufficient number of samples (Underwood and Chapman, 2005; Noble-James et al., 2018). Monitoring design with too little statistical power produces ambiguous results that are hard to use for management and difficult to justify (Karpov et al. 2010).

Power analysis is a statistical tool that can help to identify the optimal sample size to ensure robust conclusions. Power analysis (Table 8) can provide important information on how much sampling effort would be required in order to allow the detection of a meaningful change (e.g., 20% increase or 10% decrease in mean) (Karpov et al. 2010; Stanley et al. 2015; Noble-James et al. 2018). Power analysis therefore allows for a cost benefit analysis between sampling effort and detectable differences. It can also help inform the choice of indicators, as it is possible that you require too many replicates for a specific indicator, reducing usefulness in a monitoring design. Karpov et al. (2010) determined the optimal transect size and the amount of sampling effort needed to measure statistically significant differences in fish density between study areas using ROV transect data and power analysis. JNCC developed a flexible framework which can be used to help define appropriate ratios and levels of power and significance related to benthic monitoring (Noble-James et al., 2018). Power analysis generally requires knowledge based on previous research in the study area, for instance mean values and variance for the parameters of interest. Quinn and Keough (2002) provide a sequence of steps to be considered when using power analysis to design experiments.

Suggestions:

- When existing data are available, power analysis can be conducted *a priori* to determine the minimal sample size (N) needed to detect change of a given magnitude at a given level of significance for each indicator.
- Power analysis can help determine indicator choice, as it infers how much data is needed for an indicator to be able to detect change.

5.3.2.3. Statistical issues around data independence

While developing a scientific monitoring design, independence between sampling units must be considered. Autocorrelation of response variables in space and/or time is common in the marine environment, violating the statistical assumption of independence of data needed for many standard statistical tests (Table 8; independence of residuals). In addition, one of the most common errors in the design and analysis of ecological field experiments is the issue of pseudoreplication in experimental design (Hurlbert 1984). Pseudoreplication occurs when sampling units that are highly spatially or temporally correlated are treated as if they were independent.

Therefore, while developing a monitoring design, issues such as spatial autocorrelation, serial correlation and pseudoreplication need to be considered and taken into account (Underwood and Chapman, 2005; Noble-James et al., 2018) (Table 8). Many advanced statistical techniques such as mixed modelling can be used to include spatial and serial autocorrelation. Statistical techniques can, among other things, be used to detect the scale of autocorrelation and aggregate the data at the proper scale. Depending on the monitoring objectives, different statistical approaches can be used to minimize or take into account the dependency issues (e.g., re-randomizing sampling locations, repeated measures analysis).

Spatial autocorrelation is a particularly important factor to be considered when analyzing video transect data, and geostatistical models that account for the spatial dependency between observations can be used to account for this limitation (Zuur et al. 2009, Foster et al. 2020). Another possibility to mitigate autocorrelation is to subset the individual observations within a transect. However, doing so presupposes that the range of the autocorrelation is less than the distance between the sub-set observations (Foster et al. 2014, Foster et al. 2019). Spatial autocorrelation can be addressed by thoughtful sampling design (e.g., Misiuk et al. 2019).

Suggestions:

- Autocorrelation and pseudoreplication are common issues in ecological experiments, and study design and statistical techniques can address these factors.
- It is important to evaluate spatial autocorrelation when analyzing video transect data, either before data is collected through sampling design, or after using statistical techniques.

5.3.3. Temporal consideration and frequency

The timing of sample collection (i.e., seasonality) should be planned by considering the ecology and life history of the relevant indicator taxa and temporal variation of the ecosystem. Seasonal variations in benthic ecosystems can introduce variation into time-series data (Stanley et al., 2015; Noble-James et al., 2018). For corals and sponges, which are sessile/sedentary, seasonality might not directly influence their distribution or diversity metrics. However, little is known about the influence of seasonality on population dynamics, such as predation. For instance, shallow-water populations of the sea pen *Ptilosarcus gurneyi* in the Northeast Pacific are perennial and seem to be controlled by seasonal changes in the abundance of one of its predators, the nudibranch *Tritonia diomedia* (Wyeth and Willows, 2006). Although *P. gurneyi* is not one of the sea pen species found in current Canadian OECMs, it represents an example of ecological dynamics to be considered. If using imagery gear, sampling should be avoided during the spring, when water clarity can be poor due to spawning events and algal blooms (or marine snow), which can decrease imagery quality (Dunham et al., 2018a).

Frequency of sampling will depend on the conservation objectives and the taxa associated with them. Monitoring indicators such as numerical abundance need to be considered in light of the life history of the taxa of interest, such as growth rate and recruitment rate. For instance, sea pens are generally less long-lived than large gorgonians (Sherwood and Edinger, 2009; Neves et al., 2015a; Murillo et al., 2018a; Neves et al., 2018). Therefore, it is expected that it should take longer to identify trends regarding changes in the abundance of gorgonians (Bennecke and Metaxas, 2017) in comparison to sea pens. In a monitoring context, detection of change in biological response can vary from a few years to decades (Stanley et al., 2015). Therefore, optimal sampling frequency will likely vary for different taxa/indicators within and between OECMs (i.e., differences in depth, bottom types, and conservation objectives).

The appropriate frequency of sampling should also vary across habitats based on the relative risk from human pressures. Habitats under more pressure should be monitored more often (Loh et al., 2019). Smith and Hugues (2008) recommended annual surveys for the monitoring of deep-sea habitats with photo / video surveys for high value sites or where monitoring (e.g., VMS data) suggests impacts or new threats.

Frequency will also heavily depend on the resources available. Due to economic and logistical factors, the trade-off between the frequency of monitoring and sampling effort must be considered. Lewis et al. (2016) suggested frequencies ranging between annual and tri-annual surveys for the monitoring of sea pens (focus of conservation objectives) in the Laurentian Channel MPA. In the case of indirect indicators (e.g., infauna species diversity), annual monitoring frequency has been suggested, as a way to take advantage of DFO multispecies surveys in the area (assuming scientific trawl surveys continue inside of closed areas and that multispecies surveys represent a platform for such sampling). In the case of Pacific glass sponge reefs, monitoring has been suggested to take place at three to ten-year intervals, combining two different frequencies for different spatial scale sampling:

1. frequent sampling at a limited number of fixed sites, and
2. less frequent routine broad- scale surveys (Dunham et al., 2018a).

It should be highlighted that data collection is not the only factor that is restricted by the availability of resources. When not performed in-house, sample processing and analyses can be costly (sometimes hundreds of dollars per sample in the case of sediment infauna, an indirect BCB). If performed in-house, high sampling frequencies (e.g., annual) will generate a considerable increase in the workload of employees, and resources associated with the hiring of additional personnel might need to be considered.

Suggestions:

- For corals and sponges, seasonal timing might not be an issue for some indicators such as density but may be important to consider for other indicators related to population dynamics (e.g., predation) or to indirect BCBs (specific associated biodiversity).
- The use of imagery for monitoring should be planned to avoid periods of reduced visibility, such as spring spawning, algal blooms, sediment resuspension, and meltwater).
- If the main goal is to detect interannual variation, monitoring surveys should be repeatedly undertaken in the same season as much as possible to avoid potential artifact of seasonal variation. If not possible, the potential impact on the time-series should be defined.
- Detecting changes in the status of different coral and/or sponge groups or associated BCBs might require specific monitoring frequencies per group based on differences in growth rates (e.g., different frequencies for sea pens and gorgonians).
- Monitoring frequency should be re-examined for all OECMs as the monitoring progresses and data on various trends are collected and analyzed.

5.3.4. Sampling design

The sampling design will inform how sampling units will be selected and how they will be distributed throughout the study area. Sampling design can be divided into probabilistic and non-probabilistic methods (Table 8). Probabilistic methods are those where sampling units have the same theoretical probability of being selected, and therefore this type of design is generally considered more statistically robust. Designs include systematic sampling (grid), simple random sampling and stratified random sampling (Table 8, Figure 11). Non-probabilistic methods involve the subjective selection of the sampling units (judgement sampling). Each type of design has strengths and limitations, which have been reviewed by others (Noble-James et al., 2018; Foster et al., 2018), and only briefly considered in this section.

The design of a sampling regime depends on many factors, including the availability of resources. The use of probabilistic methods ensures that the information contained from the samples can be generalized to the wider population (i.e., whole conservation area) (Noble-James et al. 2018, Przeslwaski and Foster 2018). Randomized or systematic monitoring designs (Figure 11 a and b) are a good choice in areas that are well-known or relatively environmentally homogenous. Systematic sampling (Figure 11 a) is one way to ensure spatial autocorrelation is avoided, as it places sampling units at fixed distances. It is a useful way to investigate spatial patterns. Where sediments or pressures are clearly stratified across the area, and when there is good confidence in habitat maps, the use of stratified random sampling (Figure 11 c) can considerably increase accuracy and precision. This sampling design ensures that all the main habitat types can be adequately sampled (Noble-James et al., 2018). With random sampling, care must be taken to avoid spatial autocorrelation, either by ensuring enough distance between samples (using a buffer), or by including location as a model variable.

Non-probabilistic sampling, such as judgement sampling, is a useful method in cases where the study area is well-known, where resources are limited and researchers want to focus on known species or areas with rare species or habitats, or when certain areas are known to be representative. The use of judgement sampling (Figure 11 d) has a high risk of bias, as sample areas are chosen specifically rather than randomly. Care must be taken when extrapolating results from judgement sampling to a wider population, as the data is inherently biased (Noble-James et al., 2018). Fixed judgement sampling can provide a very precise measure of change by reducing random spatial variability, and can be useful for monitoring the growth, density, cover or condition of biota such as biogenic reefs, marine flora and sessile fauna (Davies et al. 2001, Noble-James et al. 2018). An important limitation of fixed sampling units is that resampling must occur in the exact same location, which is crucial because even small differences can contribute to considerable variation in cover estimates (Davidson, 1997). Even with today’s advances in navigation technologies, revisiting a specific location might pose challenges, and choosing the appropriately accurate tool will be crucial. Alternative methods might be investigated. For instance, Williams et al. (2012) suggest a grid rather than a point system, which could facilitate site-revisiting (in their case with an AUV).

A combination of different sampling designs can be used to optimize resources and sampling results. A good example is the case of the Pacific glass sponge reefs. There is considerable variability in sponge distribution between transects (sampling units) of the same reef complexes, and the ability to detect real change in reef character over time is a challenge (Dunham et al., 2018a). The authors recommended a combination of fixed and random transects at different frequencies. More frequent routine surveys with fixed transects will allow trends in relative sponge abundance to be assessed, while intensive surveys with stratified random transects are more likely to capture impacts from localized stressors (e.g., fishing due to noncompliance) (DFO, 2017b; Dunham et al., 2018a) and will allow validation of representativity of the fixed transect.

For large and/or diverse conservation areas, a more achievable and robust design could involve sampling intensively in nested boxes selected within the area that are representative of the different habitats (Noble-James et al., 2018). Nested box design involves choosing fixed sites in which sampling units can be randomly selected or using a systematic grid (Figure 11 e, f). Boxes will need to be considered as a factor in the statistical model (e.g., mixed model). Many of the coral and sponge OECMs are considerably large areas, and even with substantial resources it would be extremely difficult to conduct enough sampling across the entire area with sufficient replication to detect an effect. Nested boxes should be selected to cover the different types of habitat but each box should minimize natural variation such as bathymetry. Sampling intensively in nested boxes will increase the power of the design and the likelihood of detecting an effect within the context of each box. Inference about the un-sampled areas can only be assumed, but acquisition of data from random design extended to the entire area can improve confidence in this inference.

Suggestions:

- Probabilistic sampling design ensures that data are randomized and are statistically robust designs best employed in well-known and environmentally homogenous areas or well stratified areas.
- Fixed sites (non-probabilistic designs, judgement sampling) are useful in areas that are well-known, and resources are limited, for areas that are representative, or for rare species and habitats. Appropriate statistical techniques must be employed, and conclusions must be considered within the limitations of the bias created by this sampling design.
- A combination of sampling designs, for example nested boxes, may be an ideal way to ensure data robustness while focusing on known areas of coral and sponge distribution.

5.3.4.1. BACI design

When monitoring aims to evaluate the effectiveness of conservation measures**,** Before-After-Control-Impact (BACI) design is one of the more appropriate methods (Stanley et al., 2015; Noble-James et al., 2018). BACI sampling design considers two main factors: 1) management measures or experimental manipulation which represent the ‘Impact’ sites compared to ‘Control’ sites, and 2) sampling periods which compare ‘Before’ and ‘After’ management measures or an experimental manipulation. This type of design is particularly powerful because it controls both temporal and spatial variation, improving the robustness of conclusions on management effects. Suitable reference sites (control sites) must be available, and sites should remain fixed, whilst within-site sampling units may be re-randomized or fixed.

The use of external reference sites is highly recommended to optimize the scope of conclusions and is required for BACI or similar sampling designs. The placement of references sites should:

- Be in relatively close proximity to the conservation area but not directly adjacent to it to avoid biological ‘overspill’ or edge effects;
- Ideally have comparable environmental conditions (e.g., hydrodynamic regime and organic inputs) to those of the conservation area and have same type of substrates; and
- Consider the distribution of pressures within and between reference sites and “impact sites” (the conservation area) so they have similar historical levels.

Historical data needs to be available for BACI sampling design. In the case of conservation areas already in place like Canadian OECMs, historical data, such as trawl survey data, may be useful to inform some indicators and to be used in the BACI design. BACI design is a good theoretical goal, but it may be a practical impossibility for some of our OECMs. If “before” data are not available, specific statistical analyses (i.e., using habitat availability as a covariate) can be used, which requires a solid understanding of the indicator, system and pressure, and the relationships between them (Osenberg et al., 2011; Noble-James et al., 2018).

Suggestions:

- BACI design is useful when evaluating the effectiveness of a conservation area.
- This sampling design requires historical data to measure “before”, which isn’t always the case for Canadian OECMs. Statistical analyses can help in situations without historical data.

5.4. CONCLUSION

Large portions of Canadian OECMs remain unexplored, and major gaps in baseline data for these areas still exist. Although baseline data ideally would have been collected before designation of the OECMs, this wasn't always the case. Habitat mapping and taxonomic inventories for unexplored areas of these OECMs still need to be conducted so that more complete datasets can be available for developing monitoring design.

In practice, baseline and monitoring-associated data collection for these OECMs will represent a significant undertaking. For instance, some coral and sponge OECMs are quite large (e.g., NL has three OECMs > 80,000 km^2 in area), and others are found in more remote locations (e.g., Arctic) which might limit sampling frequency. Some regions have many OECMs (e.g., Pacific glass sponge reefs (17), Gulf of St. Lawrence coral and sponges (11), which limits the synchronicity of monitoring cycles, and hence survey optimization. Regional processes will be necessary to develop adapted specific monitoring design and prioritize monitoring actions. Once baseline data is clearly assessed and logistical constraints evaluated at the regional scale, workshops to help develop sampling design in a consistent manner between regions would be beneficial and should be considered.

The tools and techniques described in this document provide an overview of the options available in the context of cold-water coral and sponge research. Tools should be selected based on their utility and efficiency to measure a particular indicator. In practice, researchers are often limited to the available tools and resources (e.g., ships and ship time). Given these real constraints, it is critical that the proper sampling design and statistical techniques are applied to optimize data collection and facilitate data analysis. Strong collaboration between scientists with different backgrounds (e.g., statisticians and ecological scientists) here too will be paramount.

6. CONCLUDING REMARKS

The information contained in this document provides a snapshot of the most current information available on corals and sponges in Canada, and a fulsome review of potential monitoring indicators, tool, and methodologies used in Canada and worldwide. It is critical that we ensure that monitoring plans are designed to be both efficient and effective. The study of these benthic areas is costly and requires trained staff, dedicated equipment, and ship time to conduct research. We need dedicated funds and personnel to monitor these areas so that we can demonstrate that excluding bottom-contact fishing protects corals and sponges and provides associated ecosystem benefits. We hope that the information provided herein can help both scientists and managers to design effective monitoring plans for coral and sponge OECMs. This work is also applicable to many existing and proposed MPAs that have benthic conservation objectives.

A recommendation to ensure that monitoring is designed efficiently is to focus efforts on the direct BCBs of the OECMs: corals and sponges. As detailed in Section 3, corals and sponges are likely to have predictable ecosystem associations. It may be appropriate to focus monitoring on corals and sponges, while inferring indirect BCBs. If resources are available, operational monitoring could conduct targeted studies to confirm the link between corals and sponges and their indirect BCBs. Additionally, scientific trawling or other fisheries research could be tasked with collecting indirect BCB information. For example, fish stock assessment surveys could also collect sediment data for infauna studies. It would be especially helpful in areas where data is limited, and research is difficult to limit or eliminate indirect BCB monitoring. A fulsome monitoring program for corals and sponge OECMs can be achieved by focusing on direct BCB monitoring while using prior knowledge along with occasional targeted studies to infer indirect BCBs.

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8. TABLES

Table 1. List of Canadian coral and sponge OECMs by Region, their conservation objectives, size, depth range, and representative functional groups.

Region	OECM	Conservation objective	Area (km^2)		Functional groups represented											
				Min/ Max Depth of the closure (m)	**Large gorgonians**	**Small gorgonians**	**Sea pens**	**Soft corals**	**Black corals**	**Cup corals**	**Reef-building corals**	**Hydrocorals**	**Glass Sponge Reef**	**Vazella sponge grounds**	**Astrophorid grounds (e.g.Geodia)**	**Others: mixed sponges**
Pacific	Strait of Georgia and Howe Sound Glass Sponge Reefs (17 areas)	Protect glass sponge reefs	32.6	22-138	na	na	na	na	na	na	na	na	x	na	na	na
C&A	Davis Strait Conservation Area	To conserve sensitive benthic areas	17,298	355-791	x	x	x	+	na	+		na	na	na	x	x
	Disko Fan Conservation Area	Minimize impacts on winter food source and overwintering habitat for narwhal & Conserve coral concentrations	7,485	386-1619	x	+	+	+	+	+	na	na	na	na	+	+
C&A / NL	Hatton Basin Conservation Area	To conserve sensitive benthic areas	42,459	223-2307	x	x	+	+	+	+	na	+	na	na	x	x
QC/Gulf/NL (QC leads)	Beaugé Bank Sponge Conservation Area	Cold-water sponge protection	215	71-119	na	na	na	+	na	na	na	na	na	na	na	x
	East of Anticosti Island Sponge Conservation Area	Cold-water sponge protection	939	52-203	na	na	na	+	na	na	na	na	na	na	na	x
	Jacques-Cartier Strait Sponge Conservation Area	Cold-water sponge protection	346	52-144	na	na	na	+	na	na	na	na	na	na	na	x
	South-East of Anticosti Island Sponge Conservation Area	Cold-water sponge protection	845	270-424	na	na	+	+	na	na	na	na	na	na	na	x
	Parent Bank Sponge Conservation Area	Cold-water sponge protection	530	57-215	na	na	+	+	na	na	na	na	na	na	na	x

Region	OECM	Conservation objective	Area (km^2)	Min/ Max Depth of the closure (m)	Functional groups represented											
					Large gorgonians	Small gorgonians	Sea pens	Soft corals	Black corals	Cup corals	Reef-building corals	Hydrocorals	Glass Sponge Reef	Vazella sponge grounds	Astrophorid grounds (e.g.Geodia)	Others: mixed sponges
	Eastern Honguedo Strait Coral and Sponge Conservation Area	Cold-water coral and sponge protection	2,338	257-407	na	na	x	+	na	na	na	na	na	na	na	x
	Central Gulf of St. Lawrence Coral Conservation Area	Cold-water coral protection	1,284	372-441	na	na	x	+	na	+	na	na	na	na	na	+
	Western Honguedo Strait Coral Conservation Area	Cold-water coral protection	496	353-403	na	na	x	na	na	na	na	na	na	na	na	
	North of Bennett Bank Coral Conservation Area	Cold-water coral protection	821	285-433	na	na	x	na	na	na	na	na	na	na	na	+
	Slope of Magdalen Shallows Coral Conservation Area	Cold-water coral protection	335	342-425	na	na	x	na	na	na	na	na	na	na	na	+
QC/Gulf/NL (Gulf leads)	Eastern Gulf of St. Lawrence Coral Conservation Area	Cold-water coral protection	423	407-486	na	na	x	na	na	na	na	na	na	na	na	na
MAR	Corsair and Georges Canyons Conservation Area (restricted bottom fisheries zone)	Cold-water coral protection	8,797	136-4708	x	na	na	na	na	na	na	na	na	na	na	na
	Emerald Basin and Sambro Bank Sponge Conservation Areas (2 areas)	Protection of *Vazella pourtalesii*	260	103-245	na	na	+	na	na	+	na	na	na	x	na	+
	Jordan Basin Conservation Area	Cold-water coral protection	49	142-220	x	na	na	na	na	na	na	na	na	na	na	
	Lophelia Coral Conservation Area	Protection of *Lophelia pertusa* coral reef	15	755-254	+	na	na	+	na	+	x	na	na	na	na	+
	Northeast Coral Conservation Area	Cold-water coral protection	391	210-1531	x	+	+	+	na	+	na	na	na	na	na	na
NL	Division 3O Coral Closure	Coral and sponge protection	10,422	365-3505	x	x	x	+	na	+	na	na	na	na	na	+
	Hopedale Saddle Closure	Protection of corals and sponges and	15,411	55-2689	x	+	+	+	na	+	na	na	na	na	na	+

Region	OECM	Conservation objective	Area (km^2)	Min/ Max Depth of the closure (m)	Functional groups represented											
					Large gorgonians	Small gorgonians	Sea pens	Soft corals	Black corals	Cup corals	Reef-building corals	Hydrocorals	Glass Sponge Reef	Vazella sponge grounds	Astrophorid grounds (e.g.Geodia)	Others: mixed sponges
		contribution to long-term conservation of biodiversity														
	Northeast Newfoundland Slope Closure	Protection of corals and sponges and contribution to long-term conservation of biodiversity	55,353	258-3298	x	+	x	+	+	+	na	na	na	na	na	+

X Included in conservation objectives, major group represented
+ Other group that has been identified in the OECM but is secondary to the conservation objectives
Blanks do not necessarily mean absence of a functional group, but rather indicate knowledge gaps

Table 2. Cold-water coral and sponge groups and examples of species found in Canadian OECMs.

Functional group	**Species example**	**Description**	**Attachment ("+" indicates presence)**		**Notes**	**Reference**
			Soft	Hard		
Large gorgonians	*Paragorgia arborea, Primnoa resedaeformis, Paramuricea* spp.*, Keratoisis spp.*	Arborescent or fan-shaped corals in the order Alcyonacea with a proteinaceous and/or calcareous inner axis (skeleton). Large gorgonians can attain heights >2 m.	+	++	Often fragmented in trawl surveys. Generally found attached to hard substrate, but the bamboo coral *Keratoisis flexibilus* can be found directly on soft substrate (e.g. Neves et al., 2015b), published as *Keratoisis* sp.).	From Gullage et al. (2022)
Small gorgonians	*Acanella arbuscula, Radicipes gracilis, Anthothela grandiflora*	Same as large gorgonians, but smaller in their adult stages (usually < 30 cm in height). This group is mainly represented by the bamboo coral *Acanella arbuscula* and the whip-like coral *Radicipes* spp.	++	+	Although *Radicipes* spp. and *Chrysogorgia* spp. can reach heights >30 cm, here they are grouped with the small gorgonians because they are delicate and do not form massive structures like large gorgonians. *Acanella arbuscula* is found directly on soft substrate, but the other small gorgonians are usually found attached to hard substrate.	From Gullage et al. (2022)
Soft corals	*Gersemia rubiformis, Duva florida, Drifa glomerata, Anthomastus* spp., *Heteropolypus* sp.	Corals in the order Alcyonacea without an inner axis. They have a soft body supported by a hydrostatic skeleton and small $CaCO_3$ structures (i.e. sclerites) embedded in their tissue. This group is mainly represented by the families Capnellidae and Alcyoniidae (mushroom corals), but includes delicate forms such the stoloniferous (creeping) *Clavularia* spp.	+	++	Generally found attached to hard substrate, but *Gersemia fruticosa* and *Heteropolypus* spp. can be found directly on soft substrate.	From Gullage et al. (2022)

Functional group	Species example	Description	Attachment ("+" indicates presence)		Notes	Reference
			Soft	Hard		
Sea pens	*Umbellula encrinus, Anthoptilum grandiflorum, Balticina finmarchica (= Halipteris finmarchica), Ptilella grandis (= Pennatula grandis), Pennatula aculeata, Funiculina quadrangularis*	Corals in the order Pennatulacea. Include both quill pen (e.g. *Pennatula* spp.), and whip-like morphologies (e.g. *Balticina* spp., *Protoptilum* spp.).	++	-	Mainly found on soft substrate. They are permanently partly buried in the sediment (i.e. peduncle). Some species can entirely withdraw into the substrate (e.g. *Pennatula aculeata*, Langton et al. 1990) following cues not yet completely understood, but with a potential to influence their catchability in trawls. At least one species in Pacific Canada can be found attached to hard substrate (*Anthoptilum* cf. *lithophilum*).	From Gullage et al. (2022)
Black corals	*Stauropathes arctica, Bathypathes* spp.	Corals in the order Antipatharia. They have a wire-like organic skeleton composed of concentric layers of protein and chitin. Colonies range in shape from branching (e.g. *Stauropthes* sp.), to feather-like (e.g. *Bathypathes* sp.) or whip-like (e.g. *Stichopathes* sp.) morphologies. Some species can exceed 1 m in height, but most are < 50 cm.	-	++	Less commonly found in trawl surveys in comparison to other corals.	From Gullage et al. (2022)
Reef-building corals	*Lophelia pertusa* (accepted name is *Desmophyllum pertusum*, but kept here as *Lophelia* due to the large literature on this coral using the old name)	Corals in the order Scleractinia which form true reefs.	-	++	The Maritimes region host the only known living *Lophelia pertusa* coral reef in Atlantic Canada. A dense debris of *L. pertusa* in the central Strait of Georgia (Pacific Region) was found in 2005 (Conway et al. 2007) suggesting that coral communities dominated this site for several centuries in the past and in 2021 a live *Lophelia* reef was found in Finlayson Channel, within the central coast waterways of British Columbia between 180 to 210 m (Cherisse Du Preez, pers. comm).	Conway et al. 2007; (P. Buhl-Mortensen et al., 2017); Beazley et al. (2021)

Functional group	**Species example**	**Description**	**Attachment ("+" indicates presence)**		**Notes**	**Reference**
			Soft	Hard		
Cup corals	*Flabellum alabastrum, Flabellum angulare, Desmophyllum dianthus, Vaughanella margaritata, Balanophyllia elegans*	Solitary corals in the order Scleractinia. They have a $CaCO_3$ skeleton and can be found free-living (unattached) on soft bottoms or attached to hard substrates. In Atlantic Canada this group is mainly represented by *Flabellum* spp., primarily *F. alabastrum*, a free-living species found on soft bottoms.	++	+	Individuals are small (usually < 5 cm in height) but can be found in aggregations. Other species included in this group are rare and/or are found infrequently in trawl surveys in Atlantic Canada (e.g. *Vaughanella* sp., *Javania* sp., *Fungiacyathus* sp.).	From Gullage et al. (2022)
Hydrocorals	*Stylaster* spp., *Errinopora* spp., *Stylantheca* spp.,	Corals in the order Anthoathecata (class Hydrozoa). They have $CaCO_3$ skeletons and can have branching or encrusting morphologies or form lamellate sheets. Colonies found in the Newfoundland and Pacific regions are usually branching. Also known as lace corals.	-	+	Rarely found in the trawl surveys in Atlantic (*Stylaster* spp. only), but common in Pacific Canada, where colonies can be found isolated or in high densities. Species observed in these regions have a branching morphology and are < 30 cm in height.	Boutillier et al. 2010; From Gullage et al. (2022)
Glass Sponge Reef species	*Aphrocallistes vastus, Heterochone calyx, Farrea occa*, and non-structure forming species	Globally unique. Formed through centuries of growth atop fused silicious sponge spicule framework which baffles sediment, forming large bioherms.	+	++	Although reefs are formed by three species of sponge, several other non-reef forming sponges are present on and near the reef structures. Only present in the Pacific region.	Krautter et al., 2001
Hexactinellid -*Vazella* sponge grounds	*Vazella pourtalesii*	*Vazella* can form monospecific assemblages but can also be present in multi-species sponge grounds.	-	++	Only recorded in the Maritimes region	Beazley et al. (2018)

Functional group	**Species example**	**Description**	**Attachment ("+" indicates presence)**		**Notes**	**Reference**
			Soft	Hard		
Astrophorid sponge grounds (e.g. *Geodia*)	Genera *Geodia*, *Stryphnus*, *Stelletta*, and *Thenea* spp.	Dense aggregations of astrophorid demosponges are common at lower shelf, bathyal, and/or abyssal depths. These aggregations often also incorporate a mixture of other sponges. The fauna is dominated by large species of the genus *Geodia*, with additional astrophorids in the genera *Stryphnus, Stelletta*, and *Thenea*.	++	+	Reach large biomass in the slope of Newfoundland and eastern Arctic	Maldonado et al. (2017)
Others: mixed sponges	Various	Includes sponges that have been systematically recorded in the scientific surveys with underwater camera systems and in multispecies trawl surveys, but not included in the previous groups.	+	+	Most sponges are difficult to identify from video and identification from multispecies surveys has not been done systematically and many previous sponge records are at a low taxonomic resolution level. However, with the progress of sponge knowledge across regions in Canada, more sponge functional groups associated with common environmental conditions can be defined in the future.	(Beazley et al., 2019; Dinn et al., 2019; Hawkes et al., 2019)

Table 3. Linking known indirect BCBs with coral and sponge groups and OECMs.

Functional group/ group	**Indirect BCB**			
	Biogeochemical cycling		**Habitat provision**	**Predator Prey Dynamics**
	Nutrient cycling	**Bioturbation**	-	-
Large gorgonians	- Increased nutrient fluxes (Pierrejean et al. 2020)	- Colonies of the bamboo coral *Keratoisis flexibilus* are anchored directly on soft sediment (e.g. Neves et al. 2014)	- Invertebrates (Mortensen and Buhl-Mortensen, 2005; De Clippele et al., 2015) - Invertebrates and fish (Neves et al., 2015b; Pierrejean et al. 2020) - Northern shrimp (Hiltz et al., 2018) - Sponges (Dinn et al., 2020a)	- The sea stars *Hippasteria phrygiana* feeds on large gorgonians. (http://echinoblog.blogspot.com/2013/07/who-eats-who-figuring-out-feeding-in.html), Krieger and Wing (2002)
Small gorgonians	- Suspension feeding (Gili and Coma, 1998; Sherwood et al., 2008; Gasbarro et al., 2018)	- Colonies of the bamboo coral *Acanella arbuscula* are usually anchored directly on soft sediment (Deichmann, 1936)	- Anemones, amphipods and hydroids (Buhl-Mortensen et al., 2010)	-

Functional group/ group	**Indirect BCB**			
	Biogeochemical cycling		**Habitat provision**	**Predator Prey Dynamics**
	Nutrient cycling	**Bioturbation**	-	-
Soft corals	- Suspension feeding (Gili and Coma, 1998, Sherwood et al., 2008; Gasbarro et al., 2018) - Deposit-feeding (Slattery et al., 1997)	- Soft corals can live directly on soft sediments (Slattery et al., 1997) (1)	- Juvenile basket stars (Neves et al., 2020)	- The sea star *H. phrygiana* feeds on soft corals (Gale, et al., 2013)
Sea pens	- Suspension feeding (Gili and Coma, 1998; Sherwood et al., 2008; Baillon et al., 2016; Gasbarro et al., 2018)	(OSPAR, 2010) (1)	- Small planktonic and benthic invertebrates (Birkeland, 1974; Baillon et al., 2016) - Larvae red fish (Baillon et al., 2012) - Squat lobster habitat (De Clippele et al., 2015) (1)	- The sea star *H. phrygiana* feeds on sea pens (Gale et al. 2013)
Reef corals	Carbon sink (buffer for climate change effects and ocean acidification) (Maier et al., 2009)	-	- Invertebrates (Buhl-Mortensen et al., 2017b) - Fish (Husebø et al., 2002; Costello et al., 2005) (1)	-

Functional group/ group	**Indirect BCB**			
	Biogeochemical cycling		**Habitat provision**	**Predator Prey Dynamics**
	Nutrient cycling	**Bioturbation**	-	-
Black corals	Suspension feeding (zooplankton is major component of their diet) (Gili and Coma, 1998; Wagner et al., 2012) (2)	-	- Invertebrates (Wagner et al., 2012)	-
Cup corals	- Carbon sink (buffer for climate change effects and ocean acidification (Martínez-Dios et al., 2020)	- Several cup coral species live directly on soft sediment (e.g. (*Flabellum alabastrum*), indicating the potential for bioturbation activities. - At least on species of solitary coral is known to display a burrowing behavior (Sentoku et al. 2016), which might be identified in other species.		- Cup corals such as *Flabellum alabastrum* have been shown to have a carnivorous diet, in comparison to colonial corals with smaller polyps, many of which feed on particulate organic matter (Sherwood et al. 2008).
Hydrocorals	- Suspension feeding (Gili and Coma, 1998, Imbs et al., 2016; Gasbarro et al., 2018) Carbon sink (buffer for climate change effects and ocean acidification (Lewin 2006)	-	- Invertebrates (Stone, 2006), Etnoyer and Morgan (2006), Häussermann and Försterra (2007), Gomes-	-

Functional group/ group	**Indirect BCB**			
	Biogeochemical cycling		**Habitat provision**	**Predator Prey Dynamics**
	Nutrient cycling	**Bioturbation**	-	-
			Pereira et al. (2017) (2)	
Mixed sponges	- Can be assumed to be similar to information listed for other sponges.	-	- Invertebrates (Beazley and Kenchington, 2015) - Cephalopod nursery (Hayes et al., 2010)	- The sea star *Ceramaster granularis* feeds on sponges (Gale et al., 2013)
Astrophorid sponges	- Large carbon consumption (Pham et al., 2019)	-	- Increase macrofauna abundance and diversity (Ashford et al., 2019)	-
Glass sponge reefs	- Nutrient cycling (capture of energy for food web) (Kahn et al., 2016) - Carbon sink (buffer for climate change effects and ocean acidification) (Kahn et al., 2015). - Excretion of ammonia waste (contributes to primary production) (Kahn et al., 2015). - Silicon sink (contributes to diatom primary productivity) (Chu et al., 2011). -Suspension feeding (Gili and Coma, 1998; Dunham et al., 2018).	-	- Juvenile rockfish (Chu and Leys, 2010) and adult rockfish (Marliave et al., 2009). - Longhorn decorator crab habitat (Chu and Leys, 2010). -Squat lobster habitat (Chu	- The nudibranch (*Peltodoris lentiginosa*) preys on glass sponges (Chu and Leys, 2012). - The annelid, *Procerea* sp., has been reported to prey on glass sponges, also crabs, nudibranchs, sea stars, squat lobsters and spot prawns (Archer et al., 2020a).

Functional group/ group	Indirect BCB			
	Biogeochemical cycling		Habitat provision	Predator Prey Dynamics
	Nutrient cycling	Bioturbation	-	-
			and Leys, 2010) -Lithodid (King) crab habitat (Chu and Leys, 2010). - Spot prawn habitat (Cook et al., 2008). - Blood star habitat (Cook et al., 2008). - Ratfish habitat (Cook et al., 2008) - Association with polychaetes. (Archer et al., 2020a). - Association with Nemertea (Archer et al., 2020a). - Association with	

Functional group/ group	Indirect BCB			
	Biogeochemical cycling		Habitat provision	Predator Prey Dynamics
	Nutrient cycling	Bioturbation	-	-
			Platyhelminthes (Archer et al., 2020a).	
Vazella sponge grounds	- Silicon sink (modulates ocean primary productivity) (Maldonado et al., 2020)	-	- Squat lobster habitat (Hawkes et al., 2019)	-

Blanks in each group-indirect BCB combination do not necessarily mean absence of a link. As research continues, it is likely that new links will be identified and that currently known links for specific groups and regions might be further expanded.

[1] References from other areas involving similar species to those found in Canada.

[2] References from other areas not involving species found in Canada.

Table 4. Indicator selection criteria in relation to state and stressor indicators that could be potentially used in the monitoring of Canadian coral and/or sponge OECMs.

	State indicators												Stressor indicators									
Criterion	Abundance	Biomass	Distribution	Diversity indices	Size structure	Live:Dead ratio	% corals with zoanthids	Patch area and density	Patch isolation/proximity	Patch connectivity	Patch contagion index	*Lophelia* extent	Fishing Distrib./aggregation	Areas not impacted by bottom-contact gears	Distribution of oil and gas	Anthropogenic sediment	Chemical impact oil and gas	Anomalous events	Phytoplankton bloom	Sea ice cover	Seabed litter	Submarine cables
Theoretical basis	X	X	X	X	X	X	X	-	-	-	-	-	X	X	X	X	X	-	X	X	X	-
Measurement	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	?	X	X	X	?
Historical data	-	X	X	-	-	-	-	-	-	-	-	?	X	X	X	-	-	-	X	X	-	-
Sensitivity	X	X	X	X	X	X	-	X	X	X	X	X	X	X	-	X	X[1]	-	?	X[2]	X	X[3]
Responsiveness	X	X	X	X	X	X	-	X	X	X	X	X	X	X	-		X[1]	-	X	X[2]	-	X[3]
Specificity	X	X	-	X	X	X	-	X	X	X	X	X	-	X	-	X	X[1]	X	-	X[2]	X	-

Criteria based on DFO network monitoring indicators (DFO, 2013). Specific sponge reef indicators were not included here because they have already been addressed (Dunham et al., 2018a). Environmental and indirect BCBs were not included because they represent different parameters, which might respond differently. The responsiveness of a certain criterion is represented by an “X”. Nulls do not necessarily mean that a criterion does not apply (i.e. it could represent knowledge gaps). Question marks represent a potential for an indicator to respond to a criterion, but for which additional information is needed.

[1] In relation to oil and gas pressures.
[2] In relation to climate change.
[3] In relation to itself.

Table 5. Summary of suitable state indicators that could be potentially used in the monitoring of Canadian corals and/or sponge OECMs.

Indicator	**Gorgonians**	**Sea pens**	**Soft corals**	**Cup corals**	**Black corals**	**Reef–forming corals**	**Glass sponges (reef)**	**Glass sponges (non-reef)**	**Astrophorid sponges (e.g. *Geodia*)**	**Miscellaneous sponges**	**Purpose/strengths**	**Limitations**	**Preferred tools**
Abundance	x	x	x	x	x	x[1]	x[2]	x	x	x	- Biodiversity - Function - Reproductive success - Easy to measure	- Small specimens might be overlooked	- Imagery surveys
Biomass	x	x	x	x	x	-	-	x[3]	x	x	- Ecological function - Reproductive success - Identification of hotspots of diversity - Can use scientific trawl data - Direct weight is easy to measure	-[3]Biomass for glass sponges can be misleading, as they are very light. - Would need calibration of size-weight relationships if using imagery	- Imagery surveys - Scientific trawls (if surveys continue inside of OECMs)
Distribution	x	x	x	x	x	x	x[4]	x	x	x	- Ecosystem resilience - Ecosystem function - Genetic diversity - Can use scientific trawl data	- Depends on good taxonomic resolution	- Imagery surveys associated with sampling
Diversity indices	x	x	x	x	x	x	x	x	x	x	- Biodiversity - Community structure	- Depend on quality of	- Imagery surveys associated with sampling

Indicator	**Gorgonians**	**Sea pens**	**Soft corals**	**Cup corals**	**Black corals**	**Reef–forming corals**	**Glass sponges (reef)**	**Glass sponges (non-reef)**	**Astrophorid sponges (e.g. *Geodia*)**	**Miscellaneous sponges**	**Purpose/strengths**	**Limitations**	**Preferred tools**
											- Ecosystem resilience - Ecosystem function - Genetic diversity	abundance and richness data	
Size structure	x	x	x	x	x	-	-	x	x	x	- Ecological function - Reproductive success	- Limited for organisms with vertical bodies. - Difficult to measure from video. - Trawl samples might be size-biased.	- Imagery surveys associated with samples
Live:dead ratio and condition	x	-	-	-	x	x	x	-	-	-	- Mortality rate - Physiological stress	-Limited for gorgonians due to difficulty to accurately determine the number of dead colonies (2 fragments equal one or two colonies). - Clear visual contrast between living and dead portions of	- Imagery surveys - Physical samples (fitness assessments)

Indicator	**Gorgonians**	**Sea pens**	**Soft corals**	**Cup corals**	**Black corals**	**Reef–forming corals**	**Glass sponges (reef)**	**Glass sponges (non-reef)**	**Astrophorid sponges (e.g. *Geodia*)**	**Miscellaneous sponges**	**Purpose/strengths**	**Limitations**	**Preferred tools**
												Lophelia colonies and sponge reefs	
% corals with zoanthids	x	-	-	-	x	-	-	-	-	-	- Physiological stress	- Difficult to measure - Need target-imagery	- Imagery surveys (ROV preferred)
Patch area and density	x	x	x	x	-	-	-	x	x	x	- Biodiversity - Ecological function - Reproductive success	- Need clear definition of patch by functional group	- Imagery surveys
Patch isolation/proximity	x	x	x	x	-	-	-	x	x	x	- Reproductive success - Genetic diversity	- Need clear definition of patch by functional group	- Imagery surveys
Patch connectivity	x	x	x	x	-	-	-	x	x	x	- Reproductive success - Genetic diversity	- Need clear definition of patch by functional group	- Imagery surveys associated with sampling (genetic studies)
Patch contagion index	x	x	x	x	-	-	x	x	x	x	- Reproductive success	- Need clear definition of patch by functional group	- Imagery surveys associated with sampling (genetic studies)
Lophelia reef extent	-	-	-	-	-	x	-	-	-	-	- Provides information on reef extent - Can be used to assess reef physical	- Lack of data on actual reef extent	- Imagery surveys - Acoustic surveys

Indicator	**Gorgonians**	**Sea pens**	**Soft corals**	**Cup corals**	**Black corals**	**Reef–forming corals**	**Glass sponges (reef)**	**Glass sponges (non-reef)**	**Astrophorid sponges (e.g. *Geodia*)**	**Miscellaneous sponges**	**Purpose/strengths**	**Limitations**	**Preferred tools**
											damage (e.g. broken reef)		
Indicator taxa of live sponge reef	-	-	-	-	-	?	x	-	-	-	- Certain taxa have significant associations with specific habitat types and their presence can indicate reef status		- Imagery surveys (ROV preferred)
Reef structure habitat categories (no visible reef, dead reef, mixed reef, live reef)	-	-	-	-	-	x	x	-	-	-	- Relative proportions of these four habitat categories		- Imagery surveys
Recovery potential	-	-	-	-	-	x	x	-	-	-	- Recolonization and regrowth - Dead % cover - % visible habitat categories combined	- Recruits can be difficult to visualize	- Imagery surveys associated with sampling
Indirect BCBs	x	x	x	x	x	x	x	x	x	x	- Species associations (e.g. redfish and sea pens) - Infauna diversity - Oscula density and area (i.e. sponge filtration rate proxies)		- Imagery data (sponge oscula metrics), species associations - Sediment sampling (sediment grab/core, ROV

Indicator	**Gorgonians**	**Sea pens**	**Soft corals**	**Cup corals**	**Black corals**	**Reef–forming corals**	**Glass sponges (reef)**	**Glass sponges (non-reef)**	**Astrophorid sponges (e.g. *Geodia*)**	**Miscellaneous sponges**	**Purpose/strengths**	**Limitations**	**Preferred tools**
											- Contribution to biogeochemical cycles		push-core preferred)
Environmental indicators	-	-	-	-	-	-	-	-	-	-	- Collection of environmental data to assist with interpretation of changes	- No significant limitations	- Oceanographic sampling (e.g. CTD casts, plankton tows) - Satellite data

Corals and sponges are listed by functional groups. Criteria based on DFO (2013), indicators based on literature search (described in the text). List of all considered indicators is displayed in Table 4.

[1] When colonies can be distinguished from one another.
[2] Measured as percent cover.
[3] Biomass for glass sponge samples can be misleading, as they are very light.
[4] Live sponges.

Table 6. Summary of tools used in seafloor surveys which can be used in monitoring programs.

Tool	Data type	Data target	Spatial coverage	Environments
ROV	Imagery, samples, acoustic	Epifauna (infauna if push-core available)	Continuous	All
Mini-ROV	Imagery, samples[1]	Epifauna	Continuous	All
Human-occupied sub.	Imagery, samples	Epifauna	Continuous	All
AUV	Imagery, acoustic	Seafloor, epifauna	Continuous	All
Drop camera	Imagery	Epifauna	Point	Flat-bottom areas
TUVS	Imagery	Epifauna	Transect	Flat-bottom areas
BRUVS	Imagery	Demersal fish, epifauna	Point (mostly qualitative for sessile fauna)	Flat-bottom areas
Trawls/dredges	Samples (biological and sediment)	Megafauna, epifauna	Transect (qualitative)	Flat-bottom areas
Sed. samplers	Samples (biological and sediment)	Macrofauna, infauna	Point	Unconsolidated sediment areas
MBES	Bathymetry, backscatter	Seafloor	Continuous	All
eDNA	Samples (water and sediment)	DNA	Point	Water column, unconsolidated- mixed sediment areas
Moorings	Oceanographic, samples[2]	Physical, sampling[2]	Point	Flat-bottom areas
Benthic landers	Imagery, samples[1], acoustic	Physical, sampling	Point	Flat-bottom areas
Hydrophones	Acoustic	Fauna	Point	All

Adapted from Przeslwaski and Foster (2018). Sed. Samplers = sediment samplers (e.g. cores). MBES = Multibeam echosounder.

[1] Limited.
[2] For instance, sediment traps.

Table 7. Characteristics of current technologies suitable for benthic surveys.

	Imagery							Bottom-contact			Other tools				
Characteristic	**ROV**	**Mini ROV**	**Human-occupied sub**	**AUV**	**Drop camera**	**TUV**	**BRUV**	**Sled/ Trawl**	**Dredges**	**Grabs, corers**	**MBES**	**eDNA**	**Moorings**	**Benthic lander**	**Hydrophones**
Continuous broad-scale spatial coverage	-	-	-	-	-	-	-	-	-	-	X	-	-	-	-
Continuous fine-scale spatial coverage	X	X	X	X	-	-	-	-	-	-	-	-	X	-	-
Non-extractive	X	X	X	X	X[1]	X[1]	X	-	-	-	X	X	X	X	X
Repeatability	X	-	X	X	-	-	-	-	-	-	X	X	X	-	X
Able to sample over a variety of environments	X	X	X	X	X[2]	X[2]	X	-	X	-	X	X	-	X	X
Species-level identifications (i.e. sampling possible)	X	-	X	-	-	-	-	X	X	X	-	X	-	-	X
Genetics, morphological analyses possible	X	-	X	-	-	-	-	X	X	X	-	X	-	-	-
Behavior observed	X	X	X	-	X	X	X	-	-	-	-	-	-	X	X
Cryptofauna observed	X	X	X	-	-	-	-	X	X	X	-	-	-	-	X
Quantitative data	X	X	X	X	X	X	X	-	-	X	X	X	-	X	X
Concurrent physical and biological data	X	-	X	X	-	-	-	X[3]	-	X	-	X	X	X	X
Minimal technical expertise	-	X	-	-	X	X	X	X	X	X	-	-	-	-	-
Vessel flexibility	-	X	-	-	X	X	X	-	-	X	X	X	-	X	X
Access to equipment (easiness)	-	X	-	-	X	X	X	X	X	X	X	X	-	-	X
Cost (not considering ship time)	$$$	$	$$$	$$$	$$	$$	$$	$	$	$	$$	$	$$	$$$	$

The occurrence of a certain characteristic is represented by an "X", except for cost, where dollar signs were used to display costs ranging from low to high ($-$$$). Adapted from Przeslwaski and Foster (2018).

Drop cameras can be invasive when deployed blindly; Limited in areas of high vertical relief; Can be associated with CTD (but not common).

Table 8. Definition of specific terms relates to sampling design.

Terms	Definition
Sampling units	Sampling units can be considered as individual ‘items’ which provide measurements of a particular variable, attribute or characteristic.
Sample	Set of sampling units about which generalized conclusions can be drawn about the population by inference.
Precision	The degree of concordance among a number of measurements or estimates for the same population precision is reflected by the variability of an estimate.
Accuracy	The closeness of a measurement or estimate to the true value of the population, as related to the bias and precision of the measurement.
Statistical power (1-β)	Probability that a test correctly rejects the null hypothesis when it is false, and is derived base on the statistical significance level (α), the magnitude of the effect size (ES), sample size (N) and parameter variability (σ).
Spatial autocorrelation	Pattern in which observations from nearby locations are likely to have a similar value than expected due to chance alone than locations that are farther apart. This can result in erroneous inference if not accounted for in analysis.
Serial correlation	The correlation of a variable with itself across different points in a time series.
Pseudoreplication	Inflation of the number of data points through non independence. Can be avoided by proper sampling design.
Probabilistic sampling designs	Include simple random sampling, stratified random sampling, and systematic sampling. Typically, minimize systematic errors and are considered to be more statistically rigorous.
Non-probabilistic (judgement sampling) designs	Subjective selection of sampling units without any form of randomization. Care must be taken with appropriate statistical techniques and review of the data to make inferences about a wider population.

Adapted from Noble-James et al. 2018.

9. FIGURES

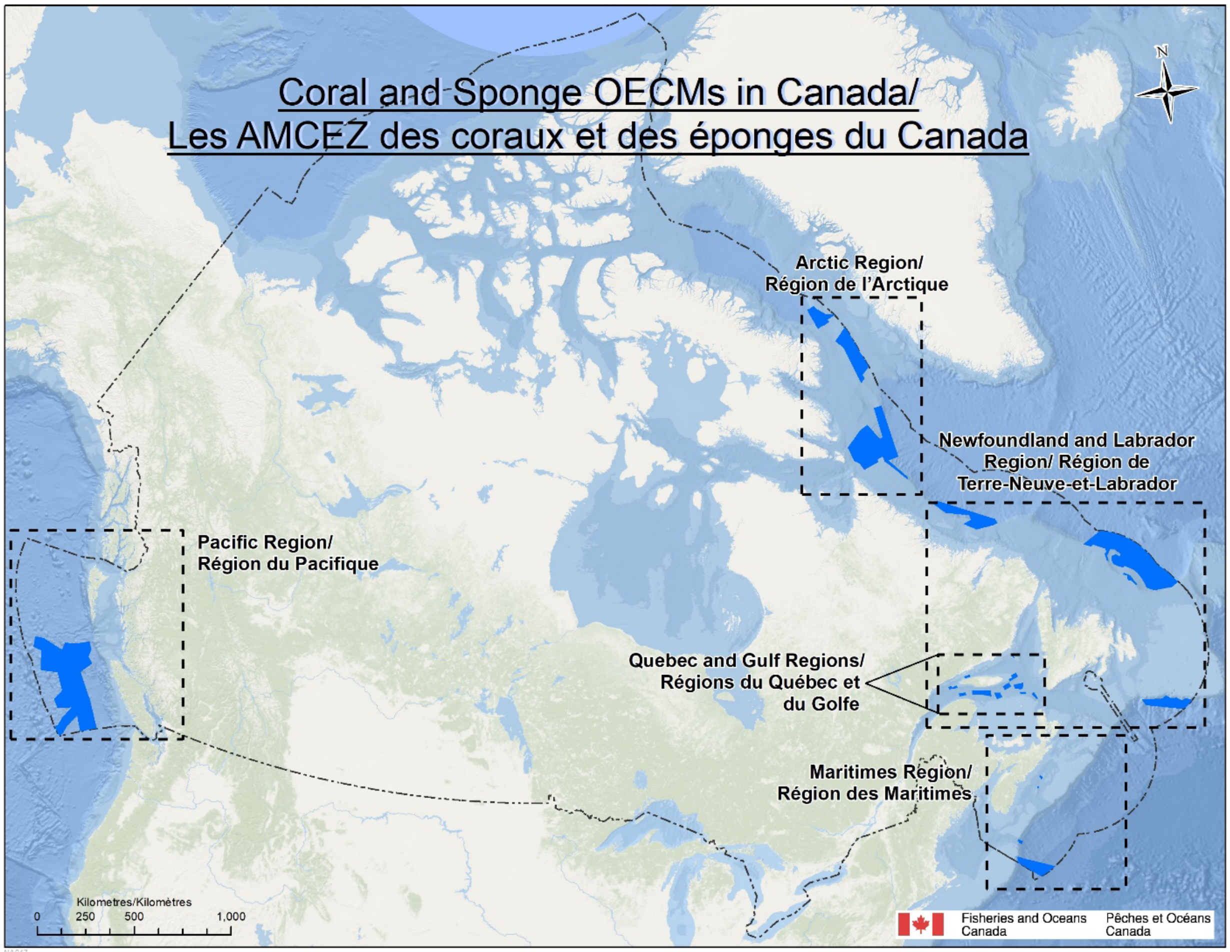

Figure 1. Location of Canada's cold-water coral and sponge OECMs. At the time of publication, Canada had 59 OECMs, 40 of which are established to protect cold-water corals and/or sponge benthic ecosystems.

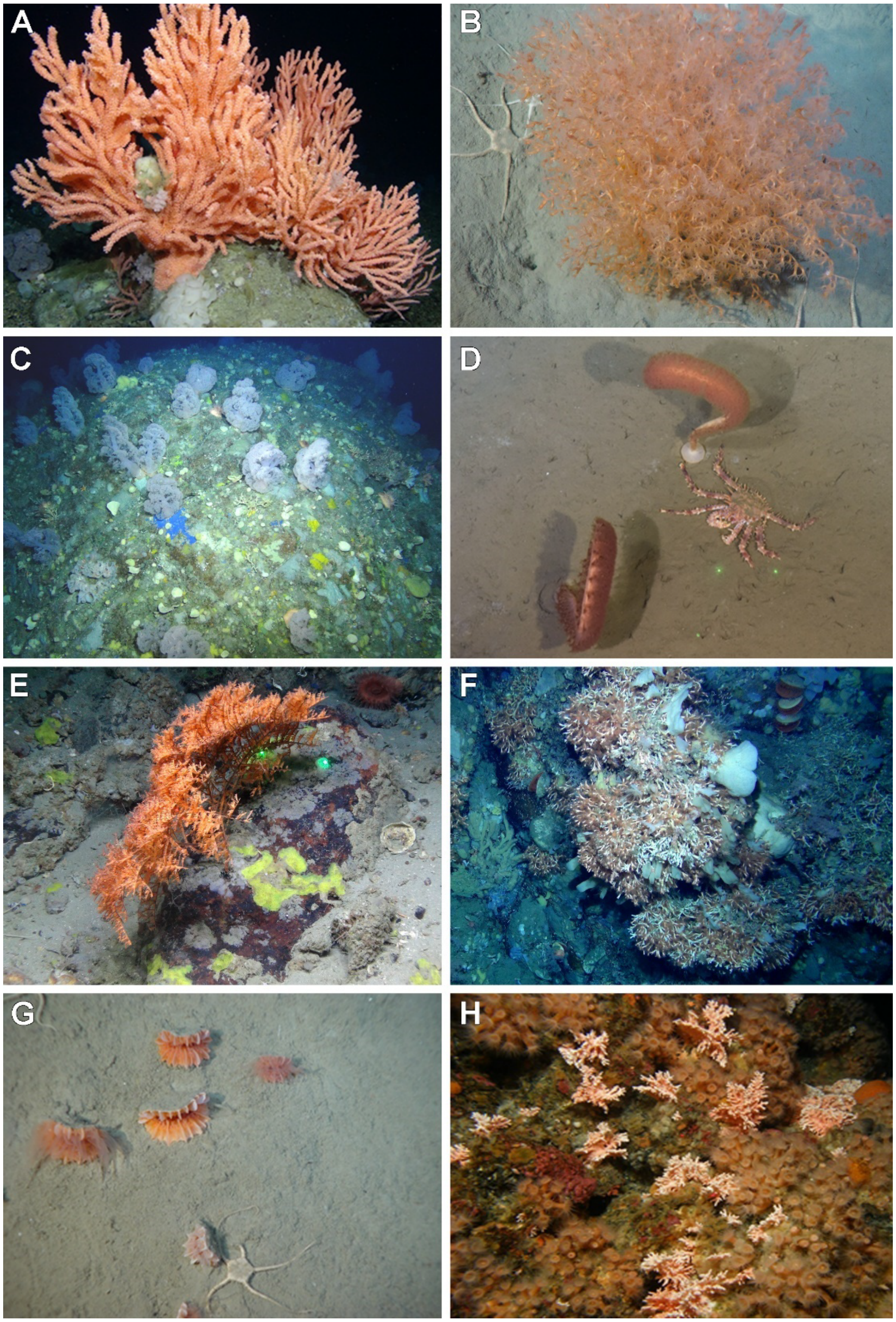

Figure 2. Coral groups for monitoring purposes. A. Large gorgonian corals, photo credit ArcticNet-CSSF-DFO. B. Small gorgonian coral, photo credit: DFO. C. Soft corals, photo credit: ArcticNet-CSSF-DFO. D. Sea pens, photo credit: DFO/Oceana Canada/CSSF. E. Black coral, photo credit: Amundsen Science 2021. F. Reef-Building corals, Photo credit: ArcticNet-CSSF-DFO. G. Cup corals, photo credit: DFO. H. Hydrocorals, photo credit DFO/NOAA.

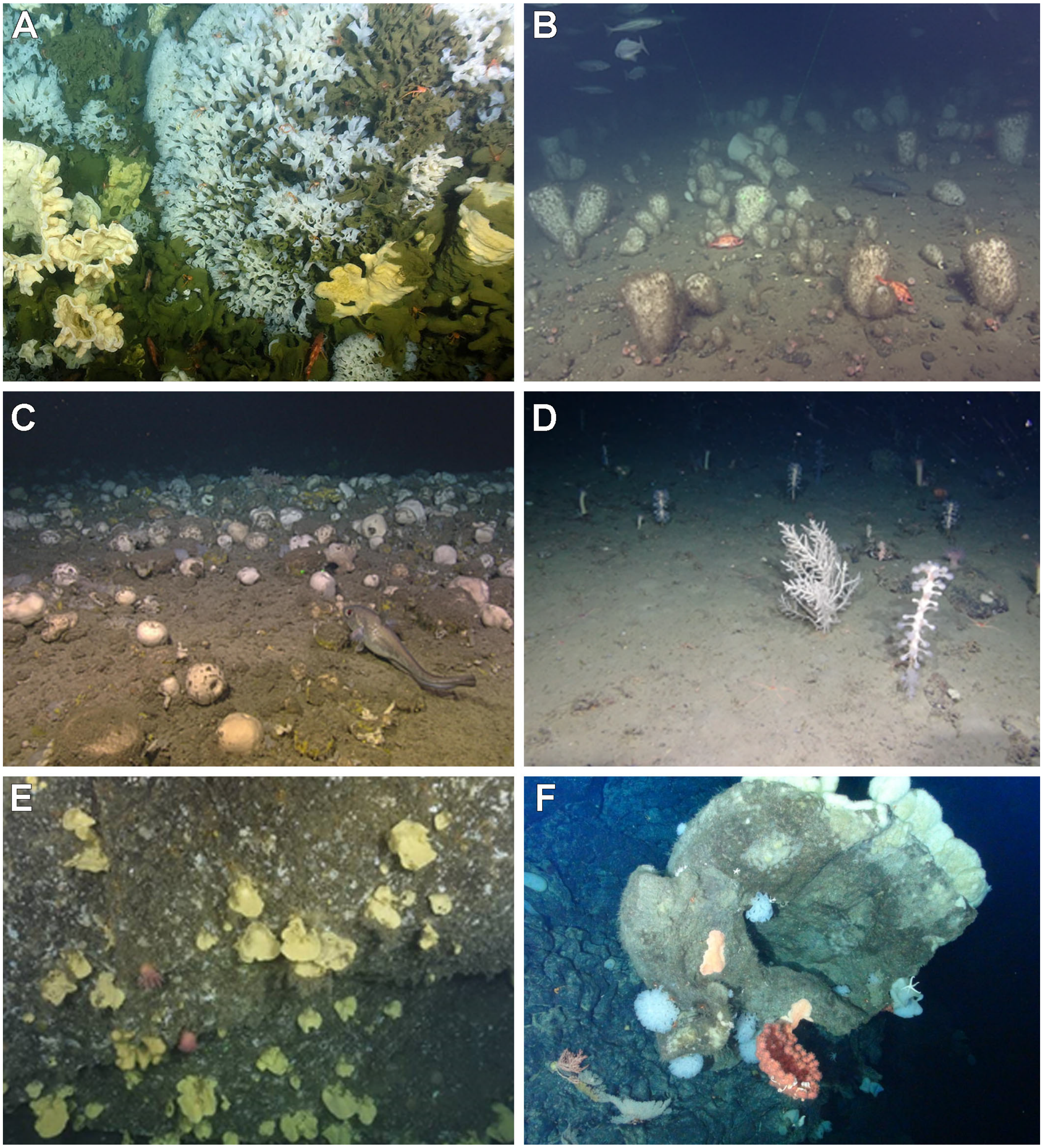

Figure 3. Sponge groups for monitoring purposes. A. Glass sponge reefs, photo credit: DFO/Sally Leys/University of Alberta/CSSF/ROPOS. B. Vazella *sponge grounds, photo credit: Beazley et al., 2018. C. Astrophorid sponge grounds, photo credit: DFO. D. Mixed sponges: carnivorous sponges in the eastern Canadian Arctic, photo credit: ArcticNet-CSSF-DFO. E. Mixed sponges:* Plicatellopsis bowerbanki *growing in the Laurentian Channel, photo credit: DFO/Oceana Canada/CSSF. F. Mixed sponges: Sponges on the Union Seamount in the Pacific, photo credit: DFO.*

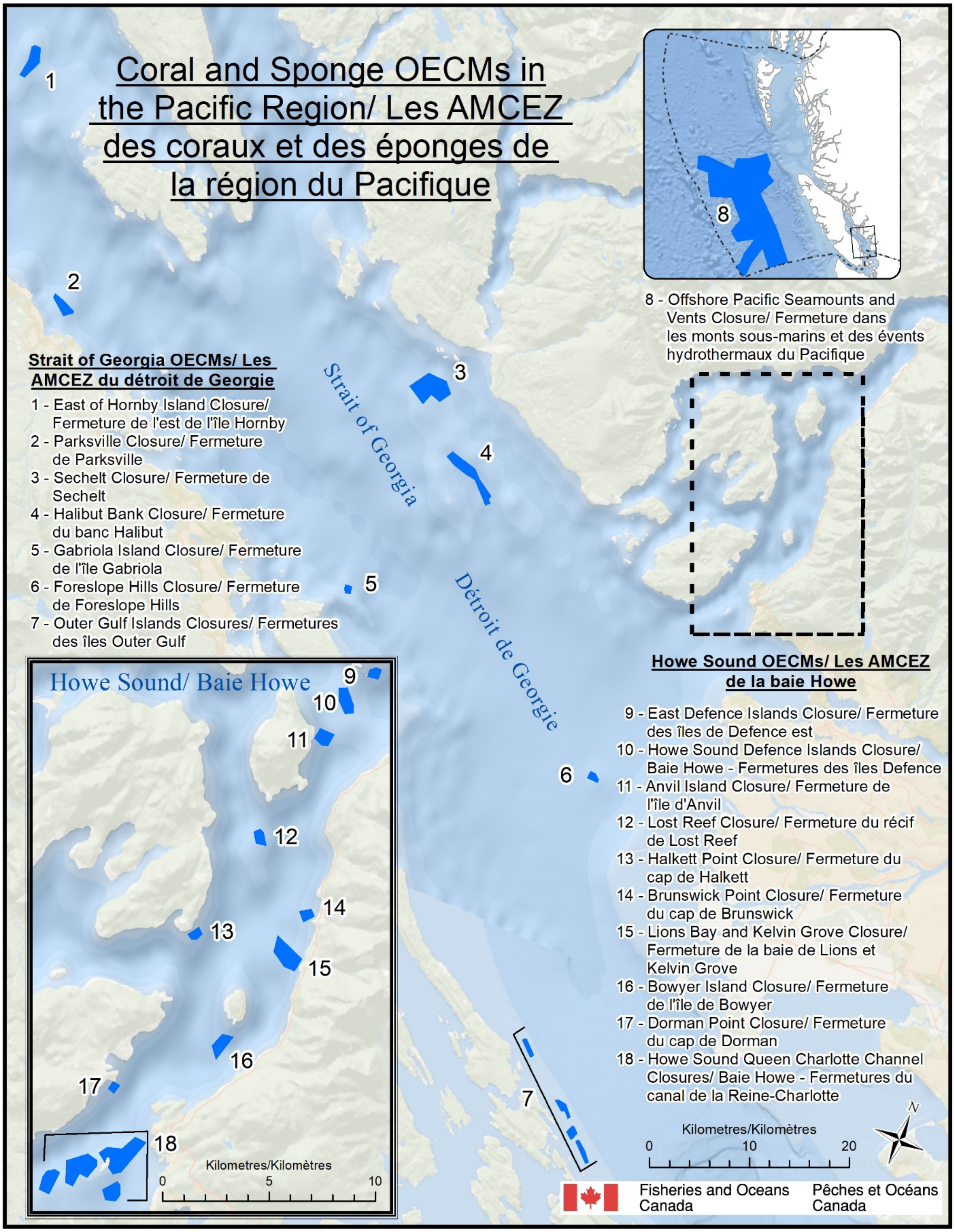

Figure 4. Location of the 17 coral and/or sponge-focused OECMs in the Pacific region.

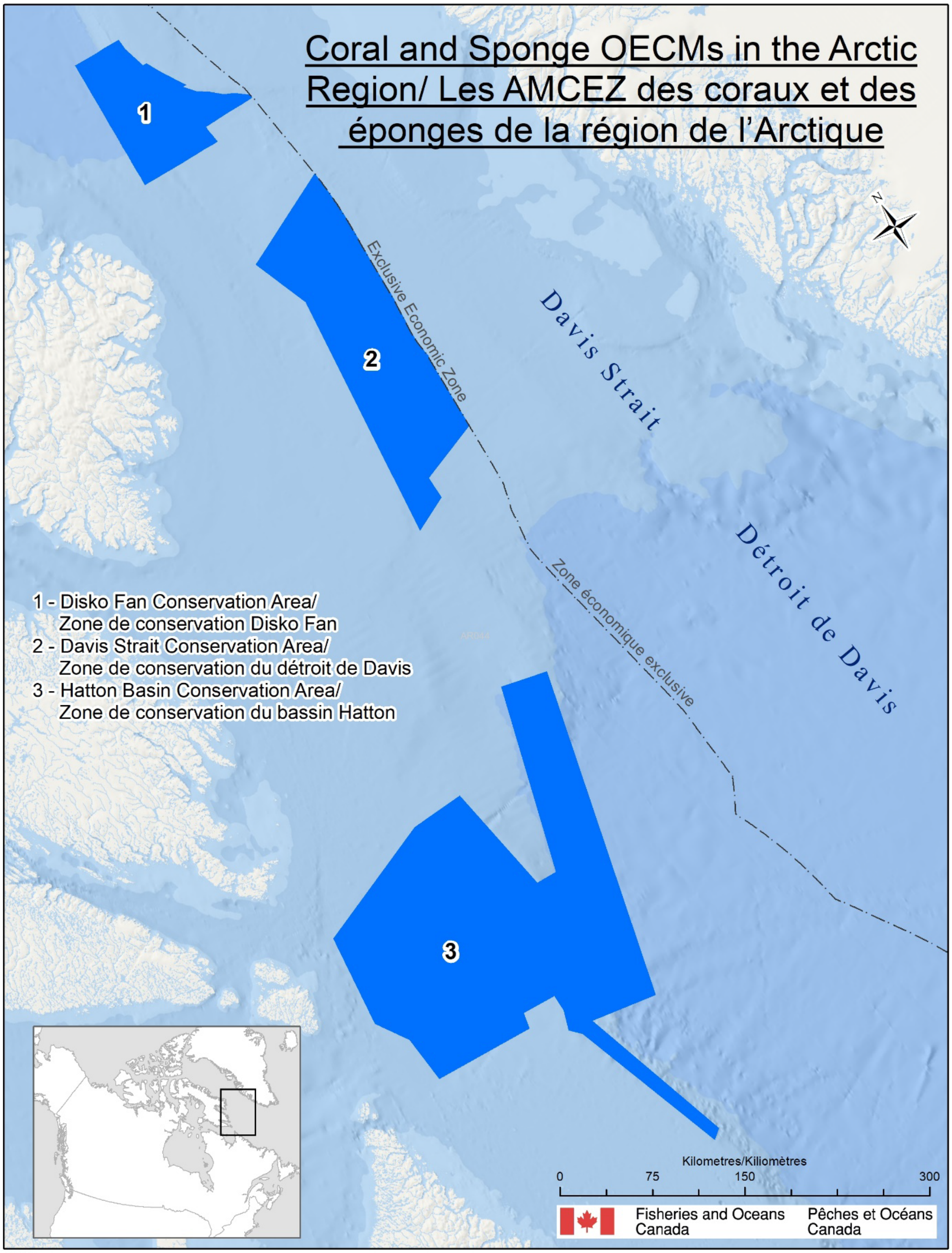

Figure 5. Location of the 3 coral and sponge-focused OECMs in the Arctic region.

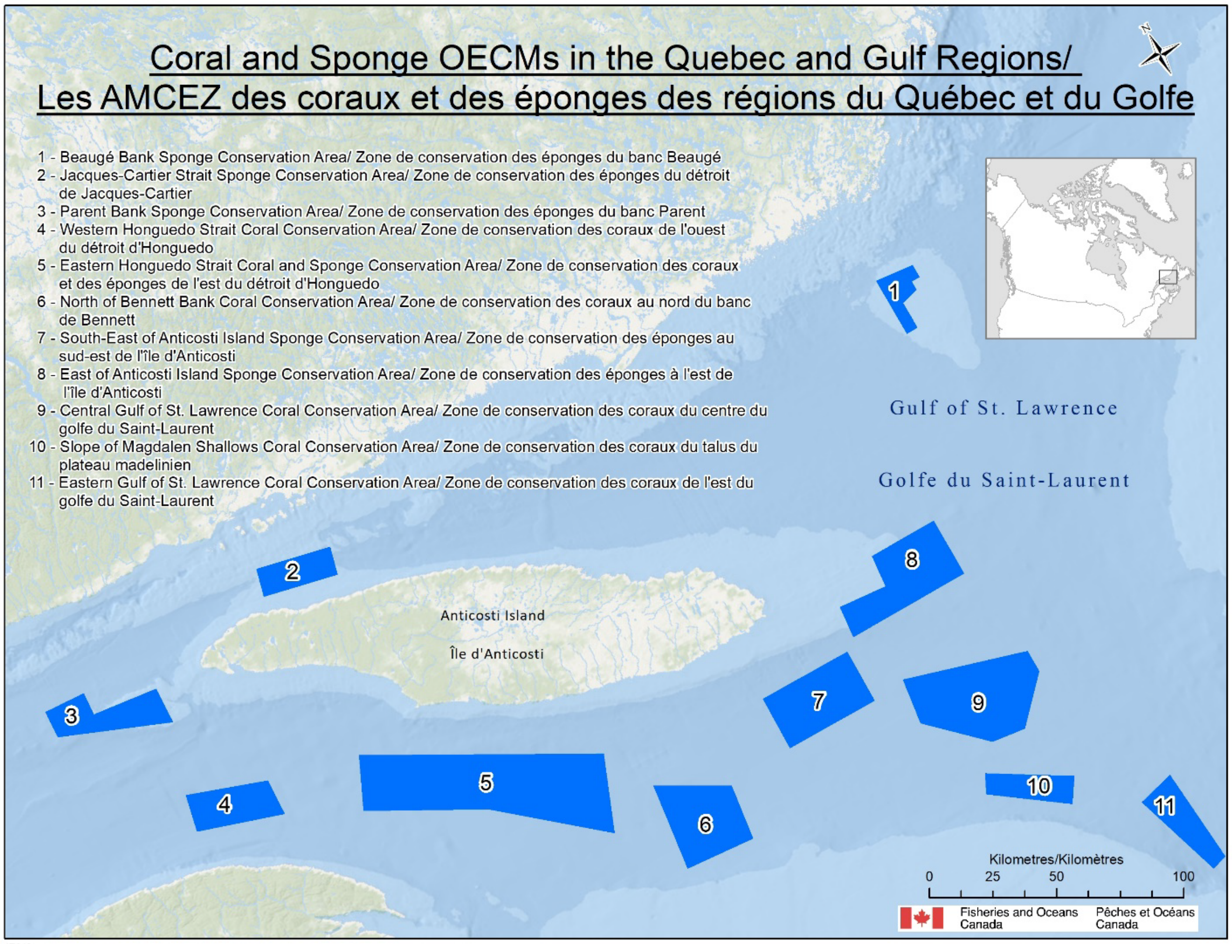

Figure 6. Location of the 11 coral and sponge-focused OECMs in the Gulf of St. Lawrence.

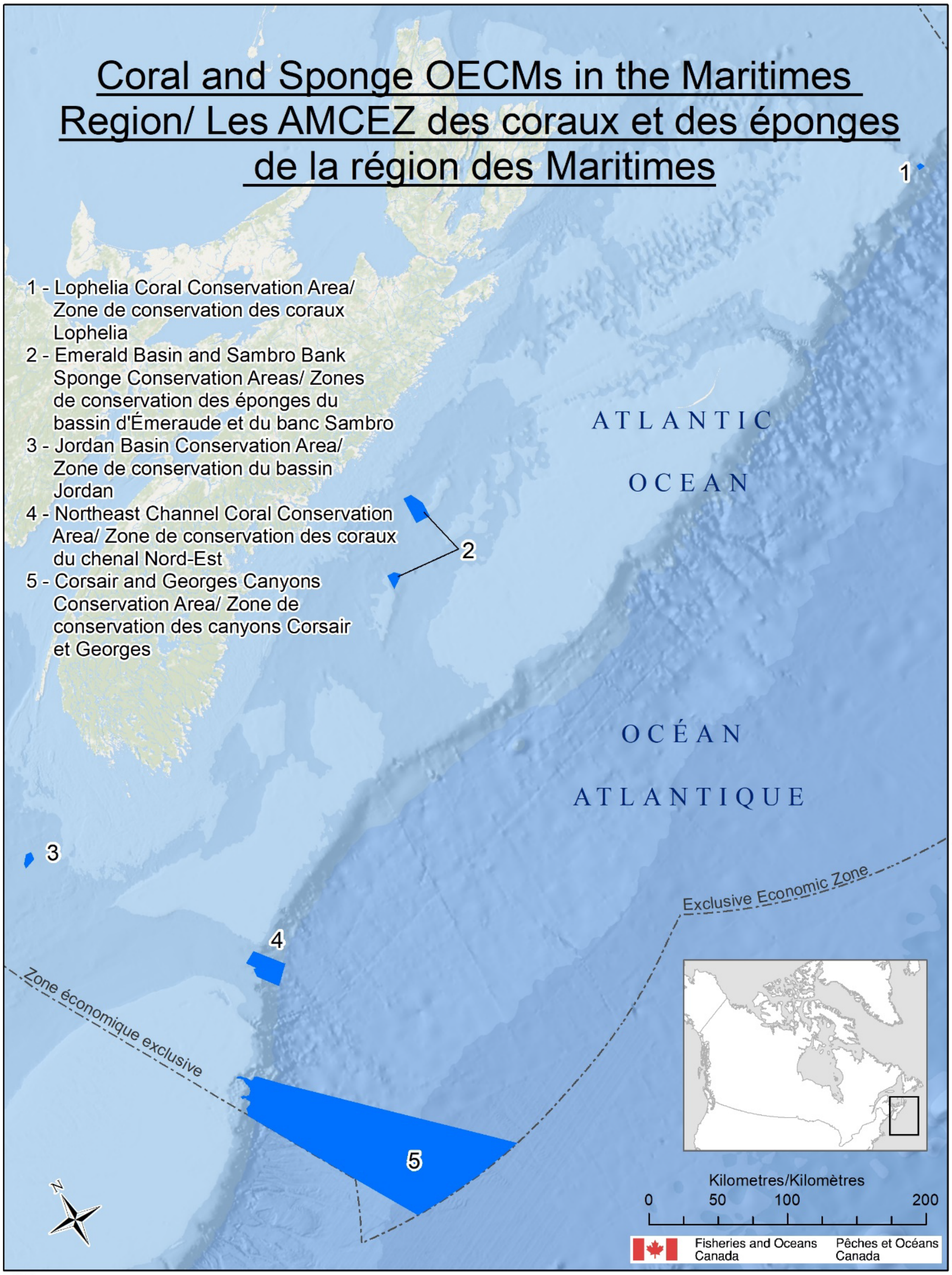

Figure 7. Location of the 5 coral and/or sponge-focused OECMs in the Maritimes region.

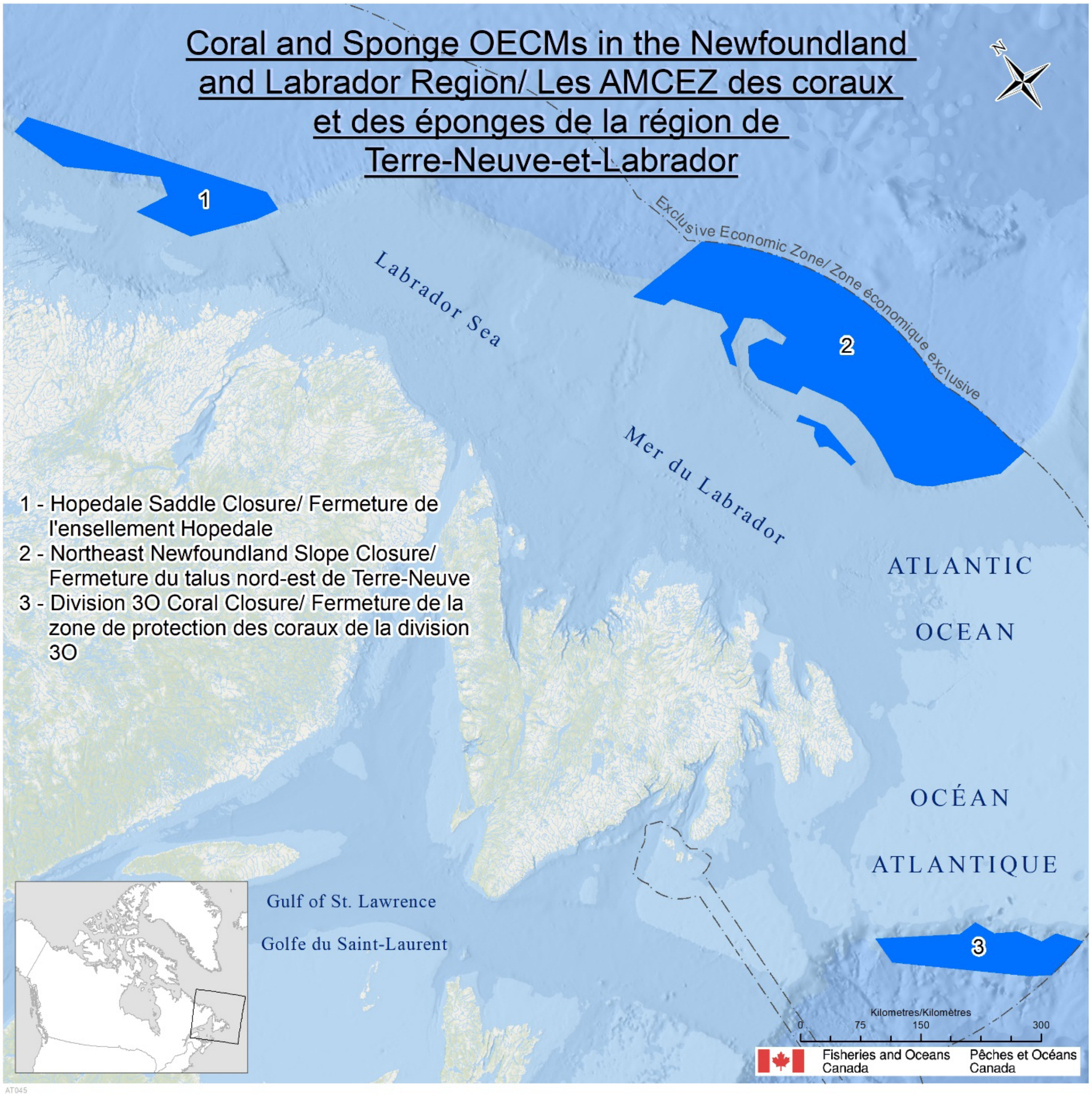

Figure 8. Location of the 3 coral and sponge-focused OECMs in the Newfoundland & Labrador region.

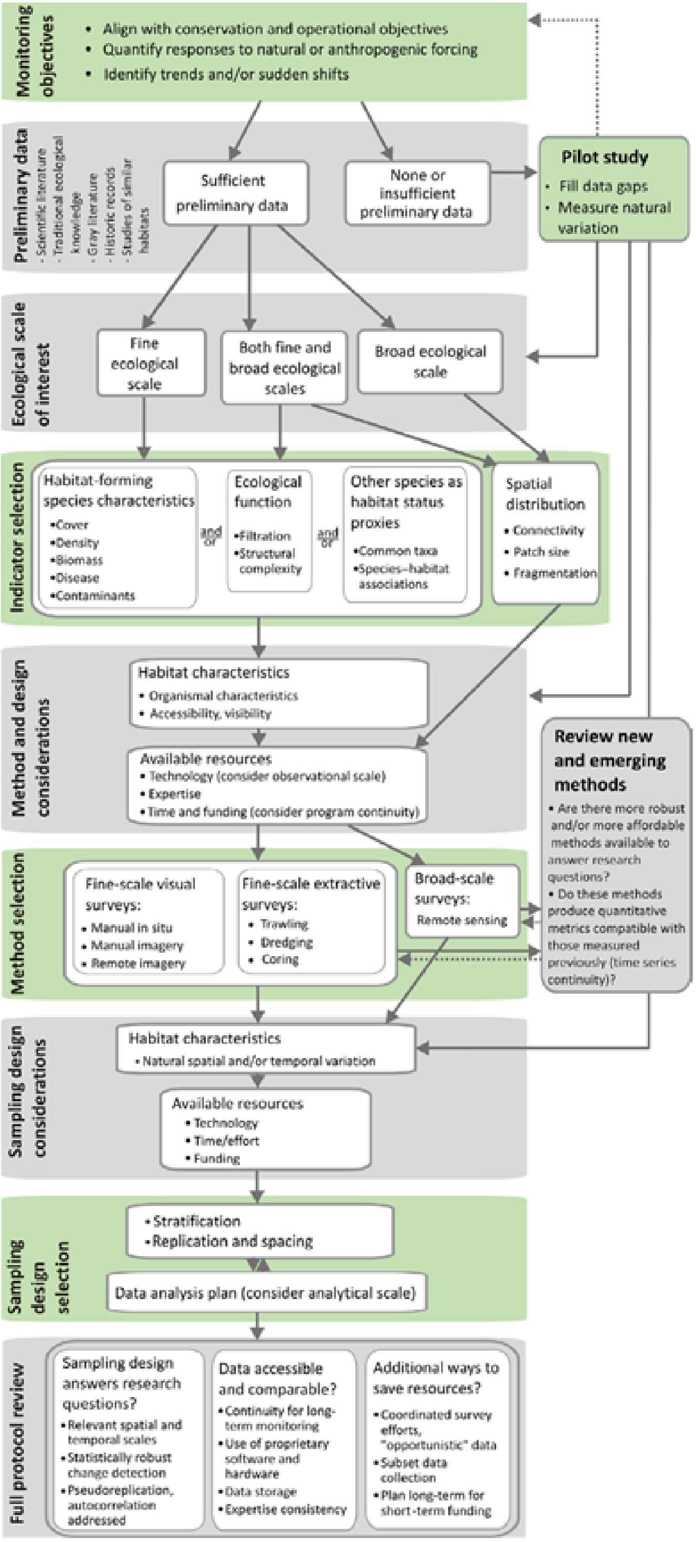

Figure 9. Flowchart illustrating main steps for developing a monitoring design. From Loh et al. 2019.

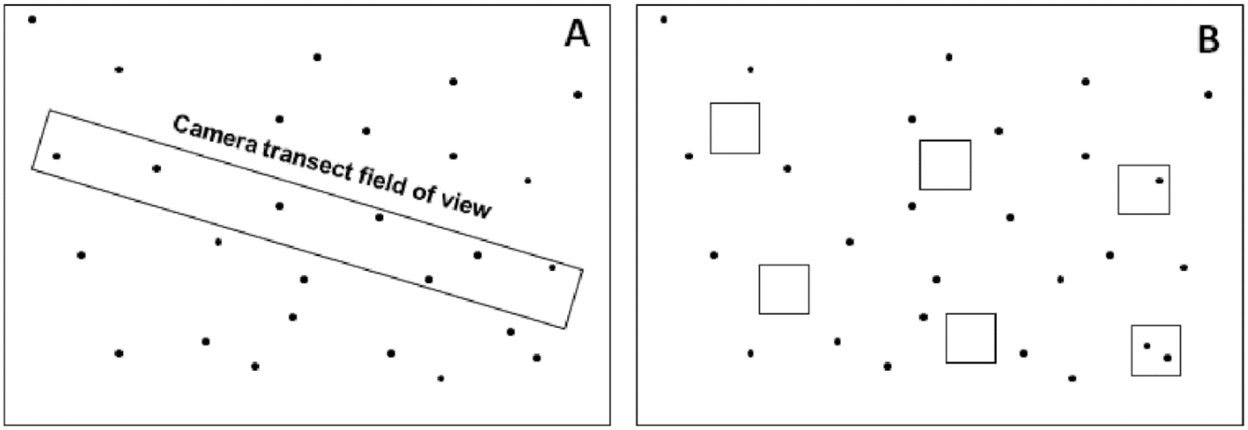

Figure 10. Comparative estimates of sea pen distribution and their sampling when using A) a camera gear, or B) grab samplers. Figure from Noble-James et al. (2018).

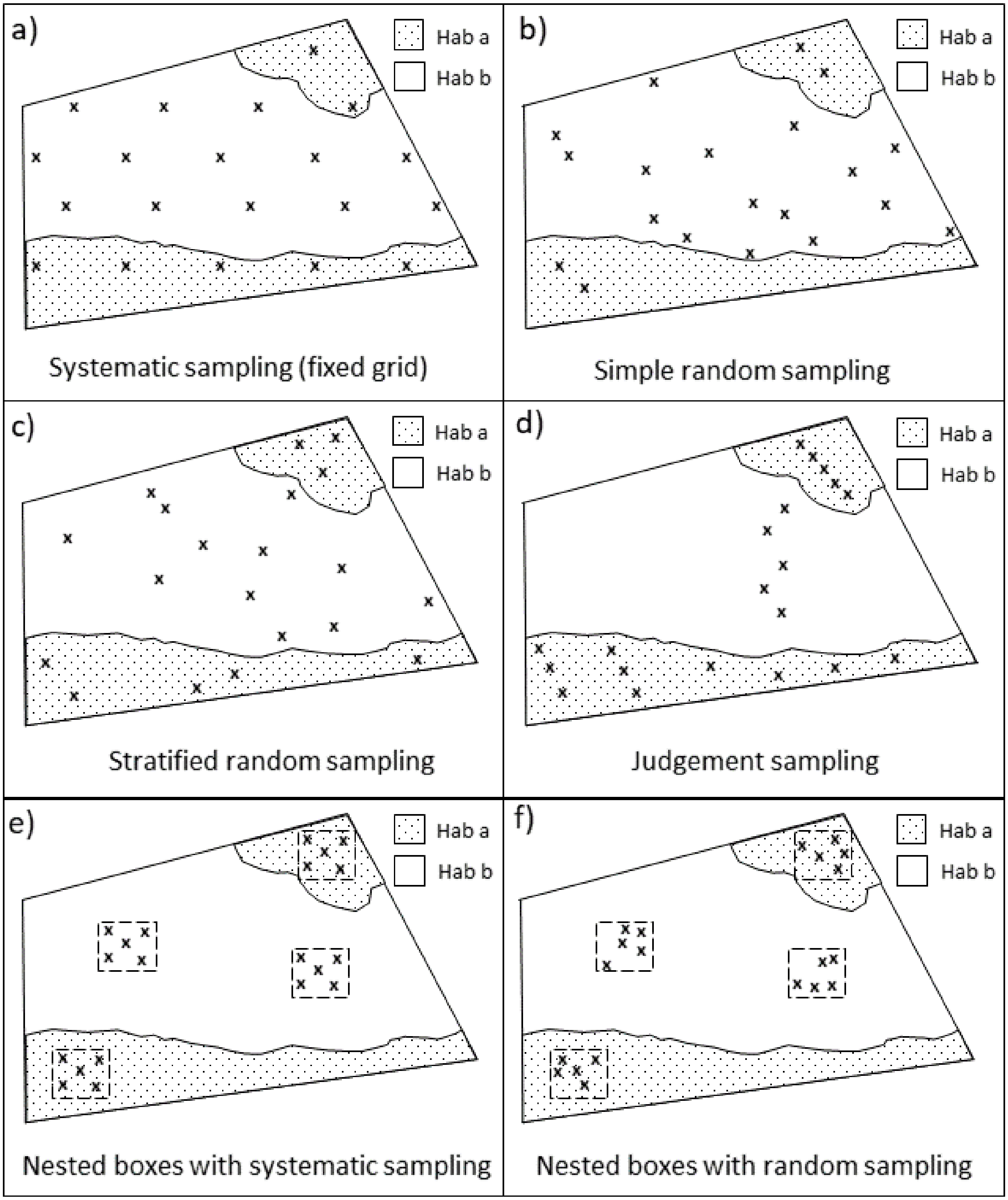

Figure 11. Different scenarios of sampling design in an area with two types of benthic habitat (Hab a = 40 % and Hab b = 60 %) and 20 sampling units ("x"). Probabilistic methods: a) a fixed grid is used with no consideration of habitat types, b) random sampling units are distributed across all the area; and c) random sampling units are stratified by habitat type with the same proportion. Non-probabilistic method: d) sampling units are selected with a main interest to specific locations in habitat a. Mixed approach: e) and f) nested boxes selected to represent habitat types.