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Elements of a framework to support decisions on authorizing scientific surveys with bottom contacting gears in protected areas with defined benthic conservation objectives

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Foreword

This series documents the scientific basis for the evaluation of aquatic resources and ecosystems in Canada. As such, it addresses the issues of the day in the time frames required and the documents it contains are not intended as definitive statements on the subjects addressed but rather as progress reports on ongoing investigations.

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ABSTRACT

Canada is rapidly increasing the number of protected areas in its domestic coastal and marine waters to meet international conservation targets. This has created an urgent need for approaches and frameworks for determining which human activities will be allowed within these areas in light of site-specific conservation objectives and monitoring requirements. Scientific activities contribute information that can support conservation-related management decision making within protected areas and in the broader ecosystem (e.g., advice for sustainable fisheries, species recovery, and ecosystem status). However, many of these same scientific activities can harm organisms, populations, assemblages and habitats within protected areas and therefore can hinder the achievement of conservation objectives, suggesting a need to evaluate the relative costs and benefits of conducting scientific activities within protected areas. This is particularly true for areas with ecologically sensitive benthic taxa and features, which can be harmed by bottom-contacting sampling gear such as bottom-trawls used in multispecies surveys. In January 2018, when a national peer review meeting on the subject was held, there were no existing frameworks or approaches in Canada or elsewhere to assist in determining under what conditions scientific surveys employing bottom-contacting gear should be permitted in protected areas. Such a decision requires consideration of the potential harm caused by the scientific activity, opportunities to mitigate this harm, potential benefits of the scientific activity for the monitoring and management of the protected area and potential consequences for science-based decision making in the broader ecosystem if the scientific activity is not authorized. This report reviews and discusses the key elements to be considered in such permitting decisions and which are incorporated in a new decision framework for Canadian coastal and marine protected areas developed at the January 2018 review. First, we review the key policy and legal frameworks in place to manage human impacts on benthic communities. Second, we review the available information, mainly from the fisheries literature, on the impact of bottom-contacting sampling gear on benthic ecosystems. Third, we propose and estimate metrics to define the potential impact of scientific activities in Canadian waters, with a focus on the full suite of ongoing long-term surveys. Fourth, we review and evaluate approaches for mitigating impacts, such as changes in the sampling gear and in sampling procedures. Fifth, we review the uses and utility of scientific monitoring activities, as well as the potential consequences that eliminating ongoing sampling activities in newly protected areas might have on the formulation of science advice and for management decisions within the broader ecosystem. These considerations and the inherent trade-offs are then illustrated with a case study and outstanding key uncertainties and considerations are discussed. The elements discussed here and linked together in the new framework do not lead to prescribed decisions. Rather, they support an information gathering process that will assist the management sectors in any region of Canada in their review for authorizing proposed scientific activities with bottom-contacting gears in protected areas.

1. INTRODUCTION

At the 10th meeting of the Convention on Biological Diversity (CBD), Parties, including Canada, agreed to a revised and updated Strategic Plan for Biodiversity that includes five strategic goals and 20 biodiversity targets (known as the "Aichi Biodiversity Targets") that are to be met by 2020. Aichi Biodiversity Target 11 specifies that "By 2020, at least 17% of terrestrial and inland water, and 10% of coastal and marine areas, especially areas of particular importance for biodiversity and ecosystem services, are conserved through effectively and equitably managed, ecologically representative and well-connected systems of protected areas and other effective area-based conservation measures and integrated into the wider landscapes and seascapes.". In 2015, Canada effectively re-iterated this target in its Canadian Biodiversity Strategy and its Biodiversity Outcomes Framework. Furthermore, Canada strengthened its commitment to meet the 2020 target for coastal and marine areas by adding an interim target of increasing protection in those areas to 5% that was met before the end of 2017. Prior to 2017, <1% of domestic marine and coastal waters were protected. By 2020, the spatial extent of protection will therefore have risen dramatically. In turn, this has created an urgent need for approaches and frameworks for determining which human activities will be allowed within the areas in light of site-specific conservation objectives and monitoring requirements.

Protected coastal and marine areas in Canada can be created using a number of regulatory tools administered by different departments or agencies and for a range of conservation objectives (further details in section 2). These include protected areas established under the Oceans Act, the Canada National Marine Conservation Areas Act and the Canada Wildlife Act, and as critical habitat under the Species at Risk Act, as well areas designated under the Fisheries Act for the protection of ecologically sensitive benthic species and habitats under DFO`s Policy on Managing the Impacts of Fishing on Sensitive Benthic Areas. Under that policy, identification of Significant Benthic Areas, defined in the Ecological Risk Assessment Framework (ERAF) as "significant areas of cold-water corals and sponge dominated communities", have also been considered for the selection of areas protected to meet the Aichi 2020 Targets. These species groups (corals and sponges) were highlighted as they are the focus of United Nations General Assembly (UNGA) resolutions calling for the protection of vulnerable marine ecosystems (VMEs) from significant adverse impacts of bottom contact fishing gear. As such they can represent extreme cases of potential fisheries impacts. Corals and sponges are explicitly mentioned in the UNGA resolutions as many member taxa have high vulnerability to damage and destruction by fishing gear and life history traits that predict long recovery periods (decades and more). However, other species that meet the biological characteristics associated with vulnerability produced by the Food and Agricultural Organization (FAO) of the UN may also qualify for protection. A Sensitive Benthic Area is a Significant Benthic Area that is vulnerable to a proposed or ongoing fishing activity.

All of these protected areas have defined conservation objectives targeting specific species, assemblages, biogenic habitats or physical habitats and features. Regulation of human activities that pose a risk of compromising these conservation objectives within the protected areas will involve either mitigation to reduce the risk of harm, or exclusion from the area. Fishing using bottom-contacting gear is often the principal human activity associated with an elevated risk of harm to benthic species and ecosystems, given the large spatial area it occupies and the manner in which the gears employed interact with the ocean floor and the species that live there (Collie et al. 2000; Kaiser et al. 2006; Hiddink et al. 2017; Koen-Alonso et al. 2018). DFO has produced risk assessment frameworks to evaluate the risk posed by those and other human activities specifically for the conservation of sensitive benthic species (DFO 2019b), and for a broader range of conservation objectives (O et al. 2015). The existing frameworks therefore provide specific guidance for such activities (reviewed in section 2 below).

1.1 PROTECTED AREAS AND SCIENTIFIC SAMPLING

An important issue for protected areas is to what degree research activities should be permitted within them. Of great but not exclusive interest are ongoing fisheries-independent research surveys. In Canada, the footprint of many marine fisheries is fully (or nearly) circumscribed by scientific surveys conducted by DFO or its partners to monitor the status of fishery resources, as well as other ecosystem components. Data gathered by these surveys has been used in the identification and planning of protected areas, either directly, by helping to identify significant areas of aggregation of sensitive species such as corals and sponges (e.g., Kenchington et al. 2016), or indirectly, by helping to define ecologically and biologically significant areas requiring protection (e.g., Savenkoff et al. 2007; Ban et al. 2016; Wells et al. 2017). The spatial overlap between existing scientific surveys and planned or existing protected areas is therefore very high.

Scientific surveys often employ bottom-contacting gear that is identical or similar to commercial fishing gear, though the fishing intensity is considerably reduced. Importantly, they follow sampling designs, which in some cases, may include stations allocated in areas that are not commercially fished. This means that the surveys may have stations over pristine or lightly fished sea floor, which was initially an advantage in enabling the identification and monitoring of benthic habitats.

There is, however, a range of views on whether any benthic impact is acceptable once areas are designated for protection. Through their impact on benthic organisms and features, it is possible that in some instances research surveys may enhance the risk of failing to achieve benthic zone conservation objectives and would therefore constitute unacceptable harm in these areas. However, a given activity has degrees of impact depending upon seabed type (hard or soft bottom), gear type and configuration, and the biological characteristics of the resident species, all of which must be evaluated against the conservation objectives for the closed areas and their associated management and monitoring plans.

Fishery-independent surveys play extremely important roles in determining population status and trends of commercial and non-commercial species, and underpin Canada’s sustainable management of its fisheries, which are valued at close to 3.2 billion dollars (Fisheries and Oceans Canada 2017). Of the approximately 300 commercially harvested marine and diadromous finfish and marine invertebrate stocks managed by DFO, approximately 260 are managed or tracked using scientific advice that is based on scientific surveys, of which 180 are based on bottom-contacting surveys (DFO unpublished inventory).

Changes to the survey sampling area or station allocation may affect the quality of scientific advice for the commercial fish stocks and recovery planning for species of conservation concern. Modification to survey sampling plans to lessen risks to conservation of benthic taxa or features may therefore, in some instances, enhance risks for the conservation of other species. When < 1% of coastal and marine areas was closed to protect species/habitats of conservation concern, this issue was dealt with on a case-by-case basis within DFO regions. However, with the increasing spatial extent of closed areas inspired by the Aichi 2020 targets, a national decision-making framework is urgently needed.

The decision to allow or exclude scientific surveys led by DFO or its partners employing bottom-contacting gear in protected areas involves balancing several tradeoffs related to marine conservation, which we discuss herein. It is clear that in some cases, the evaluation of these trade-offs will require inter-departmental co-operation where closed areas and their conservation objectives are set by ECCC or Parks Canada, and with the provinces and indigenous communities where there is shared jurisdiction or collaborative governance.

1.2 AVAILABILITY AND SUITABILITY OF EXISTING APPROACHES

This document was presented at a national science peer review meeting in January, 2018. At that time there were no published frameworks or approaches to assist in determining under what conditions scientific surveys or other research activities employing bottom-contacting gear might be excluded from protected areas. Existing approaches in the United States and at NAFO (Northwest Atlantic Fisheries Organization), briefly reviewed below, provided some elements but did not address the issue completely. In June 2018, authors from the United States published an ecological framework for informing permitting decisions on scientific activities of various sorts in protected areas in marine waters (Saarman et al. 2018). Because that framework was not available at the time of the peer review, it did not influence the conclusions of the meeting, the elements of which are documented in this report. To remain consistent with the information presented and discussed at the meeting, the contents of Saarman et al. (2018) are therefore discussed and contrasted with the present approach only later in this report.

The United States National Marine Fisheries Service has undertaken Programmatic Environmental Assessments to evaluate the impacts of their scientific activities on marine organisms, features and the ocean environment in general, including in National Marine Sanctuaries and areas identified as Essential Fish Habitat (URGS Group, 2016). The assessments are a requirement for authorizations to undertake federal research activities, but are not limited to protected areas. The assessments evaluate the overall impacts and the acceptable harm of status quo scientific sampling, as well as possible modifications to research programs, surveys or survey coverage and intensity to mitigate impacts on biota and habitats. The assessments are broad and not intended for case or location specific review. Furthermore, the assessments consider impacts on specific groups of biota (e.g., benthos) or features only in general terms and in light of overall environmental impacts only. In that context, the assessment for the northeast US concluded that status quo monitoring, with additional surveys and research, was acceptable. It also concluded that bottom-trawl surveys are generally “likely to have only minor and short-term effects on benthic communities”, given that they are “of short duration, generally of randomized design, are rarely repeated in the same location over time, and are collectively much smaller in scale… compared to intensive and chronic bottom trawling conducted by commercial fisheries” (URGS Group 2016). Furthermore, the surveys were deemed “essential to developing robust fisheries management measures that prevent overfishing and rebuild overfished stocks” and to support programs to address research challenges such as the potential effects of climate change and ocean acidification.

The issue of excluding surveys from coral and sponge closure areas has been studied at NAFO. To date, the only available published work considered abundance/biomass indices for fishery resources in the NAFO Convention Area and the impact on them of excluding surveys from areas in which they formerly operated (González Troncoso et al. 2016; Rideout and Ollerhead 2017). In the specific area studied (NAFO Divisions 3LMNO), exclusion of surveys had little or no impact on the relative abundance indices. While the studies provide an example of methodology for evaluating the impacts of changes to surveys on the provision of scientific advice, this is only one of the elements that may need to be considered in other situations and which are discussed in this report.

1.3 CONTEXT FOR THIS REPORT

DFO Oceans Management requested advice on the conditions under which scientific research surveys with bottom-contacting gears may be authorized for sampling in protected areas with defined benthic conservation objectives. The impacts of scientific surveys on benthic organisms and habitats and the importance of sampling in the protected areas to the integrity of the historical time series from the scientific surveys will be case specific. To assist in this decision making process, this report reviews a suite of considerations that constitute the important elements of a framework that can be applied consistently across Canada. A national science peer review meeting held in Ottawa on January 16-18, 2018, reviewed and revised these considerations and developed such a national framework (DFO 2018). The principal objectives of the meeting were:

- To develop descriptors of the features of the important benthic components that are vulnerable to the bottom contact gears. These could include sensitivity to disturbance (structure), mobility, resilience, generation time, etc.;
- To develop criteria to assess scale and scope of impact of the scientific activity (for example proportion of protected area potentially impacted, frequency of surveys, seasonality, type of gear used), including the impact on achieving conservation goals;
- To develop criteria to assess the consequences of excluding / modifying survey protocols and design on the integrity of the time series information and reliability of harvest advice on ecosystem components under study. This also includes the consequences of excluding / modifying survey protocols on monitoring valued benthic and ecosystem components in the protected area; and
- To provide guidance on applying the framework to specific cases.

Information and advice concerning the first two objectives were intended to be general and relevant to any proposed or ongoing research activity using bottom-contacting gear in a protected area. The information and advice for the third objective are pertinent for existing and ongoing research activities, typically surveys aimed at long-term monitoring of marine species and communities. The framework is intended to be general and not specific to application in Canadian protected areas exclusively and is not intended to be limited to research activities undertaken by the Government of Canada. A national framework that ties together the elements of this report and operationalizes their application is presented in detail in DFO (2018).

2. OVERVIEW OF THE POLICY AND LEGAL INSTRUMENTS AND PROCEDURES AND GUIDANCE RELEVANT TO MANAGING IMPACTS OF HUMAN ACTIVITIES ON BENTHIC COMMUNITIES AND FEATURES

The management of the impact of human activities on benthic communities and features employs a diversity of policy and legal instruments supported by guidance aimed at gauging relevant ecological risks to support decision making. While these tools are designed for human activities in general, many will apply or be relevant to managing disruptive and destructive research activities. Furthermore, many already involve procedures for authorizing research activities that could be informed by the considerations outlined later in this report and synthesized in the national assessment framework (DFO 2018). It is therefore relevant to briefly review these tools and procedures before a detailed discussion of the considerations for authorizing bottom-contacting research activities.

2.1 INTERNATIONAL COMMITMENTS

Internationally, Canada is signatory to a number of binding and non-binding commitments related to marine habitat conservation. Canada has committed to identifying Ecologically and Biologically Significant Areas (EBSAs) in its national waters through commitments to the

Convention on Biological Diversity (CBD). Eight criteria for assessing candidate EBSAs have been identified including uniqueness, special importance for life history, aggregation, vulnerability, naturalness, importance for at risk species and/or habitats, biological productivity, and biological diversity (DFO 2004, 2011). At the international level, steps have been taken to protect marine biodiversity at vulnerable marine ecosystems (VMEs), which have similar criteria as EBSAs and additionally one for structural complexity (Ardron et al. 2014).

In 2009 the United Nations General Assembly adopted the International Guidelines for the Management of Deep Sea Fisheries in the High Seas by the FAO Committee on Fisheries (UNGA Resolution 61/105). These guidelines are designed to limit the impacts of deep-sea fishing on fragile deep-sea fish species and habitats, including certain cold water corals and some types of sponge dominated communities (FAO 2009). In developing the guidelines, the experts were particularly concerned with:

> ”the sensitivity and vulnerability of some species, communities and habitats (i.e. VMEs) and the significance of direct and indirect impacts (i.e. significant adverse impacts; SAI) of fishing based on its ability to recover which is linked to key biological parameters including: the extreme longevity (100s to > 1,000 years) of individuals of some types of organisms or the long periods over which some habitats develop; the low resilience of particular species, communities and habitats; a high proportion of endemic species with risk of loss of biodiversity, including extinctions; distribution of some vulnerable seafloor communities as spatially discrete units often within a small area of the seabed so that small perturbations may have significant consequences; fragmentation and risk of loss of source populations; and poor current knowledge of the ecosystem components and their relationships.”

2.2 LEGAL INSTRUMENTS FOR BENTHIC CONSERVATION

A suite of federal programs and statutory mechanisms are utilized to achieve benthic conservation policy objectives and biodiversity outcomes for Canada. Mechanisms of primary concern are area-based management measures implemented via restrictions to commercial bottom fisheries in sites where benthic conservation objectives are clearly articulated. Areas meeting these criteria are presently dominated by long-term fishing closures and MPAs.

2.2.1 Fisheries Closures

The *Fisheries Act* and its subsidiary regulations provide DFO with the mandate and tools necessary to manage the fisheries. Restrictions to fishing activities are implemented using licence conditions and variation orders that set out times and areas where certain gear types cannot be used. Fisheries closures are enacted for many reasons and to meet a variety of objectives, including fisheries conservation purposes and the protection of SBAs. Table 1 provides examples where some or all bottom-contacting fishing gear has been restricted to achieve benthic conservation objectives. Management approaches and legislative tools are further elaborated in the *Status Report on Coral and Sponge Conservation in Canada* (Campbell and Simms 2009), the *Pacific Region Cold-Water Coral and Sponge Conservation Strategy* (DFO 2010a) and the *Coral and Sponge Conservation Strategy for Eastern Canada* (DFO 2015a). Additional benthic fisheries closures, that qualify as other effective area-based conservation measures for Aichi Target 11 reporting purposes (DFO 2016), are described in the online *List of Marine Refuges* maintained by DFO (DFO 2019a).

2.2.2 Marine Protected Areas

Canada’s protected areas include an assortment of federal, provincial, territorial, aboriginal and voluntarily imposed marine sites that have been protected by law and other effective means to maintain and enhance the status of species, habitats, unique and representative features, submerged archaeology, culturally significant areas and recreational opportunities. Most of the country's protected areas established for marine ecological purposes share the common conservation goal of protecting marine biological diversity. Three federal departments are involved: Parks Canada establishes National Marine Conservation Areas; the Canadian Wildlife Service (CWS) oversees Protected Marine Areas, Bird Sanctuaries and Wildlife Areas; and DFO creates Marine Protected Areas (MPAs). Programmatic background is summarized in the *National Framework for Canada's Network of Marine Protected Areas* (Government of Canada 2011). To date, Parks and CWS have not imposed extensive bottom fishing restrictions to protect vulnerable benthos, so the scope here is narrowed to a discussion of DFO sites where the assessment framework is expected to find more immediate application.

The *Oceans Act* provides DFO with the mandate and regulatory tools for establishing and managing Marine Protected Areas (MPAs) to accomplish several goals, including statutory purposes that are invoked to safeguard significant and sensitive benthic environments (e.g., the conservation and protection of unique habitats and marine areas of biodiversity). Each MPA is custom-designed to meet a set of site-specific conservation objectives which themselves are dictated by site properties elaborated during the ecological assessment phase of planning. Boundaries, zoning schemes, restrictions and allowable activities are set out in MPA regulations. Site management plans are then used to articulate site-level ecosystem values and conservation objectives plus governance approaches, compliance and enforcement needs, educational opportunities and administrative oversight, including authorization mechanisms.

MPAs employ a regulatory model consisting of a blanket prohibition against disturbance, damage, destruction and removal. Exceptions may be provided for activities that have been assessed for risk and deemed compatible with the conservation objectives. Thus if the conservation focus is benthic and bentho-pelagic, as in the Endeavour Hydrothermal Vents MPA in the Pacific offshore, surface activities like shipping and pelagic fisheries may be allowed to continue. Restrictions can apply across the site or they can vary according to area partitions derived from MPA design principles and spatial risk assessment. For example, low-impact fixed-gear fisheries are allowed to continue in three management zones comprising 25% of the St. Anns Bank MPA. Certain other activities, including scientific activities are subject to Ministerial approval in DFO MPAs. Table 2 provides a brief summary of DFO sites, highlighting benthic conservation objectives and restrictions on bottom fisheries.

2.3 RESEARCH AUTHORIZATION PROCEDURES

A variety of guidelines, policies and laws govern the conduct of environmental monitoring and marine scientific research (MSR) in Canadian marine waters. Legal requirements, procedural steps and decision-making processes of specific relevance to bottom-contacting survey methods are summarized here as they pertain to both domestic and foreign investigators proposing work in fishery closures, MPAs and other areas of conservation concern.

2.3.1 Scientific Fishing Licences

The Fisheries Act and its subsidiary instruments comprise the primary mechanism for authorizing domestic MSR activities on or affecting living marine resources and associated marine habitats. Section 35 prohibits the alteration or destruction of fish habitat without an authorization, however, that provision is rarely triggered by MSR proposals. Sections 7 and 45 establish the Minister's authority to issue licences and regulate the fishery and it is those powers that are applied most often to research.

Domestic scientists intending to deploy biological survey and sampling equipment are normally required to apply for and obtain a clearance under the Fishery (General) Regulations. Section

52 allows the Minister to issue a licence to fish for experimental, scientific, educational, aquatic invasive species control or public display purposes. Licences are required whether the sampling involves the collection of entire marine organisms (animals and plants) or only parts thereof (e.g., organs, tissues, skeletal fragments). If live specimens are being collected and moved between sampling platforms and rearing or laboratory facilities, scientists are also required to obtain a transfer licence pursuant to section 56. Research application forms generally request a project description and information on collection methods. Location maps and station coordinates allows these proposals to be screened against known and designated areas of benthic sensitivity. Terms and conditions may be formulated and attached to scientific licences as for other categories of fishing when species, gear or area restrictions are imposed.

In general practice, DFO Science personnel are authorized to conduct unspecified fisheries MSR under a blanket Section 52 licence whereas researchers affiliated with universities and the private sector must apply for and be granted a scientific licence authorizing them to conduct a specified program of sampling. Non-government scientists and the fishing industry often work with DFO investigators, especially in offshore settings where joint cruises and collaborative arrangements are commonplace. In circumstances where the DFO partner already holds a scientific licence, it may be possible to have external collaborators covered by the authorization issued to the departmental employee.

Program and project-based fisheries research activities undertaken by the Government of Canada and partner organizations have not been subject to comprehensive environmental assessments as they have been in other jurisdictions, such as the United States, where scientific proponents must address protected species and Habitat Areas of Particular Concern (URGS Group 2016). However, risks are assessed as a routine part of DFO survey planning and operations. DFO scientists are also obligated to issue a Fisheries Research Notice. Although there appears to be little precedent for imposing no-take or no-contact zones via scientific license conditions for DFO investigators, area-based mitigation measures have been implemented as illustrated by the removal of trawl survey sets from coral conservation areas in Maritimes Region (DFO 2006). Such actions are consistent with the Pacific and Atlantic coral and sponge conservation strategies, which call upon researchers to minimize removals and employ non-destructive survey and sampling methodologies.

2.3.2 MPA Activity Plan Approvals

Oceans Act MPA regulations contain provisions for describing the process to approve activity plans. Approval from the Minister of Fisheries and Oceans is necessary for certain activities including scientific research and monitoring. Although research is supported in principle in all MPAs, scientific activities have not been granted an unqualified exception to the general prohibitions. Scientists are required to submit an activity plan for Ministerial approval. Two sections of the regulatory model for *Oceans Act* MPAs govern these authorizations: the first section lays out the information submission requirements (e.g., dates, location, gear, methods, impacts) while the second section stipulates approval conditions and timelines for a Ministerial decision. Risk-based approval conditions can vary from site to site across the country according to the specific conservation needs of the area. There are also standard approval conditions for scientific research and monitoring, such as the scientific activity must contributing knowledge and understanding to the conservation, protection and management of an MPA.. Benefits accrued by potentially disruptive research must be demonstrably tied to the purposes for which the MPA was established. Support for lethal or destructive sampling may be justified when a site monitoring framework has identified harmful methods as both necessary and responsive to the MPA conservation objectives.

In terms of procedures, research proponents trigger the DFO application process by formulating and submitting an activity plan for undertaking scientific research and monitoring to the MPA program. An assessment of environmental risk is conducted with input from specialists and experts as necessary. The governance approach for some MPAs requires that activity plans be distributed to multi-stakeholder advisory committees for review and input. MPA Program officials further assess research proposals against site level conservation objectives and pre-existing monitoring frameworks that prescribe ecological indicators and recommend appropriate sampling protocols (for an example see Stanley et al. 2015). Consideration is also given to cumulative environmental effects. Feedback is compiled and a recommendation is put forward to the Minister. Notwithstanding variations on timing, specific information sought from proponents and mandatory regulatory conditions that must be met, submitted plans can be approved, denied or modifications sought from the applicant. Once a plan has been approved, the activity is effectively authorized to proceed. Depending on the MPA regulations, proponents may be required to submit to the department incident reports, activity reports, data or resulting publications once the research activities have concluded.

2.3.3 Other Conservation-Oriented Authorizations

Large conservation areas being established and managed by Parks Canada (e.g., the proposed Lancaster Sound National Marine Conservation Area) and the Canadian Wildlife Service (e.g., the Scott Islands Marine National Wildlife Area) are also expected to have considerable overlap with existing survey areas where DFO and other parties have conducted research for many years. Additional permitting frameworks will apply to bottom-contacting research as those areas gain full legal protection.

The Parks Canada Agency operates a centralized Research and Collection Permit System through which proponents access a Researcher’s Guide and application forms (Parks Canada 2017). Investigators proposing work in the Agency’s existing marine areas, like the Saguenay-St. Lawrence Marine Park, will be directed to research coordinators at the site level. Permitting procedures administered by the Canadian Wildlife Service are guided by the *Wildlife Area Regulations* and the Environment and Climate Change Canada protected areas permitting policy (Environment Canada 2012).

The Species at Risk Act (SARA) applies throughout Canadian marine waters. Sections 73 and 74 of SARA establish an authorization regime for activities that are otherwise prohibited (e.g., capture, damage to a residence, destruction of critical habitat). However, no benthic VME species have been scheduled under SARA or identified as a component of critical habitat for a listed species. Although at-risk demersal finfish including cusk (COSEWIC 2012) and wolfish (Gilkinson and Edinger 2009) have been found in coral areas, those habitat associations have not been assessed as critical to survival and recovery of the depleted populations. Therefore, there is no SARA permit requirement at this time for research on or affecting VME organisms.

The Committee on the Status of Endangered Wildlife in Canada (COSEWIC) assesses the status of wildlife in Canada. At present, there are no coral or sponge species on COSEWIC’s prioritized list of candidate wildlife species (last updated July 10, 2017). Corals (n = 190) and sponges (n = 212) were examined by the Canadian Endangered Species Conservation Council (2016) with several species being ranked as vulnerable.

Stony corals (*Scleractinia* spp.) and black coral (*Antipatharia* spp.) are listed in Appendix II of the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) and this places certain authorization obligations on Canada. Specimen removals could trigger applications for export permits, interprovincial transportation licenses and scientific certificates under the *Wild Animal and Plant Protection and Regulation of International and Interprovincial Trade Act*. DFO is the designated Management Authority and Scientific Authority for CITES-listed aquatic species in Canada. In practice, CITES applications are processed by the DFO Catch Certification Program. Before issuing a permit, DFO Science may be asked to confirm that export of specimens of a certain species will not be detrimental to its survival in the wild.

2.3.4 Foreign Marine Scientific Research Consents

Foreign governments, institutions, entities and individuals wishing to conduct MSR in waters under Canadian jurisdiction must obtain advanced consent from the Government of Canada in accordance with UNCLOS. Research proponents are also expected to comply with applicable domestic law. UNCLOS, which was ratified by Canada in 2003, establishes the global framework governing access to the waters of another country for research purposes. The Convention addresses overseas MSR in territorial seas, exclusive economic zones (EEZ) and in some circumstances, on outer continental shelves. The last provision is especially relevant for Canada owing to the significant benthic features recognized beyond the EEZ on those portions of the outer continental shelf claimed by Canada in its Atlantic submission (Government of Canada 2013), especially where those claims overlap the NAFO VME closures.

UNCLOS Guidelines for MSR direct researchers to seek foreign consent through their diplomatic mission using the standard UN application form and corresponding instructions (DOALOS 2010). Global Affairs Canada (GAC) administers the domestic authorization process on behalf of the Government of Canada (CBSA 2017). Upon receipt, GAC initiates the foreign vessel clearance process by forwarding the application to implicated government departments for review, evaluation and compliance with Canadian law and policy. If approved, a final MSR consent that bundles mandatory licences or permits, including those under DFO purview, will be issued by GAC through diplomatic channels. As per UNCLOS provisions, the authorization may impose “reasonable conditions” pursuant to Canadian laws. This is one avenue for having DFO authorities incorporated if the proponent has not already made separate application to obtain the necessary clearances.

Several DFO authorities can be triggered by a foreign MSR application. Depending on where the proposed research is to be undertaken, and the fate of biological samples, proponents may need to obtain authorizations under the *Fisheries Act*, SARA, CITES and MPA regulations in which case the above application and review procedures apply. A fishing license is a fundamental requirement. Foreign MSR involving biological removals is subject to the licensing provisions of the *Coastal Fisheries Protection Act* and *Regulations* which govern sampling of: i) any living marine organism in the EEZ, and ii) “any living organism that, at the harvestable stage, either is immobile on or under the seabed or is unable to move except in constant physical contact with the seabed or subsoil” of the outer continental shelf. These instruments are applied to foreign researchers in much the same way that the *Fisheries Act* and its regulations are applied to domestic researchers; investigators submit an application and may be granted a foreign fishing vessel license authorizing entry to Canadian waters for purposes of scientific research subject to any species, gear or area conditions that might be imposed by Canada. Accordingly, it is expected that the framework under development here will be helpful for assessing foreign MSR requests involving bottom contact.

2.4 DECISION SUPPORT FRAMEWORKS AND GUIDANCE

No suitable frameworks currently exist that specifically focus on admissibility of scientific activities in protected areas with benthic conservation objectives. However, a number of decision support frameworks and other departmental guidance have been developed to identify the level of ecological risk of fishing and other human activities on benthic communities and features, and they can be drawn upon in this process.

2.4.1 Ecological Risk Assessment Frameworks for Corals and Sponges

A key tool for use in the implementation of the *Policy on Managing the Impacts of Fishing on Sensitive Benthic Areas*, is the *Ecological Risk Assessment Framework for Coldwater Corals and Sponge Dominated Communities* (SBA ERAF; DFO 2019b). The SBA ERAF is a process for identifying the level of ecological risk of a fishing activity and its impacts on benthic areas. It was developed specifically in consideration of cold-water corals and sponge-dominated communities, referred to as significant benthic areas. The application of this framework to additional benthic habitat, communities and species will require modifications to the SBA ERAF. It is also important to keep in mind that the SBA ERAF examines only the ecological component of risk.

The SBA ERAF assessment methodology is loosely based on risk assessment frameworks described in Fletcher (2005) and Hobday et al. (2011). The main objective is to provide managers with an estimated level of risk for serious or irreversible harm to significant benthic areas resulting from fishing activity using a specified gear type. A secondary objective of the process is to provide advice on potential management measures to mitigate or avoid serious or irreversible harm to significant benthic areas.

The risk assessment process evaluates available data on the interaction between the gear type and the significant benthic area in question. It involves the following steps: estimation of the *consequence* and the *likelihood* of an overlap between the significant benthic area(s) and the fishing activity which utilizes the gear type under consideration (range 1 to 4 for each) and construction of the risk matrix; scoring of the risk (range 1 to 16); and categorization of the risk (low to high). Advice on management options is provided based on the determined level of risk.

2.4.2 Pacific Region Ecological Risk Assessment Framework

The SBA ERAF is a simplified process that considers only direct impacts of single fishing gear types on significant benthic areas in general. To assist with more complex applications and to support ecosystem approach to management, DFO Pacific Region developed a broader Ecological Risk Assessment Framework (ERAF) that evaluates single and cumulative threats from stressors associated with various anthropogenic activities to Significant Ecosystem Components (O et al. 2015). This approach has been guided by other risk assessment exercises, including the Australian Ecological Risk Assessment for the Effects of Fishing framework (Hobday et al. 2011; Williams et al. 2011), risk frameworks developed within DFO (Park et al. 2010; Hardy et al. 2011, Holt et al. 2012), and those applied in the US (Samhouri and Levin 2012). Significant Ecosystem Components (SECs) are ecological components identified as significant to the health and functioning of the specific ecosystem and may be selected at species, habitat, or community / ecosystem level. The framework consists of a scoping phase, where SECs, anthropogenic activities and associated stressors are identified. The SECs are then tabulated against the stressors and each SEC-stressor pair is assessed for a potential interaction (1 indicates a potential negative interaction and 0 assumes no negative interaction based on current knowledge). This is followed by a risk assessment where components of exposure and consequence (divided into resilience and recovery aspects) are scored for stressors and SECs expected to interact. The ERAF provides three risk assessment options: Level 1 (qualitative), Level 2 (semi-quantitative) or Level 3 (quantitative).

To date, the utility of the ERAF process has been evaluated through application of a Level 1 (qualitative) risk assessment to the Pacific North Coast Integrated Management Area (PNCIMA; Murray et al. 2016) and Level 2 (semi-quantitative) risk assessments to three MPAs in the Pacific Region: SGaan Kinghlas-Bowie Seamount MPA and Endeavour Hydrothermal Vents MPA (DFO 2015b), as well as the Hecate Strait and Queen Charlotte Sound Glass Sponge

Reefs MPA (HS/QCS MPA; Hannah et al. unpublished MS). In these past applications, SECs selected during the scoping phase reflected the conservation objective or were related to its elements. For example, for the HS/QCS MPA whose conservation objective is *"to conserve the biological diversity, structural habitat, and ecosystem function of the glass sponge reefs*", SECs include reef-building glass sponge species and glass sponge reef skeleton matrix. Pathways of Effects models were used to identify stressors associated with each human activity in question, including fishing, and the SEC and stressor pair were evaluated for interaction and scored as part of the risk assessment.

2.4.3 Oceans Program Risk Module

The Oceans Program is developing an Oceans Management Risk Module to provide practical guidance and mandatory requirements for DFO oceans practitioners when undertaking risk assessments to inform decisions for Canada's oceans management. The guidance follows the international standard for risk management ISO 31000:2009 but has been adapted to reflect specific application needs within the Oceans Program. The module uses standardized risk criteria for impact, likelihood and uncertainty, a set of risk management terms, and guidance for risk tolerance, risk evaluation and risk treatment.

One of the applications of the risk approach is to inform decision making regarding the prohibition and permitting of activities in an *Oceans Act* Marine Protected Area. This is determined based on the risk posed by human activities on the achievement of the conservation objectives of the MPA. Scientific research and monitoring activities are addressed through the activity plan process described in the MPA regulations. The activity plan process allows the evaluation of such activities on a case by case basis to ensure compatibility with the achievement of the conservation objectives of the MPA. Scientific research and monitoring activities that compromise the achievement of the conservation objectives of the MPA would not be approved.

At the MPA establishment / design phase, MPA practitioners must formulate risk-based conditions for approving the activity plan for undertaking scientific research and monitoring activities in the MPA. These conditions for approval are to be included in the MPA regulations.

Another application of the risk approach is at the MPA management phase once it is designated. Practitioners need to undertake a risk analysis to evaluate the risks posed by proposed activities including scientific research and monitoring activities with the information that was submitted by the proponent as part of the activity plan. This informs whether the activity plan for undertaking scientific research and monitoring activities will be approved or not.

Key elements of the risk module that are relevant for this report are the standardized risk criteria for determining the impact of an activity on the conservation priority. These elements are to be used to understand and manage the risk associated with scientific research and monitoring relative to the achievement of the MPA conservation objectives. Key to interpreting the Oceans Program risk module is the scale at which impacts are expected. Therefore, while a bottom-contacting scientific activity can be destructive at a given location, it may not have an impact at the scales of concern. This motivates the need for the considerations outlined in this research document and the authorization framework that brings them together.

2.4.4 Other Departmental Guidance on the Impacts of Human Activities on Benthic Habitats

The Pacific Region Cold-Water Coral and Sponge Conservation Strategy (DFO 2010a) and the Coral and Sponge Conservation Strategy for Eastern Canada (DFO 2015a) review human activities known to impact cold-water corals and sponges. Of these human activities, a subset is relevant to scientific surveys that employ bottom-contact gear and have the potential to adversely impact cold-water corals and sponges (and other benthic communities and features with defined conservation objectives). These activities are fishing, introduction of aquatic invasive species, and direct research (for example, scientific sample collection) or commercial harvesting.

DFO Science has also published a number of Science Advisory Reports on fishing gear impacts to benthic habitats, including corals and sponges, which are discussed in more detail below. It was determined that mobile bottom-contacting gear impacts corals and sponges the most, due to the extent of the seafloor affected and the force exerted (DFO 2006, 2010b, 2010c). Whereas the impacts of mobile gear on coldwater corals and sponges have been extensively documented (Freese et al. 1999; Chuenpagdee et al. 2003; Ardron and Jamieson 2006), the impacts of other gear types such as pots and traps on coldwater corals and sponges are not as well studied. In all cases, the impacts were generally quantified at the scale commercial fisheries operate, and not necessarily at the scale of research activities.

3. IMPACTS OF BOTTOM-CONTACTING FISHING AND SCIENTIFIC GEAR ON BENTHIC ECOSYSTEMS

3.1 DEFINITIONS

The following definitions for terms used below were taken from DFO (2010b), with some small modifications to broaden their scope where appropriate, and with additions from ICES (2016).

- *Sensitivity* (converse *resilience*): the capacity of benthic taxa or features to respond to impacts resulting from bottom-contacting fishing or survey gear, and is dependent on the physical and/or life history characteristics that affect their capacity to respond.
- *Susceptibility:* interpreted as the vulnerability of benthic taxa or features to impacts resulting from fishing activities, and is affected by the likelihood of an impact occurring and the sensitivity of the benthic attribute in question.
- *Recovery:* refers to the return of the benthic attribute to the state from which the current fishery or survey activity impacted it, once that stressor is removed. It is not intended to refer to the return of the benthic attribute to a pristine state i.e., prior to any fishing or other human activities.
- *Recovery time:* refers to the elapsed time between the impact of the current activity and the return to the former pre-impact state.
- *Recurrence time interval*: the average time between the impact of activities at a given. Minimally, recurrence time intervals that are shorter than recovery times are expected to adversely affect the recovery potential of the benthic attribute (Thrush et al. 2005).

3.2 INSIGHTS FROM STUDIES OF BENTHIC IMPACTS OF COMMERCIAL FISHERIES

The impacts of commercial fishing on benthic species and habitats have been extensively studied for more than two decades and include both immediate and cumulative effects. Collectively those studies show that the immediate impacts of the direct effects of bottom trawling are the loss of erect and sessile epifauna, smoothing of sedimentary bedforms with reduction of bottom roughness, and removal of structure-forming taxa such as corals and sponges (National Research Council 2002; DFO 2006; Rijnsdorp et al. 2016; Sciberras et al. 2018). Mortality mostly occurs through direct removals, however, dead, moribund and damaged individuals can be left on the sea floor where they may be vulnerable to predation by fish and benthic scavengers as well as to disease. Sediment plumes created by the bottom-contacting trawls can also cause smothering in filter-feeding organisms such as sponges, which can lead to death a short time later (Leys 2013). The degree of the impact depends strongly on the fishing activity including the characteristics of the gear (Jennings and Kaiser 1998), tow speed, weather conditions (National Research Council 2002), fishing history at the site, natural disturbance regimes, and on the species composition of the affected benthic system (DFO 2006). Immediate impacts can range from very severe (Koslow et al. 2001) to minor (Kenchington et al. 2001). The most severe impacts occur in biogenic habitats which have enhanced biodiversity and may take a long time to form (Rijnsdorp et al. 2016). A recent meta-analysis of the global depletion of seabed biota after bottom trawling disturbance based on a comparison of sites having experienced different degrees of fishing showed that otter trawls removed on average 6% of organisms per trawl sweep and penetrated to an average depth of 2.4 cm into the sediment (Hiddink et al. 2017), with impacts that may be strongest on muddy bottoms and in biogenic habitats (Rijnsdorp et al. 2016; Sciberras et al. 2018). Another meta-analysis based on results of mobile gear impact experiments found that the average gear pass reduced benthic invertebrate abundance by 26% and species richness by 19% (Sciberras et al. 2018). Reductions in abundance were greatest for gears that penetrate the sediment the most, e.g., hydraulic dredge. Longer lived, often sessile, species took longer to recover (>3 years; the maximum resolution of the study), while shorter lived mobile taxa took <1 year to recover, on average. Recovery times were also a function of initial depletion and were therefore, on average, shorter for otter trawls and beam trawls (<1 year) compared to towed dredges and hydraulic dredges (>3 years). It should be noted that the reviews of Hiddink et al. (2017) and Sciberras et al. (2018) are based on results from generally shallow water (< 100 m) well studied areas, often with a long history of exploitation. Therefore the results may not be applicable to impacts in unfished areas exposed to low natural disturbance situations.

In regularly fished areas, “single tows” result in rapid recovery times (weeks to months, dependent on bottom composition), whereas in areas where commercial fishing has been inexistent or prohibited long enough for biological communities to have recovered partially or completely, depletion is more severe and recovery periods can take months or can be longer (Stevenson et al. 2004; Hiddink et al. 2017; Sciberras et al. 2018). These differences likely result from a shift in community composition favoring species with faster life histories in routinely fished areas (Jennings et al. 2001, 2005).The depletion of large, emergent, attached fauna such as sponges, corals and whips can be high in the first tow over an area. Studies reviewed for DFO (2010b) reported that within the path of a single trawl, 1% to 8% of corals and 20% to 70% of sponges were removed, and damage occurred at varying extent to the corals and sponges that remained (ranging from 23% to 100% for corals and 14% to 67% for sponges), depending on their growth form and size.

DFO reviewed the impacts of mobile gears on seafloor and benthos (DFO 2006) and the impacts of other gears, including longlines, gillnets, traps and pots (DFO 2010c). It was concluded that the “impacts of bottom trawl gears are initially greater on sandy and muddy bottoms than on hard, complex bottoms. However, the duration of impacts is usually greater on hard complex bottoms than on sandy bottoms and probably longer than on muddy bottoms.” (DFO 2006). Both reports discussed mitigation measures that could reduce the severity of the impacts and emphasized the importance of case-specific evaluations given the many combinations of gear / fishing history / bottom type / species. However, it was felt that “cautious extrapolation of information across sites is legitimate” where it is reasonable to do so (DFO 2006).

Bottom trawls are generally considered to have the greatest impact of all fishing gears on benthic habitats due to the net configuration, the net components in contact with the sea floor (i.e. trawl doors, footgear, etc.), and particularly the scale of the swept area interpreted as the area swept between the doors during a haul (Morgan and Chuenpagdee 2003; Pham et al. 2014; Clarke et al. 2015). The doors penetrate the sediment the most producing a narrow furrow that can be up to 35 cm deep in mud and up to 10 cm deep in coarser sediments, at least for commercial trawls (Lucchetti and Sala 2012; Eigaard et al. 2016; O'Neill and Ivanovic 2016). The sweep lines represent the largest portion of the swept area but appear to have the least impact on the seabed, with penetration in the top few centimeters if at all (Buhl-Mortensen et al. 2013; O'Neill and Ivanovic 2016). Even if the sweep lines do not penetrate the sediment, they can break or dislodge large or emergent benthic organisms via a shearing action (Ewing and Kilpatrick 2014).

Bottom-set fixed gears such as longlines, pots, and gillnets generally result in a lower impact than trawl gear due to a generally smaller footprint, but the impact can still be significant (Wareham and Edinger 2007; Suuronen et al. 2012; Pham et al. 2014; Clarke et al. 2015). Most of the damage is caused during the deployment and retrieval processes, which creates lateral movement of gear with long lines (Sampaio et al. 2012; Ewing and Kilpatrick 2014; Welsford et al. 2014), and dragging of pots across the sea floor, particularly during retrieval when the pots are deployed in strings (Wareham and Edinger 2007; Doherty et al. 2018). Fixed gears can also swing laterally due to movements caused by the catch, boundary layer currents, and storms. While some emergent taxa with flexible body structures can bend and withstand the shearing interactions with moving gear, attached organisms and those with fragile forms can be detrimentally harmed (Eno et al. 2001; Ewing et al. 2014). In a detailed study in deep waters of the Southern Ocean, Ewing et al. (2014) found that lateral movements from single demersal longline sets killed or damaged generally <30% but up to 45% of individuals of sensitive benthic taxa. Lost gear, particularly gillnets and pots and traps lacking adequate dissolvable escape panels, can ghost fish for extended periods of time and can entangle and damage fragile emergent taxa. Lost gear contributes to marine pollution, and depending on the materials used (Thomsen et al. 2010), can break down in into smaller pieces called microplastics, hazardous to benthic fauna such as corals (Hall et al. 2015).

Recent research on the impacts of trawling on benthic communities have examined changes in the functional composition of assemblages based on biological traits as opposed to single species (Bolam et al. 2014). This approach allows for impacts to be assessed in terms of ecosystem function and has direct links to international policy requirements (UNGA 61/105) which call for an assessment of significant adverse impacts of fishing at the ecosystem level, i.e. the VME (FAO 2009). DFO trawl surveys which operate at low sampling densities and sample randomly in space are unlikely to affect ecosystem function in most instances. However, if annual surveys cause serial depletion of habitat, as is possible with long-lived species with low recruitment and restricted distributions and where survey operations are undertaken at high density, then local ecosystem function could be impacted.

Existing trait-based frameworks effectively divide traits into two broad categories: exposure to the gear and sensitivity upon contact (Hewitt et al. 2011; Bolam et al. 2014, 2017; Clark et al. 2016). These traits are summarized in Table 3. Exposure is related to the form of the taxon, whether erect or emergent at the surface, and the living position in the sediment. Large species living at the surface will be exposed to breakage, crushing or dislodgement from all bottom-contacting gears, whereas species living within the sediment will be affected to differing degrees based on their position and the sediment penetration of the gear. Sensitivity to contact depends principally upon the fragility of the taxon, with taxa possessing exoskeletons or rigid structures often associated with higher incidence of damage or death. Sensitivity is also affected by body size, presumably in part as a result of the relationship between the force of impact, body mass, and fluid dynamics.

Many research surveys are undertaken at low sampling density (sets per km^2) and employ some form of random selection of stations. First-pass “single-tow” impacts are therefore likely to be more important that cumulative impacts, except in situations where the recovery times of species, features or habitats are very prolonged. One exception is situations where fixed stations are sampled, although in those cases the spatial impact is likely much smaller and probably most important when the protected space is also small. Cumulative impacts are also well studied and show a greater loss of biomass and change in species composition than seen in a single pass of the gear (Hiddink et al. 2006). There is also a change in the biological traits associated with the benthic species with a loss of attached, fragile, epifaunal, filter-feeding taxa and enhancement of motile scavengers, and robust, burrowing filter-feeders (Bolam et al. 2014) and a general shift towards species with high turn-over rates (Jennings et al. 2001, 2005).

3.3 IMPACT AND RECOVERY – CORALS, SPONGES AND OTHER VULNERABLE SPECIES

Cold-water corals and sponge biogenic habitats (reefs, grounds etc.), as well as hydrothermal vents and cold seeps, have been identified for protection in both Canadian and international policy (FAO 2009; DFO 2011, 2015a). These species, communities and habitats have low resilience, are severely impacted by bottom contacting fishing gears and have poor abilities to recover from such impacts (Rijnsdorp et al. 2016). Individuals for some of these species can be extremely long lived (100s to > 1,000 years) and the time required for biogenic habitat to develop can be even longer. Many species are endemic to these unique and spatially restricted habitats, such that damage can lead to population fragmentation, loss of biodiversity or extinction (e.g., Pacific glass sponge reefs; Conway et al. 1991, 2007; Dunham et al. 2018a,b). Boutillier et al. (2010) provide details on the life history traits of corals and sponges with respect to the FAO (2009) guidelines. Table 4 summarizes the recovery prognosis for coldwater sponges that were directly impacted by the gear or resulting sedimentation, extracted from their work.

More generally, the degree of impact and prognosis for recovery at a site depends both on the nature of the impact (e.g., type and magnitude) and the characteristics of the affected species. Some species of corals and sponges are more susceptible and/or sensitive than others. In evaluating a species’ degree of sensitivity and susceptibility to fishing activities, the following factors have been proposed (DFO 2019b) and are also pertinent for any benthic fauna affected by fishing activities:

- range and spatial distribution,
- morphology and skeletal composition (rigid vs. flexible),
- means of attachment to the substrate,
- life history characteristics, and
- habitat preferences.

Based on summaries in Hewitt et al. (2011), Clark et al. (2016) defined sensitivity categories for deep-sea benthic taxa which are relevant here, particularly for corals, sponges and other sensitive taxa. Table 5 of Clark et al. (2016) is reproduced, with modifications, in Table 5.

The potential for recovery for corals, sponges and other benthic fauna following disturbance by fishing gear is also determined by the characteristics of the affected species (FAO 2009; DFO 2019b), notably:

- recruitment potential,
- distance from other potential sources of colonizers,
- age at first maturity,
- reproduction strategies,
- maximum age (longevity), and
- growth rates.

3.4 IMPACT AND RECOVERY OF BENTHIC BIOTA – EXAMPLES AND LESSONS FROM ATLANTIC CANADIAN EXPERIMENTS

The sensitivity and recovery of the benthic ecosystem following fishing impacts has been studied for otter trawls, hydraulic dredges, and scallop dredges in Atlantic Canada. In all cases, the experiments were undertaken in areas already perturbed by commercial fisheries and which experience some level of large-scale natural disturbance. The following studies may therefore provide estimates of disturbance impacts and recovery times that may be relevant to surveys employing the same gear in these types of habitats, but are not relevant for habitats that have evolved in the absence of large-scale natural disturbance.

A three-year otter trawling experiment was conducted in the 1990s on the Grand Bank of Newfoundland and Western Bank on the Scotian Shelf using impact and reference sites (Gordon et al. 2008). This was among the first to be conducted on offshore fishing grounds. In each impact area a heavy otter trawling disturbance (12-14 sets along the same line using an Engels 145 trawl) was created each year for three years. Surveys of seabed habitat and organisms were conducted before and after trawling each year in both the impact and reference sites using a variety of acoustic, imaging and sampling tools.

The Grand Bank experiment was conducted on a high-energy sandy seabed at 120-146 m depth. Significant immediate impacts were observed on the biomass of large surface-dwelling organisms such as snow crab, sand dollar, brittle star, sea urchin and soft coral, all of which showed significant declines (Prena et al. 1999). Impacts on organisms living in the sediment were relatively minor and restricted primarily to some species of worms (Kenchington et al. 2001). No impacts were observed on molluscs, and recovery of the impacted species was within a year.

The effects of repetitive otter trawling on a gravel bottom ecosystem were evaluated on Western Bank (Gordon et al. 2008). It is one of very few multi-year otter trawling impact experiments conducted on gravel seabeds, a habitat type that is generally regarded to be particularly vulnerable to bottom fishing gear. The study site was located within the 4TVW Haddock Nursery Area that has been closed to most types of trawling since its inception in 1987. The most impacted organisms were epifaunal species such as horse mussels and a tube-dwelling polychaete, which represented 90% of biomass initially, declining over the three-year experiment to 77%, indicating longer-term impacts from the annual trawling disturbance.

Both the Western Bank experiment and the Grand Bank experiment show that the impacts of otter trawling are greatest on epibenthic organisms that are exposed to direct contact with the gear. Collectively, the results clearly indicated that the immediate impacts of otter trawling were greater on the sandy seabed studied on the Grand Banks than on the gravel lag seabed studied on Western Bank. However, both the habitat and communities at the Grand Banks site recovered in approximately one year while recovery at the Western Bank site was longer and expected to take at least 5 years.

A similarly designed three-year experiment to investigate the impacts of hydraulic clam dredging was conducted on Banquereau, a deep (70–80 m) offshore sandy bank on the Scotian Shelf (Gilkinson et al. 2003, 2004, 2005). The furrows created by the dredges dramatically changed the micro-topography of the seabed. The furrows degraded with time and trapped empty shells, altering the surficial substrate. The furrows were still evident 10 years later, although they were only just detectable with sidescan sonar in most areas (Gilkinson et al. 2015). The density of clam burrows in the furrows was reduced by up to 90% and no signs of recovery were observed over three years. Immediately after dredging, most species decreased in abundance. Large numbers of propeller clams were excavated and subsequently died on the seabed surface. After two years, fast growing opportunistic species of worms and crustaceans were more abundant than before the disturbance. Target bivalves were reduced up to 67% and showed no signs of recovery after two years. The dredging disturbance had a major impact on the relative abundance of seabed organisms and full recovery was not evidenced until a decade later (Gilkinson et al. 2015).

A smaller-scale two-year scallop dredge experiment, examining a range of dredging intensities from zero to 14, was conducted in two nearshore areas of the Gulf of St. Lawrence in the 2000s, with sampling of endo and epibenthic animals before, following and a year subsequent the dredging (LeBlanc et al. 2015). The authors found few taxa that were significantly affected by the dredging both immediately and the year after. In contrast, short-term natural abundance fluctuations across experimental plots were much more prevalent and were of a magnitude similar to that estimated to be produced by fairly intense fishing, as would occur in only very limited geographic areas in the commercial fishery. The experiments were carried out in areas that had been closed to scallop fishing for some time, but that may be exposed to natural disturbances, such that resident taxa may be resilient to some disturbance. This situation provides somewhat of a contrast to that in the other studies above. Nonetheless, the general lack of dredging impacts estimated by LeBlanc et al. (2015), was in contrast to other studies in similar environments (e.g., Collie et al. 2000; Kaiser et al. 2006), which the authors, and others with similar findings (Pitcher et al. 2009), have attributed to weak experimental designs (e.g., low or no replication, confounded natural and experimental effects) used in many previous studies. The use of systematic review methodology (Hughes et al. 2014), which retains studies based on their quality and relevance, has resulted in more representative and robust meta-analyses of gear impact studies (Hiddink et al. 2017; Sciberras et al. 2018). Results from studies vetted by the systematic review of Hughes et al. (2014), or that meet criteria similar to theirs, are likely most suited for developing scientific advice involving the impacts of bottom-contacting gears.

4. BOTTOM-CONTACTING SURVEYS IN CANADIAN MARINE WATERS

In this section, we provide a summary of bottom-contacting surveys that are undertaken in marine waters off Canada, including the data that are collected during these surveys, the scientific advice that is supported by these data, and descriptors of the potential impact of surveys on benthic fauna and features. The summaries are restricted to surveys that employ bottom-contacting gear, and therefore exclude surveys that do not significantly contact the bottom as part of their normal operation, such as aerial, acoustic, dive and pelagic trawl surveys. The summaries are further limited to existing, ongoing and routinely occurring surveys. The summaries therefore also exclude one-off surveys aimed at characterizing the demersal and benthic biodiversity or geological characteristics of an area. The focus on ongoing and routine surveys is in part pragmatic, because the list of small scale one-off surveys or projects is potentially very long. The focus is also practical because the assessment of surveys regarding their admissibility in protected areas will generally be more straightforward for one-off surveys compared to ongoing surveys, where the suite of considerations and trade-offs identified in this report will generally be brought to bear in the decision making.

Based on these selection criteria, all selected surveys involve the use of some type of fishing gear and fishery resource monitoring is at least one of the survey objectives. There were no surveys that met the inclusion criteria that were conducted using video monitoring or benthic grabs or cores.

4.1 OVERVIEW OF EXISTING SURVEYS

Fifty-eight re-occurring bottom-contacting research surveys that take place in the coastal, shelf and slope waters off Canada were identified (Table 6). The majority of surveys are primarily intended to collect data and/or provide advice for single or a small group of related species, and of these approximately two-thirds target demersal invertebrate species. There are 22 surveys that are considered multi-species surveys or surveys that are regularly used in the provision of scientific advice for a number of species.

Twenty-six of the surveys are conducted using a bottom trawl, of which 15 are primarily annual multispecies surveys. The multispecies trawl surveys are among the longest running ecosystem monitoring surveys in Canada, with a number of surveys in operation since the 1970s (Chadwick et al. 2007). These synoptic surveys also cover the largest surface areas among the various surveys listed in Table 6, with a combined sampling frame of over 1.5 million km^2 after accounting for overlaps. Most multispecies trawl surveys are undertaken by DFO (Canada), though other countries involved in joint stock management with Canada also undertake trawl surveys on Georges Bank and the Scotian Shelf (United States) and parts of the Grand Banks and the Flemish Cap (Spain and Portugal).

There are seven major single species pot or trap surveys with the largest of these targeted to snow crab, principally on the Newfoundland and Labrador shelves (Table 6).

Single species scallop dredge surveys are undertaken by all DFO administrative regions with jurisdiction on the Pacific and Atlantic Coast (Table 6). These surveys are limited to scallop fishing grounds, with a total survey area of around 50,000 km^2.

Demersal longline surveys occur in Canada’s three oceans and are typically undertaken by commercial fishery partners. These surveys follow a scientific sampling design and are typically aimed towards a single groundfish species.

Finally, there are five surveys that employ bottom-set gillnets. Two are Sentinel surveys, conducted by industry partners with input from DFO on scientific sampling design, which have been in place since the mid-1990s in the northern Gulf of St. Lawrence and coastal Newfoundland. These surveys provide abundance indices for Atlantic Cod for areas that are often not accessible to the multispecies bottom-trawl as a result of the sea floor not being conducive to trawling. In addition to the Sentinel surveys, there is a gillnet survey undertaken by DFO in the Saguenay fjord and a small index-gillnet herring survey in the southern Gulf of St. Lawrence (sGSL). Note that index-gillnet herring surveys undertaken around Newfoundland were not included because contrary to the one in the sGSL, the gillnets are pelagic rather than set on the bottom.

4.2 INFORMATION AND SAMPLES COLLECTED BY SURVEYS

The most basic information collected by all of the surveys listed in Table 6 is the number and/or weight of each species captured in each tow, which can be used to estimate relative population size. Additional measurements undertaken on individual animals often include size and body weight, as well as the weight of individual organs that provide information on energy storage and reproductive investment. Furthermore, additional individual-level sampling may be undertaken for sex and maturity determination, the collection of tissues for age-determination and the collection of gut contents or tissues for diet analysis, among other parameters. Many of the surveys are also important platforms for synoptic physical, biological and chemical oceanographic sampling. The concurrent collection of these varied data at the same location is important for population and community modelling, and modelling habitat associations and shifts in species distributions in response to environmental change (Chouinard and Dutil 2011; Shackell et al. 2014; Pinski et al. 2013).

The surveys are all undertaken following a formal temporally-consistent sampling design. All mobile gear surveys except those directed at snow crab follow a stratified random design, where fishing locations are selected at random within each stratum. In contrast, most fixed gear surveys follow a fixed stations design (Table 6). Adherence to these sampling designs ensures that derived abundance indices are (ideally) unbiased and representative for the sampling area (domain). Furthermore, the designs generally help to ensure that the precision of estimates is as high as possible. When data are collected and analyzed based on the sampling designs, the resulting estimates will also be representative of the sampling area. As such, to the extent that a survey covers the distributional area for a population, information on sizes, ages, maturity status, diet and such can be used to derive estimates for those parameters that are representative of the population.

In addition to data that are routinely collected and used in the provision of scientific advice for resource management, the surveys commonly serve as the primary sampling platform for many specific scientific studies. These sampling requests from researchers within DFO and from a wide range of academic institutions are often so numerous that they need to be vetted in terms of priority prior to the start of the surveys to ensure that they can be completed without compromising the primary objectives of the surveys. The types of requests and target species are too wide-ranging to summarize here but increasingly include the collection of tissues for genetic and genomic analysis. Single species surveys in particular are often used as a platform for tagging of released animals. These tagging programs are used directly in stock assessments to help inform on the population size or on important demographic rates (e.g., mortality) or movement (migration) rates. They are also used to deploy tags for the characterization of animal movements and habitat use.

4.3 SCIENTIFIC ADVICE SUPPORTED BY EXISTING SURVEYS

The existing regular bottom-contacting surveys contribute to the creation of a large diversity of scientific advice. In many instances, the data from surveys are a cornerstone of that advice (Claytor et al. 2014). Surveys provide indices of abundance for analytical assessments and information on species diversity and species distribution, which is used in identifying conservation areas and developing ecosystem indicators. Furthermore, they are the sole source of information on abundance and distribution of secondary commercial species, which have no analytical assessment, and on bycatch species, which must be monitored in relation to the potential impacts of commercial fishing.

All of the existing regular surveys summarized in Table 6 collect demographic information on one or more species of present or former commercial interest. This information is used in stock assessments to provide scientific advice for sustainable fisheries management. Though assessments are not necessarily conducted annually, those that rely on survey-derived abundance indices typically require annual estimates of relative abundance. Advice on stock abundance, projected short-term trends and productivity is required for fisheries that are managed principally based on outputs (landings) using quotas. This advice is typically considered more precise and reliable when it is supported by long-term high quality fishery-independent abundance monitoring because surveys are more likely to track abundance (Hilborn and Walters 1992).

In the absence of scientific surveys, such advice relies on abundance indices derived from fishery-dependent data. Fishery-dependent indices often are not proportional to abundance and there is an elevated risk of failing to detect decreases in stock abundance (Hilborn and Walters 1992). Notably, an over reliance on commercial fishery catch rates that produced a positively biased view of the abundance of Northern cod in the 1980s due to increases in harvesting efficiency over time, contributed to overfishing and the collapse of the stock (Hutchings and Myers 1994; Walters and Maguire 1996). Similarly, changes in fishery catchability leading to positively biased fishery-dependent catch rate indices has led to serial overestimation of the abundance of the depleted NAFO 4T spring spawning herring stock (Swain 2016). Those results suggested that the stock was set to increase above its limit reference point, whereas recent analyses accounting for the source of the bias have estimated that the stock remains well below that level, in the critical zone of the Precautionary Approach, where fishing mortality should be kept to the lowest level possible. For secondary stocks and non-commercial bycatch species, catch rate indices from the commercial fishery are not generally informative on trends in abundance. The fisheries are not targeting these species and may actively avoid them, so fishery independent surveys are essential for monitoring these species.

Since 2000, the surveys have become important for the status assessment and recovery potential assessment of species of conservation concern, including species at risk of extinction listed under the Species at Risk Act. These assessments are not limited to species of commercial interest, and data from the multispecies bottom trawl surveys have increasingly become an important component (e.g., Swain et al. 2012a, 2015; Swain and Benoît 2017). Furthermore, under the federal-provincial *Accord for the Protection of Species at Risk*, there is a requirement to periodically review and report on the status of all wild species in Canada. These General Status reports have been produced every five years since 2000, and are used, among other things, by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC) as an aid in prioritizing its activities. Data from the multispecies surveys have played a large role in completing these reports (e.g., Benoît et al. 2003).

Indices of population, community and ecosystem status are derived from single and multispecies surveys for state-of-the-oceans (SOTO) reporting, both domestically and internationally. Ecosystem Status and Trend Reporting under Canada`s Biodiversity Outcomes Framework is used to measure progress towards the Convention on Biodiversity targets (DFO, 2010d). Reports compiled in 2010 for the major Pacific and Atlantic ecoregions all involved reporting of indices derived from bottom-contacting surveys (Cummins and Haigh 2010; Dufour et al. 2010; Johannessen and McCarter 2010; Templeman 2010; Worcester and Parker 2010). Similarly, more recent domestic state of the oceans reporting also involved such indices (e.g., Benoît et al. 2012).

The multispecies trawl surveys collect information on the relative abundance and distribution of a large number of fish and macro-invertebrate taxa. These data have been used extensively in producing science advice in support of DFO's activities related to marine spatial conservation planning. Notably, the trawl surveys figured prominently in the definition of significant areas for cold-water corals and sponges (e.g., Kenchington et al. 2016) and in the definition of Ecologically and Biologically Significant Areas (EBSA; Savenkoff et al. 2007; Ban et al. 2016; Wells et al. 2017). The data from these and other surveys can now be used for species distribution modelling, allowing for predictions in un-sampled areas that can then be verified using other means, such as video surveys.

Environmental assessments for large undertakings in Canadian marine waters have relied on the data from both single and multispecies bottom-contacting surveys to evaluate the ecological risks associated with those activities. Notably, these data are used routinely in environmental assessments related to the exploration and exploitation of oil and gas (e.g., Jamieson and Davies 2004; Swain and Benoît 2001).

4.4 THE IMPACTS OF SURVEYS ON BENTHIC ECOSYSTEMS OFF CANADA AND THE POTENTIAL FOR RECOVERY

The FAO (2009) guidelines for assessing the scale and significance of an impact caused by fishing gear, and recovery potential following the impact, suggest that the following six factors be evaluated:

1. the intensity or severity of the impact at the specific site being affected;
2. the spatial extent of the impact relative to the availability of the habitat type affected;
3. the sensitivity/vulnerability of the ecosystem to the impact;
4. the ability of an ecosystem to recover from harm, and the rate of such recovery;
5. the extent to which ecosystem functions may be altered by the impact; and,
6. the timing and duration of the impact relative to the period in which a species needs the habitat during one or more of its life history stages.

The first three factors characterize the direct impacts of fishing and nominal recovery following impact and are also pertinent for surveys. While the extent to which scientific surveys impact ecosystem functioning is not known, it will depend on the size of the survey footprint relative to the distributional areas of benthic taxa or features (or of the protected area), and the frequency at which the surveys impact these taxa or features, relative to their ability to regenerate and recover following disturbance. The proposed approach described in this subsection aims to explicitly account for FAO factors 1 to 4, though the treatment for each is not necessarily quantitative and some involve the use of a proxy. The last two FAO factors address the functionality of habitats post disturbance, and are much more difficult to assess due to data and knowledge limitations, including the difficulty of properly quantifying ecological functionality (DFO 2017b). Given an inability to directly address these, precaution is built into the proposed approach via the use of a buffer.

4.4.1 Footprint of Impact – Severity and Spatial Extent

Eigaard et al. (2016) and Rijnsdorp et al. (2016) developed a generic method to estimate the acute impacts of mobile fishing gear on benthic species and communities (see also ICES 2016). The method is based on basic principles and is therefore easily applicable to other bottom-contacting gear. Though the method was developed for evaluating the effects of commercial fishing, it is scalable and can therefore be applied to bottom-contacting surveys. Consistent with the extensive body of existing literature on benthic gear impact, the physical impact of bottom-contacting gears on seabed habitat and fauna is related to the penetration of the gear in the sediment, collision impacts, and sediment mobilization (Rijnsdorp et al. 2016). It starts with the characteristics of the gear (design and size) to estimate the interactions with the seabed for an individual fishing haul, and then evaluates impacts over all hauls in a fishery, or in this case, a survey.

The approach is based on estimating seabed pressure, here called the footprint, defined as the area and severity of seabed impact. The area impacted is a function of the area covered by a single haul (swept area) and the area affected by sediment resuspension from a haul, where relevant, summed over the distribution of all hauls. The swept area for a haul is a function of the size of the opening of the gear and distance towed during fishing for bottom-trawls and dredges. For fixed gear, 'swept area' can analogously be defined as the area of contact by the gear during fishing, and any area swept laterally or longitudinally during gear deployment and retrieval. Swept areas can be calculated by making some assumptions, which are described below. The majority of direct impacts occur within the swept area, and this is used as the areal basis of the footprint in Rijnsdorp et al. (2016). Impacts resulting from sediment resuspension will often affect a broader area depending on numerous factors such as the sediment type, sediment penetration by the gear, and local currents (O'Neill and Ivanovic 2016). The effects are much more difficult to quantify and are likely situation specific, though basic principles are mentioned below. In some instances involving taxa that are particularly susceptible, case-specific investigations could be required. Otherwise, a rough accounting for possible effects can be made when estimating the risks of long-term harm (see the discussion on a buffer in section 4.4.3.2).

In the approaches of Eigaard et al. (2016) and Rijnsdorp et al. (2016), the severity of seabed impact is related in part to the penetration depth of the gear and damage due to collision with the gear. While the authors focused on collision during fishing of mobile gear, crushing of organism when the gear is landed (fixed and mobile gear) and as a result of lateral and longitudinal gear movements during deployment and hauling (fixed gear) would also be relevant. Severity also depends on the characteristics of the taxa or features in and around the swept area as was described above (sections 3.2 and 3.3).

The methods used by Eigaard et al. (2016) to define the footprints of individual fishing hauls for the major mobile gear types used in surveys are presented below. The principles are then applied to determine footprints for the fixed-gear types and for other research gear types.

4.4.1.1 Mobile gear (general)

The degree of contact with the seabed of bottom-trawls and dredges depends on the design and rigging of the gear (details below), the speed at which the gear is towed, and the characteristics of the seabed, such as particle size (He and Winger 2010; Lucchetti and Sala 2012). The amount of seabed penetration of the gear or parts of the gear will affect the amount and diversity of taxa affected by a haul, with surface impacts limited to epibenthic taxa, and subsurface impacts associated in addition to endobenthic taxa. Rijnsdorp et al. (2016) modelled the penetration impact of mobile gear as a function of the product of the mass of the gear or gear component and inverse of the surface area that is in contact with the seabed.

All bottom-contacting fishing gear (fixed and mobile) will result in some re-suspension of sediment when they are landed on soft-sediment seabeds. In addition, mobile gear will mobilize sediment in the wake of the gear, with amounts depending principally on the sediment particle size distribution and the hydrodynamic drag of the gear (Lucchetti and Sala 2012; O'Neill and Ivanovic, 2016). The hydrodynamic drag is in turn determined by the square of the towing speed and by the frontal surface area of the gear (O'Neill and Summerbell 2011; O'Neill and Ivanovic 2016).

4.4.1.2 Bottom trawls (otter trawls)

A traditional single bottom trawl has three basic components of seabed impacts during a haul associated with the trawl doors, the sweep lines, and the footgear. The swept area in all three cases has a length defined by the haul length (generally on the order of 10^3 meters in surveys). The swept area width depends on the orientation of the trawl part with respect to the tow path, and is on the order of 1-5 meters for both doors, 10^1-10^2 meters for the pair of sweep lines and 10^1 meters for the footgear for a medium sized trawl of the type used in many research surveys. The doors penetrate the sediment the most, producing a narrow furrow that can be up to 35 cm deep in mud and up to 10 cm in coarser sediments for commercial trawls (Lucchetti and Sala 2012; Eigaard et al. 2016; O'Neill and Ivanovic 2016). The sweep lines represent the largest portion of the swept area, but appear to have the least impact on the seabed, with penetration in the top few centimeters if at all (Buhl-Mortensen et al. 2013; O'Neill and Ivanovic 2016). Even if the sweep lines do not penetrate the sediment, they can break or dislodge large or emergent benthic organism. The impact of the footgear depends on its type and configuration, and its penetration depth is typically small (up to 5 cm in mud) to nil (Eigaard et al. 2016).

4.4.1.3 Beam trawl

The swept area of a beam trawl haul is the product of the gear width and the tow length. The greatest penetration of the sediment, up to about 10 cm, occurs at the shoes of the beam, and as a result of a tickler chain if one is employed (Eigaard et al. 2016). The beam penetrates the sediment surface only. In their global meta-analysis, Sciberras et al. (2018) estimated an average penetration depth of 2.72 ± 0.72 (SE) cm.

4.4.1.4 Bottom dredges

The swept area of a dredge haul is also the product of the gear width and the tow length. Standard dredges used to capture epibenthic taxa such as scallops have been demonstrated to create furrows up to 6 cm deep in mud (Pranovi et al. 2000). Sciberras et al. (2018) estimated an average penetration depth of 5.47 ± 1.28 (SE) cm.

4.4.1.5 Hydraulic dredges

The swept area of a hydraulic dredge haul is also the product of the gear width and the tow length. The degree of sediment penetration is dictated by the dredge design. Sciberras et al. (2018) estimated an average penetration depth of 16.11 ± 3.35 (SE) cm. Hydraulic dredges are used in only one ongoing Canadian survey (Table 6) and the impact of dredging has been studied in detail in that survey area (Gilkinson et al. 2003, 2004, 2005, 2015). The results of those studies can directly inform on the impacts of that survey and subsequent recovery following sampling.

4.4.1.6 Fixed gear (general)

The swept area for fixed gear is expected to be the sum of the seabed areas covered by the gear during fishing operations and any area swept by lateral or longitudinal movements of the gear during deployment and retrieval, and possibly during fishing. Additional potential impacts on benthic organisms and features resulting from movements of lost and detached nets and lines in currents and tides are acknowledged but difficult to quantify. Furthermore, any impacts on mobile demersal fauna resulting from ghost fishing by lost gear are not addressed here.

4.4.1.7 Gillnets and longlines

The nominal surface area covered by longlines and gillnets during fishing is small, limited to the area in contact with the bottom and the anchors. Impacts to benthic organisms or features occur mainly as a result of gear movement. Longlines and gillnets are principally expected to have a shearing interaction with surface dwelling organisms when the gear moves (Ewing and Kilpatrick 2014), but contrary to the linear furrows in the seafloor created by trawls, the area impacted by longlines and gillnets is more haphazard (Stone 2006). Penetration of the seabed is likely to be limited, but rocks and boulders with attached invertebrates may be tipped (Stone 2006). In addition, longlines can hook and gillnets can entangle emergent organisms. Longline movements are largely restricted to the retrieval process and movements caused by the catch, whereas gillnet movements can additionally result from currents and tides. Gear movements during retrieval are a function of slackness in the gear, the length of the hauling line which increases with depth, a mismatch between the anchor location and the location of the vessel during hauling and the winds and sea state during hauling (Ewing and Kilpatrick 2014). Both longitudinal (along-axis) and lateral movements are possible. There is very little information on the magnitude of movements, though movements and therefore the size of the footprint at a site are expected to be highly variable. For the southern Ocean toothfish fishery, Ewing and Kilpatrick (2014) estimated the lateral movement of longlines with a mean of 6.2 m and a maximum of 31 m, though they believed that true values could be higher. When estimating the area swept by a single longline haul for the same fishery, Welsford et al. (2014) assumed a nominal width of 10 m and a length equivalent to the length of the gear on the bottom. In the absence of more studies, and given that swept areas are likely to be variable, we assume for the present purpose a precautionary swept area equivalent to the length of the gear multiplied by a nominal width of 100 m for gillnets and longlines.

4.4.1.8 Pots and traps

Pots and traps may crush or damage organisms when they first contact the bottom, but have been observed to be generally stationary during fishing (Doherty et al. 2018). The main movement occurs during retrieval and the area swept during the deployment and hauling of a string of pots or traps is expected to be greater than the total area swept by an equivalent number of units fished singly. While the line linking the units may be buoyant and not interact with the seafloor, pots and traps may be dragged along the seafloor during retrieval, creating a shallow furrow. Conical traps (1.37 m diameter) used in the Pacific sablefish fishery have an individual static footprint of 1.47 m^2; however, when they are retrieved they create an average individual swept area of 53 m^2 or 36 times the static footprint (Doherty et al. 2018). The authors estimated an average swept area for a 60-trap line of around 3,200 m^2 (interquartile range 1,500 to 6,000 m^2). Much of this swept area is from longitudinal movements and the furrows created by dragging the pots are on the order of 20 cm wide. In contrast, large box-shaped king crab pots (2.4 m x 2.4 m) can create relatively wider furrows (2 to 9 m) as a result of the longer edge that is in contact with the sediment compared to conical traps (Stone 2006).

The detailed study of Doherty et al. (2018) provides a direct estimate required to calculate the footprint of the DFO Pacific region sablefish survey, which fishes 25 traps in a 1,200 m line at approximately 48 m between pots. This results in a footprint of 1,325 m^2 (53 m^2 x 25) or 0.001325 km^2 per survey site. Conical pots are also used in snow crab surveys in eastern Canada and the spacing of pots along the line is similar to the sablefish traps, approximately 45 m between pots. Following Doherty et al. (2018), a swept area calculation of 36 times the static footprint multiplied by the number of pots is proposed: 36 x 10.18 m^2 x 10 = 3,664 m^2 or 0.003664 km^2 for the Newfoundland offshore post-season survey for example.

When this document was first presented at the peer review meeting of January 2018, the authors were not aware of the study by Doherty et al. (2018) and had proposed a precautionary swept area calculation for trap gear of 0.1 km assumed lateral (or equivalent longitudinal) sweep, multiplied by either the gear diameter (for single traps) or the total gear length (traps in a string). This assumption results in a much larger assumed swept area compared to estimates based on the results of Doherty et al. (2018). For the sablefish survey, this original assumption results in an swept area for a haul of 0.12 km^2, which is two orders of magnitude greater that the calculated value of 0.001325 km^2 based on Doherty et al. (2018). For the Newfoundland snow crab surveys, the original calculation results in haul-specific swept areas of 0.045 km^2 (for the post season offshore NAFO 2J3KLNOPs4R survey) and 0.036 km^2 (Coastal survey), compared to the revised calculations of 0.003664 km^2 and 0.002932 km^2, respectively. These comparisons suggest that the 0.1 km lateral sweep assumption was likely a highly precautionary assumption for conical traps, and the more precise empirical estimates are therefore used in Table 6 for surveys using this type of trap.

The ratio of the width of the furrows to the trap diameter for box-shaped traps in Stone (2006) is about 0.8 to 3.5, compared to 0.15 for the conical traps in Doherty et al. (2018). Though this inference is based on limited information, it suggests that the swept area for box-shaped traps could be about 5 to 23 times larger than a similar diameter conical trap, all else being equal. This difference suggests that the 0.1 km originally assumed lateral sweep may be roughly correct for box-shaped traps (e.g., prawn and lobster traps). In the absence of additional evidence, this general assumption was also used in Table 6 for other non-conical traps (e.g., the Dungeness crab survey).

4.4.1.9 Other scientific gear

The acute impacts of other scientific gear and instrumentation used to sample or observe benthic organisms, seabed properties, and sediments will depend on the area covered or swept by the gear and their action on the bottom. This will be best evaluated on a case-specific basis and is not discussed further here.

4.4.2 Footprint of Marine Bottom-Contacting Research Surveys Off Canada

4.4.2.1 Swept area

Following Eigaard et al. (2016) and the discussion above, the swept areas of individual hauls (in km^2) for the Canadian surveys were estimated as follows, with all distances in units of km:

- Bottom trawls: product of target tow distance and mean door spread;
- Dredges and Beam trawls: product of target tow distance and gear width;
- Longlines and gillnets: product of total gear length and 0.1 km;
- Pots and traps (conical): specific calculation based on Doherty et al. (2018);
- Pots and traps (other, singles): product of gear bottom diameter and 0.1 km;
- Pots and traps (other, in a string): product of total string length and 0.1 km.

For bottom trawls the total swept area based on door spread was assumed as a worst case scenario for endo-benthic taxa, which would be less or unaffected by the action of sweep lines, and as a correct approximation for emergent epibenthic taxa. Swept area based on the opening of the gear (wingspread) is reported in Table 6 because for some surveys door spread was not available and was calculated from wingspread assuming an average ratio of 3. This ratio is approximately the average value for the surveys in Table 6 that reported both wing and doorspread. For longline and gillnet gear, an average lateral movement of the gear of 100 m was assumed. For simplicity, the area covered by the anchors was assumed to be negligible with respect to the swept area of the gear and the lateral movement. For pots and traps, the total swept area calculation depended on whether the traps were conical or not, and whether fished singly or in a string, as described in section 4.4.1.8.

Research surveys conducted by, or in partnership with, DFO cover a broad surface area of Canadian marine waters (Table 6). The multispecies bottom-trawls surveys in Atlantic Canada alone annually cover an area of 1.138 million km^2 (Fig. 1), which represents over 65% of the area of the Newfoundland and Labrador, Gulf of St. Lawrence and Scotian Shelf bioregions (DFO 2009). While the survey areas are large for many surveys, annual survey swept areas are considerably smaller.

Estimated swept areas for individual hauls were greatest for the Atlantic Halibut longline surveys and other gillnet and longline surveys as a result of the extended length of the gear and the assumed 100 m lateral swept area (Table 6). The only mobile-gear survey with a similar swept area was the Banquereau Bank and Grand Banks hydraulic dredge surf clam survey (Table 6). The next largest swept areas were for the multispecies bottom trawl surveys conducted by the Gulf and Maritimes regions (0.14 km^2) (Table 6). Swept areas for the multispecies trawl surveys conducted in the Quebec and Newfoundland regions (0.068 km^2), were approximately half those of the Gulf and Maritimes region, reflecting target tow durations that are also half. The single species scallop dredge and snow crab trawl surveys are associated with some of the smallest swept areas, which are generally at least an order of magnitude smaller that the large scale multispecies trawl surveys.

The annual swept area of each survey was estimated as the product of the mean number of annual hauls, multiplied by the haul-specific swept area. Though some surveys such as the Sentinel gillnet and longline surveys fish repeatedly during the season at or near preselected locations, we assumed that random variation of deployment location resulted in non-overlapping hauls in a year. This assumption may exaggerate the annual swept area of these surveys and underestimate the severity of impacts, but is made here for simplicity. The estimated annual swept area of surveys varied from 0.1 to 0.2 km^2 for the Gulf Region herring index net and sea scallop dredge surveys respectively, to around 280 km^2 for the Banquereau Bank and Grand Banks hydraulic dredge surf clam survey when it takes place, which has been only occasionally and not since 2009-2010. Values for most other surveys were at scales of single to 10s of km^2, with annual swept areas of around 100 km^2 for the Atlantic Halibut and Newfoundland Sentinel gillnet surveys.

4.4.2.2 Impact

The footprint of impact for the surveys also depends on the severity of seabed impact. As described previously, the severity of impact is a function of the frequency of sampling at a given location and the impacts of individual hauls, which in turn depends on numerous factors such as the gear and how it is employed, and the sensitivity of the species, feature or habitat affected by the impact. A case-by-case evaluation of the potential severity of seabed impacts relevant to the environment in which the survey occurs would be required. While this may be feasible for some surveys, it will not be possible in many cases because the information is not available, or will otherwise be time consuming and likely to lead to imprecise evaluations. In light of this, it is simplest and more precautionary to assume that impacts are uniformly severe within the footprint of surveys and resulting in complete removal of biota or features, unless there is specific scientific information demonstrating otherwise for a given case.

The footprint of survey activities relative to the total area occupied by a sensitive taxon or type of feature (FAO principle (2), section 4.4) provides a measure of the potential harm caused by the activity, which increases as this ratio increases. Elevated risk of harm occurs, amongst other things, because of increased habitat fragmentation and ensuing decreased recolonization and recovery potential. This is particularly relevant for spatially restricted and unique taxa and features, where the risk of harm may be too high to allow surveys to proceed. Conversely, broadly distributed taxa or features with large spatial extent or common recurrence over the landscape, and for which the overlap with survey footprints is small, will be exposed to a lower risk.

In the context of protected areas and potentially multiple overlapping surveys, the following calculation of the proportion of the area impacted can be made. First, the mean annual number of sample stations within a protected area is the product of the sampling intensity (samples per km²) of the corresponding survey domain (or stratum) and the surface area of the protected area overlapped by the domain/stratum, summed over all overlapping strata in the case of a stratified random design. Here strata are considered, as sampling density may vary among strata. Second, the average proportion per year of the protected area which would be impacted by the bottom-contacting scientific sample stations in random or stratified-random surveys over all strata (K; with K = 1 for a random survey) for all surveys (S) is calculated as:

$$\text{Prop. Impact} = \frac{\sum_{s}^{S} \overline{swept\ area_s} * freq_s \sum_{k}^{K} sampling\ intensity_{s,k} * protected\ area\ size_{s,k}}{protected\ area\ size} \qquad (1)$$

where $\overline{swept\ area_s}$ is the average swept area for a sample (km²; calculated as described at the beginning of this section) in survey s, $freq_s$ is the annual frequency (1 for annual surveys, 0.5 for biennial surveys, etc.) of survey s, sampling $intensity_{s,k}$ is the average number of sampling stations per km² within stratum k for survey s, protected area $size_{s,k}$ is the surface area (km²) of the protected area contained in stratum k of survey s, and the denominator is the total surface area (km²) of the protected area. The proportion impacted has units of year-1.

The proportion of the bottom within a protected area that is impacted by a fixed station survey design will be constant in time and equal to the sum of the swept areas for all tows occurring in the area, divided by the surface area of the protected area.

4.4.3 Recovery Potential - Disturbance Frequency and Biological Traits

In the absence of specific directed studies, a quantitative metric of the long-term harm to species, communities, and benthic habitat forming structures (e.g. biogenic structures) resulting from scientific survey activities in a protected area is not possible at this time. However, the potential for recovery following disturbance is expected to be a function of the frequency of disturbance and the productive capacity of benthic taxa or features at an affected site. Disturbance frequencies that are small, or conversely disturbance recurrence intervals that are large, relative to productive capacity should allow for local recovery before the site is again disturbed, thereby avoiding long-term degradation on the broader scale, all else being equal. These principles are employed to define a proxy for long-term harm of scientific activities.

4.4.3.1 Disturbance frequency and recurrence time intervals

The average recurrence time interval (R_s; in years) for hauls of a given survey s at any given location, defined as the average time between successive benthic sampling impacts at a given site, is the inverse of the disturbance frequency and can be calculated as:

$$R_s = \frac{survey\ domain\ area}{\overline{swept\ area_s} * sampling\ intensity_s * freq_s} \qquad (2)$$

To the extent that survey sets are chosen randomly within the survey domain, the value of R_s will be true for any subset of that domain, for example within the boundaries of the protected area.

In the context of survey disturbance in protected areas, it is necessary to consider jointly the recurrence of all survey activities, i.e., the cumulative effects of multiple surveys. The recurrence time interval for an entire protected area is the inverse of the annual proportion impacted (from eq. 1):

$$R = \frac{1}{\text{Prop.Impact}} = \frac{\textit{protected area size}}{\sum_s^S \overline{swept\ area_s} * freq_s \sum_k^K\ sampling\ intensity_{s,k} * protected\ area\ size_{s,k}} \quad (3)$$

Furthermore, in instances in which surveys overlap only partially with a protected area, it may be more appropriate to consider the recurrence time interval for the portion of the protected area which is overlapped and therefore at risk of harm, such that:

$$R = \frac{\textit{protected area size * proportion protected area overlapped}}{\sum_s^S \overline{swept\ area_s} * freq_s \sum_k^K\ sampling\ intensity_{s,k} * protected\ area\ size_{s,k}} \quad (4)$$

In cases in which sampling intensities within surveys vary among strata, or in which the overlap between sampling frames and the protected area varies among surveys, the recurrence time will be heterogeneous in space. Though the estimate from equation (4) will be correct for the average parcel of overlap, more refined calculations would be required if characterization of this heterogeneity is desired.

The value of R as defined above applies to random-based survey designs. For fixed-station designs, the calculation and interpretation are different. At fixed station locations within the protected area R = $freq_s$ while at all other locations R is infinite because as long as the locations of fixed stations do not change in time, impacts will never recur at other locations.

It is not feasible to describe here the recurrence times for specific existing Canadian protected areas, though these should be calculated when evaluating the case-specific admissibility of survey activities. Instead, we present the estimated overall average recurrence times for the ongoing surveys off Canada, based on eq. 2. These assume a random selection of stations. In stratified random surveys, this assumption will be approximately true if the number of stations selected per stratum is proportional to stratum size. This is roughly the case, for example, in the southern Gulf of St. Lawrence multispecies trawl survey, but not so for the similar survey on the Scotian Shelf (Fig. 2; note the distribution of haul locations in the two areas). When the number of stations assigned per stratum is not proportional to stratum area, some strata will be associated with local recurrence time intervals that are shorter or longer than the survey average.

The shortest recurrence times were calculated for the Maritimes region sentinel surveys (630 to 800 years) and shrimp survey (758 years) (Table 6). Most other surveys were associated with average recurrence times on the order of thousands to tens of thousands of years. By far the longest recurrence time was for the spatially rotating southern Gulf scallop survey at over half a million years.

Differences in sampling density across strata in the Scotian Shelf multispecies surveys results in some strata with local recurrence times of around 1,000 years and others with times of over 10,000 years (Fig. 2). Spatial heterogeneity in station locations therefore require estimates of mean recurrence times that are specific to the area in which a protected area occurs when determining the potential impacts of a survey, as indicated above.

4.4.3.2 Lifespan and a precautionary buffer

Most taxa with lifespans that are shorter than recurrence times should, on average, have time to recover to the levels that existed prior to the impact of the survey haul (Thrush et al. 2005; Rijnsdorp et al. 2016). The known or expected lifespans of the most sensitive and difficult to recover taxon or feature (e.g., the age of a reef) should therefore provide an indication of the potential for recovery of the assemblage at the site or sites following disturbance by survey activities. However, there is uncertainty in the relationship between the lifespan of benthic taxa / feature and recovery times. First, longer recurrence times in some instances may be required to recover the size structure and ecological functions at the sites disturbed by the survey. Second, there are pertinent life history constraints likely present in benthic organisms that are expected to affect the lifespan-recovery time relationship such as episodic spawning and recruitment events, intermittent conditions for dispersal, and the time required for substrate or a benthic feature to become suitable for colonization. Third, recovery time and impact footprint can be affected by indirect effects of sampling gear that can be difficult to quantify, such as the impacts of sediment re-suspension. These are not included in the swept area and recurrence time calculations described above. Similarly, there is general uncertainty in the calculation of footprint, particularly for fixed gear, as discussed above.

To minimize the risks of overestimating recovery potential in light of these uncertainties, an order of magnitude precautionary buffer between lifespan and recurrence times is recommended. Where lifespans are at least one order of magnitude smaller that survey recurrence times, recovery appears likely at impacted sites and long-term degradation of the status of populations, communities or features in the protected area resulting from survey activities should be unlikely. Alternatively, where lifespans are of similar or greater magnitude as recurrence times, long-term degradation may be possible.

For those coral and sponge species with lifespans extending into 100s and 1000s of years (e.g., Freiwald et al. 2004; Sherwood and Edinger 2009; Clark et al. 2016), the recurrence interval of most surveys in Table 6 may be too short to guarantee avoiding long term degradation in a protected area. Conversely, despite being considered SBA taxa, sea pens are often considered relatively more resilient than other corals due to their shorter life spans and quicker growth rates, with expected recovery timescales of decades rather than centuries (DFO 2017b), well within the recurrence intervals of all individual surveys in Table 6. However, where multiple surveys overlap, recurrence intervals will be shorter and may not be long enough to avoid harm with a reasonable likelihood.

The relativity between longevity of the most sensitive taxa and recurrence interval, in light of the order of magnitude buffer, addresses FAO (2009) guidelines (4) and (5) (ability and rates of recovery, and alteration of ecological function) given considerable gaps in available information. In some instances findings from experimental manipulations or detailed comparative analyses will provide suitable and sufficient information to determine the time required to recover the status and ecological function of taxa or features disturbed by particular gears. In these instances, the available information should take precedence in scientific activity permitting decisions given the likely greater precision of the conclusions. Such is the case, for example, for hydraulic dredging, scientific or otherwise, in Atlantic Canada (Gilkinson et al. 2003, 2004, 2005).

4.4.4 Unsampled Areas in the Survey Domains

Within the survey domains of all Canadian multispecies trawl surveys, and likely many other mobile gear surveys, there are locations that are not sampled because of rough bottom or steep topography. These features, which present elevated risk of damaging the gear or are too constrained to accommodate a haul of standard length, are also typically avoided by commercial fisheries employing the same gear. These locations effectively become *de facto* closed areas, and benefits similar to those observed for marine protected areas have been observed there (Link and Demarest 2003). By the nature of the seafloor and topography, these are also areas where vulnerable benthic taxa may have longstanding refuge (Clark et al. 2016). The natural exclusion of trawl surveys from these types of areas was one of the factors that contributed to the decision to allow bottom-trawl surveys to continue following the northeast USA programmatic environmental assessment for the area (URGS Group 2016). Documenting the location of these areas for Canadian survey should be a priority. The same is true for locations that have resulted in catches of corals or sponges that were so large that the survey haul was considered as an invalid sample for other taxa. First, documenting these locations will avoid inadvertent, ineffective and potentially highly damaging sampling from re-occurring. Second, these areas may constitute important candidates for protection from other activities that could harm the resident benthic communities.

4.5 ALTERNATIVE MONITORING METHODS, GEAR MODIFICATIONS AND PROCEDURES MITIGATING POSSIBLE IMPACTS ON BENTHIC ECOSYSTEMS

A switch to alternative monitoring methods or modifications to existing survey gear or survey procedures may be used to mitigate impacts of bottom-contacting gear on benthic ecosystems and may be an option for continuing surveys in closed areas. In this section, we review options for mitigation and discuss the important considerations concerning changes to survey gear or procedures.

4.5.1 Options for Monitoring Tools

For the purposes of this report, sampling tools are divided into two classes. Extractive survey methods are those that employ fishing gear and benthic grabs or cores to extract biota and sediments for ship or land-based processing. Observational survey methods are those based on the non-extractive sampling of marine organisms and include visual survey methods using divers, video monitoring, acoustic surveying and remote sensing (satellite telemetry).

Murphy and Jenkins (2010) provide a review of methods for monitoring the abundance and distribution of fish and associated habitats. A summary of their review, with additional considerations from the present authors, is presented in Table 7 for broad classes of survey methodologies and is further discussed below. The summary is limited to methods that can provide the type of data required for the various kinds of ongoing scientific advice presently provided by DFO. Methods for characterizing physical habitat features (e.g., multibeam echosounding), or for taxa that are not sampled by the routine surveys, such as small benthic invertebrate (e.g., grabs and cores), are not discussed for parsimony.

4.5.1.1 Less impactful fishing gear (extractive sampling methods)

A change to less impactful fishing gear is a potential option for mitigating the effects of a survey on benthic organisms and features (e.g., Morgan and Chuenpagdee 2003; Fuller et al. 2008). However, there can be important trade-offs in the data that are collected, which may render a gear change an unfavorable or unviable option. Furthermore, changes in survey gear require calibration to ensure the integrity of survey time series, which is an important consideration discussed in section 4.6.

Bottom-trawls and dredges rank the highest in terms of the anticipated severity of their impact on both physical habitat and benthic biota (Morgan and Chuenpagdee 2003; Fuller et al. 2008). There is nonetheless scope for gear modifications that could greatly reduce the impact of bottom-trawls and non-hydraulic dredges (Valdemarsen et al. 2007). Three basic methods to reduce the bottom impact of a ground gear are:

- reduce the length of the ground gear and thus the width of the trawl, thereby reducing the swept area;
- reduce the physical pressure from gear components on the bottom, for example by reducing the weight of gear components (e.g., doors); and
- reduce the number of bottom contact points along the ground gear.

For example, Nguyen et al. (2015) compared the effectiveness of a reduced seabed impact footgear versus a traditional rockhopper footgear on identical bottom trawls targeting northern shrimp (*Pandalus borealis*) in Newfoundland and Labrador. The experimental trawl used in their study was designed to have low seabed impact by replacing the traditional heavy rockhopper footgear with a few drop chains lightly in contact with the seabed. Preliminary results were promising with respect to engineering and catch performance and could help reduce potential disturbance to the benthos, particularly minimizing encounters with snow crab. Similarly, a shrimp trawl employing footgear composed of small rubber disks was demonstrated to have significantly lower seabed impact yet improved performance catching shrimp and finfish (Winger et al. 2017). In the Gulf of Maine a raised footrope was developed for the small mesh whiting fisheries (Valdemarsen et al. 2007). This trawl is referred to as a sweepless trawl and uses drop chains scattered along the fishing line to greatly reduce bottom contact. This ground gear has apparently been adopted voluntarily by industry. However, while such modifications have been successful in commercial fisheries where there is a desire to maximize wanted catch and minimize bycatch, these will often not be the objectives of multispecies surveys.

Bottom- trawls in particular, as well as dredges, are the least selective with respect to the capture of different species and size of marine macro-organisms (Millar and Fryer 1999). This property, along with large swept areas that facilitate the sampling of low-density biota, have made bottom-trawls the gear of choice for multi-species marine surveys worldwide (Doubleday and Rivard 1981; Cooper et al. 2004; Jouffre et al. 2010; ICES 2012a; Axelsen and Johnsen 2015). Fixed-gear alternatives do not capture the diversity of species that are currently monitored by multispecies trawl surveys. Fixed fishing gears also tend to be more size selective, capturing a narrower range of sizes of fish and invertebrates (Millar and Fryer 1999). This greater selectivity has important negative consequences for age- and size-based stock assessment models and assessment frameworks that rely on both an index of incoming recruitment and an index of adult biomass or abundance. The greater selectivity would also limit the ability of a survey to simultaneously track the abundance of both smaller-bodied forage species and larger demersal animals, thereby limiting the monitoring of ecosystem-level change.

Mid-water trawls, though much less impactful (DFO 2010c), do not constitute a viable alternative to bottom-trawls in a demersal multispecies survey. Mid-water trawls sample a different assemblage of fish species of bathypelagic or pelagic distribution and typically do not contact the bottom during normal fishing operations, unless they are fished closed to the bottom as is the practice in some fisheries outside of Canada (Tingley 2014; NMFS 2015). The use of mid-water trawls fishing in mid-water, likely combined with an acoustic survey, may however be a potential alternative to the use of bottom-trawls for the single-species monitoring of certain bathypelagic taxa such as redfish (*Sebastes* spp.) in the Atlantic.

The overall impact of bottom-set gillnets and longlines, and pots and traps, on physical benthic habitats is considered low to moderate, but impacts on emergent sensitive taxa such as corals and sponges can be elevated (Morgan and Chuenpagdee 2003; Fuller et al. 2008) and there may be a requirement for impact mitigation in existing surveys that employ these gear types. As discussed previously, threats related to these gears generally occur during deployment and retrieval, or as a result of gear loss. Possible mitigation measures include deployment only under favourable weather conditions, the used of biodegradable materials and shortening soak times (DFO 2010c). An advantage of these fixed gears for monitoring is that they can be employed in habitats that cannot be sampled by bottom-trawls (e.g., jagged rocky bottoms or areas of rapidly varying topography). Furthermore, fixed gear surveys using hook and line or trap gear generally capture animals with the least amount of harm and are therefore often used in tagging studies (e.g., Cadigan and Brattey 2006).

Hook and line gears, such as handlines, rod and reel, and trolls, generally pose a low risk to benthic biota and physical habitat (Morgan and Chuenpagdee 2003; Fuller et al. 2008). However, this gear type does not provide a viable alternative survey sampling method in a large majority of cases. Like the other fixed gear methods, hook and line gear is relatively species and size selective (e.g., Millar and Fryer 1999). In a survey context, such gear is likely to be limited to medium to large pelagic species, particularly ones that may be difficult to sample by other means such as large sharks (Murphy and Jenkins 2010). For taxa that would otherwise be better sampled using other bottom-contacting gear, considerable sampling effort would likely be required to sample sufficient numbers of animals over the distributional area of the targeted populations to be able to reliably discern changes in relative abundance from sampling error. The required effort would likely render the approach unfeasible in most situations.

Egg and larval surveys are used worldwide to provide estimates of spawning fish abundance for some stocks, including Atlantic Mackerel and Pacific Herring in Canada (Smith and Richardson 1977; Schweigert 2001; Grégoire and Beaudin 2014). For pelagic eggs, sampling involves no contact with the bottom, whereas for demersal eggs localized selective sampling is required. The method effectively involves scaling egg densities as a function of spawner characteristics (e.g., age or length-dependent fecundity and maturity schedules). The method is often employed for stocks for which it is difficult to obtain abundance indices from trawl surveys, such as pelagic fish. Because of specificity in spawning season and the horizontal and vertical distribution of the eggs and larvae, combined with the need to collect ancillary data on spawner characteristic, this approach is used principally in a single species context.

4.5.1.2 Observational methods

The principal advantages of diver-based underwater visual methods (UVM) and underwater imagery methods are their low or no impact on habitats, little or no need to extract organisms, and their potential for high sampling density at fine spatial scales and for describing bottom habitats (Table 7). This is particularly useful in the context of research on small scale associations between species and habitat, and on selected characteristics of behavior, which are major fields of activity for underwater observational methods (Murphy and Jenkins 2010; Mallet and Pelletier 2014; Brandt et al. 2016). Underwater imagery has also been demonstrated to be superior to fixed-gear capture-based methods for surveying inactive fish (Morrison and Carbines 2006). Imagery-based methods can be deployed as a drop camera for stationary imaging (each still image represents a videographic sample) or continuous recording, often using bait, as a towed unit with continuous video recording along transects, or using remotely operated or autonomous underwater vehicles. Observational methods based on SCUBA, as well as remotely-operated vehicles (ROV), also allow for real-time adaptive sampling, and specific features can be viewed in detail. Furthermore, diver and ROV based methods allow for the selection and recovery of specific, and potentially small and delicate objects more precisely than any other sampling method (Brandt et al. 2016).

The use of a diversity of video-based methods has increased in the USA, and has principally focused on single species monitoring and to describe demersal habitats and communities (National Research Council 2015). Video based surveys have proven to be viable approaches for monitoring the relative abundance and distribution of scallops in a number of settings (Rosenkranz and Byersdorfer 2004; Stokesbury 2002; Stokesbury et al. 2004; Singh et al. 2014), including Pink and Spiny Scallops stock assessment in the Pacific Region (Surry et al. 2012). However, despite being superior to dredge surveys for detecting scallop recruitment, they have not replaced the traditional surveys in the NE USA because dredge surveys are considered superior for estimating length composition, distinguishing live and dead scallops and obtaining information on physiological and life-history attributes of individual scallops, amongst other considerations (Cryer 2015). Effective trawl-based video systems have been developed for surveys of larger-bodied groundfish such as cod (DeCelles et al. 2017; Stokesbury et al. 2017). While these allow for monitoring without capture, thereby reducing harm to fish, the trawls nonetheless employ doors and footgear, and therefore do not necessarily reduce harm to benthic organisms or features. Baited video systems have also proved useful for sampling reef fish as well as large highly mobile species that may not be well sampled otherwise (Mallet and Pelletier 2014; Devine et al. 2018). Appropriate standardization of the area of attraction of the bait, particularly where there are currents (Taylor et al. 2013) and standardized deployment times (Coghlan et al. 2017) are required, among other factors, for quantitative assessments. Underwater video methods have the advantage of providing a permanent record of the survey that can be revisited in the case of data uncertainties or if research foci change (Murphy and Jenkins 2010). This advantage comes with a high cost associated with post-survey image processing, although developments in automated image processing are reducing these costs (National Research Council 2015).

The US National Research Council (2015) report states that the main challenges with image analysis in fisheries stock assessment include accurate identification of species, unclassified targets, movement, double counting of fish, and catchability. Murphy and Jenkins (2010) identified the following disadvantages associated with UVM and video methods; their restricted field of view resulting in limited spatial coverage at a sampling site and therefore enhanced among-site sampling variability, the potential for biased abundance estimation; and, observations that are limited to conspicuous species (Willis 2001) and in waters with high visibility (Table 7). Densities of some species, such as certain groundfish, can be very low, rendering existing video-based methods other than trawl-mounted video potentially ineffective (see chapter 3 in National Research Council 2015). Where video-based methods have been applied in a large scale survey context, they have produced conflicting results with longstanding trawl surveys (MacDonald et al. 2010). UVM are constrained by restrictions related to the safety of divers, and therefore are possible only in certain areas such as shallower depths, under certain conditions with respect to tides and sea-state and are of limited duration. The deployment and retrieval of underwater video gear, and ROVs in particular, can be time consuming, resulting in constraints on the number of sites that can be visited.

The above considerations likely explain why there appear to presently be no large-scale video-based surveys anywhere in the world employed for routine multispecies monitoring of mobile fauna, though small scale surveys of reef systems and of more sedentary fishes in complex habitats have been undertaken (see review by Mallet and Pelletier 2014; Yoklavich et al. 2007; Haggarty et al. 2016; Wilson et al. 2018). However, the development and application of underwater video monitoring is a rapidly growing field (Mallet and Pelletier 2014) and suitable technology and methods could become available in the near future. Provided that they can be calibrated to existing capture-based monitoring methods and that they can be shown to reliably track changes in relative abundance over time, for example by tracking the abundance of individual cohorts (e.g., Benoît et al. 2009; Swain et al. 2012b), these could become valuable low-impact monitoring methods for surveys in general, let alone in areas with sensitive benthic features.

Hydroacoustics methods comprise the other main observational method for marine surveys. They are routinely used by DFO to survey specific pelagic or bathypelagic fish species such as Atlantic Herring, Capelin and Pacific Hake (Martell 2009; Mowbray 2014; McDermid et al. 2016), and appear well adapted for routine surveys of krill (McQinn et al. 2016). In fact, hydroacoustics are particularly well suited for sampling species that are difficult to sample reliably and efficiently using other sampling methods. Hydroacoustics can be used in a broad range of habitats, including untrawlable areas, and surveys can be undertaken over broad spatial areas. In some applications they can provide estimates of absolute abundance, whereas all other survey methods provide estimates of relative abundance, given that only a fraction of the biota that are in the sampling location or corridor are observed due to factors such as gear or observer avoidance (additional details provided below). The main disadvantage of hydroacoustics is that they cannot reliably detect demersal species near or on the seafloor, and fish lacking swim bladders are not sampled as effectively (Simmonds and MacLennan 2005). They are therefore not a suitable sampling method for flatfish and many other demersal groundfish species. Furthermore, the ability to distinguish species and sizes of fish is limited, and additional sampling using a trawl or underwater video (Jones et al. 2012; Boldt et al. 2018) is required to calibrate the acoustic signal and to apportion the estimated biomass among species and groups.

4.6 CONSIDERATIONS FOR CHANGING MONITORING TOOLS IN AN EXISTING SURVEY

Time series data are the foundation of fisheries science, and consistency in the way that these data are collected (or some means to compare and convert data between collection methods) is paramount in the ability to use this data to provide scientific advice for the management of marine resources. Any substantial change in survey methodology can result in newly collected data not being comparable to the previous time series. Even though two different survey gear types may provide data or information on the same properties, these cannot necessarily be used interchangeably due to the nature by which data or samples are obtained. Changes in survey gear/methodology can therefore be a significant obstacle in the provision of scientific advice unless these changes are properly calibrated.

Of paramount importance in understanding why changes in survey gear/methodology are an issue is the fact that most surveys, including all of those listed in Table 6, provide estimates of abundance that are relative rather than absolute. These estimates are therefore typically referred to as indices of abundance, rather than abundance proper. The surveys capture and retain some proportion of the animals that are in the area swept by mobile survey gear (bottom trawls and dredges) or in the vicinity of fixed gear (gillnets, longlines, pots and traps). This proportion, often called catchability, typically varies among species and sizes of organism within a survey, but can also vary due to other factors such as depth and time of day. The utility of a survey in providing a reliable index of abundance for a species depends critically on the temporal stability of catchability. Changes to survey protocols or operations that cause a systematic change in catchability, will result in a change in the abundance indices. Failure to account for these changes risks confounding actual changes in abundance and changes due to survey operations, when interpreting patterns in relative abundance from a survey (Katsanevakis et al. 2012). It is for this reason that every effort is made to maintain consistency in survey design. When changes in survey operations are unavoidable or deemed necessary, such as a switch from day-only to 24-hr surveying (e.g., Benoît and Swain 2003) or a change in survey vessel or trawl (e.g., Miller 2013) considerable efforts are undertaken to standardize or calibrate abundance indices.

Differences in survey gear configuration and composition (e.g, mesh or hook sizes) within the same gear type (e.g., trawl, gillnet) can result in very large differences in relative catchability (Bourdages et al. 2007; Surette et al. 2016). Even when calibration studies are undertaken, there can be size and species of animals that are effectively captured by only one of the gear configurations (Warren et al. 1997). In these cases, survey series cannot be adjusted reliably, resulting in a break in the relative abundance indices, or the need to establish new indices based on sizes or species that are adequately sampled by both configurations (Brattey et al. 1999). Changes to an entirely different sampling gear type is likely to result in even greater differences in catchability, with a greater number of instances where calibration is not possible. Willis (2013) reviewed the monitoring programs used in New Zealand marine protected areas and concluded that a lack of inter-comparability among data sources was common and often limited abilities to gauge the effectiveness of the protection. It will be important for managers to link conservation objectives with a monitoring plan that is properly designed such that effectiveness can be evaluated in future. This will require using the same sampling tool inside and outside of the closure.

All of the surveys listed in Table 6 use extractive methods and are founded on the capture and enumeration of organisms. The availability of captured specimens allows for the collection of data that is not directly possible using purely observational survey methodology (acoustic and video). This includes, but is not limited to, information on sex, reproductive status and age composition, species or stock composition (in the case of cryptic species or mixed stocks), food habits and health indicators such as condition indices and parasite loads. Furthermore, not all observational survey methods provide high resolution information on size or length composition, and when they do, individual masses needs to be inferred indirectly from length-mass relationships (Murphy and Jenkins 2010). For some of the parameters noted above, it may be possible to infer their values from survey samples collected outside the protected area, provided that those values can be considered representative (see section 5.2.1.3). In other cases, values specific to the area may be required. This will be true if conservation objectives for a protected area involve monitoring area-specific parameters, such as trophic relationships or demographic or population composition.

4.7 CHANGES IN SURVEY PROCEDURES

Survey procedures can be modified to reduce the footprint of individual survey hauls, notably by shortening tow distances in mobile-gear surveys. Furthermore, reductions in tow speeds can reduce the degree of penetration of mobile gear into the sediment and the amount of sediment resuspension (O'Neill and Ivanovic 2016), thereby reducing impacts on endo-benthos and on sensitive neighboring species.

Like changes in survey gear, changes in survey procedures require calibration to ensure the integrity of survey series. Many species of fish swim ahead of bottom-trawls until they become exhausted and fall into the net (Wardle 1986). Swimming endurance, and hence catchability, has been found to be a function of tow speeds and durations, among other factors (Wardle 1986; Engås 1994; Winger et al. 2000).

Furthermore, as the duration of a standard trawl haul decreases, the time spent by the net in the water column during deployment and retrieval becomes relatively more important. While in the water column, the net may capture fish, but likely not in a consistent manner because of variable deployment/retrieval durations and variations in net opening caused by currents and tides. Variability in catches is therefore expected to become greater as the duration of a standard haul is shortened, all else being equal.

It is for these reasons that haul duration within a survey is standardized and that there are pre-defined criteria for acceptable duration when hauls must be terminated pre-emptively, as for example when rough bottom is encountered (Hurlbut and Clay 1990). Survey hauls that do not meet these pre-defined criteria are considered invalid and the data provided by such hauls are not used in subsequent analyses of resource status. Hence, large reductions in survey haul durations within protected areas aiming to minimize harm on benthic taxa will therefore generally not be feasible as a mitigation measure for the fisheries-independent surveys. However, reductions in swept areas by up to one-third are possible given existing survey protocols.

The overall footprint of surveys can also be reduced by decreasing the density of sets in the protected area. For a single survey, this amounts to defining the protected area as a new stratum, or two or more strata if the area contains contrasting habitats that are expected to affect animal densities, and redefining the areas of the strata that formerly overlapped with the protected area. The overall survey sampling design will remain valid provided there is sufficient sampling in the new strata to estimate variance in density (minimally two sets, ideally at least three) (Krebs 1989). Local reductions in sampling density are also feasible for non-stratified survey designs, provided that a suitable model (e.g., geostatistical model such as in Thorson et al. 2015) is employed to estimate density. Whether the design is stratified or not, the estimation should remain unbiased provided the assumptions of the estimation method are met. However, reductions in sampling density will result in a loss of precision for the estimates. The magnitude of this loss will depend on the variability in survey catches in the less sampled areas and the size of these areas relative to the remaining survey area. A retrospective analysis of existing data involving re-stratification and empirical simulations of sampling density reduction could be used to evaluate effects on precision.

Set density within a protected area can also be reduced by removing redundant spatially overlapping surveys that largely provide the same information. If the information provided by the surveys in the area is (nearly) identical, as for example Sentinel mobile and research vessel multispecies surveys in the Gulf of St. Lawrence (Swain et al. 2012b) and particularly if the surveys can be intercalibrated for all the taxa that they monitor (see Benoît and Cadigan 2013 for snow crab), then this can be an effective means of lowering the overall footprint of surveys in protected areas. However, the ability to intercalibrate is not guaranteed and considerable model testing would be required before deciding to drop surveys.

Sensitive benthic features may be heterogeneously distributed within a protected area. To the extent that the locations of these features are known based on past observations, or inferred based on habitat characteristics, it might be possible to continue survey sampling in the other portions of the protected area, while substantially reducing possible harm to the features. If the area occupied by the features is small, there should be little consequence to the integrity of survey series.

5. EXCLUDING BOTTOM-CONTACTING SURVEYS FROM PROTECTED AREAS: CONSEQUENCES FOR SCIENTIFIC UNDERSTANDING AND CONSERVATION OBJECTIVES

Regular ongoing scientific surveys provide monitoring for temporal changes in the abundance and distribution of marine taxa. Ecological monitoring is required within protected areas to ensure the efficacy of the imposed management measures with respect to the defined conservation objectives for the areas (Hilborn et al. 2004). This monitoring could come, in whole or in part, from existing surveys or from new targetted surveys. In the broader ecosystem, monitoring is required to evaluate the efficacy of management measures employed to meet objectives related to the sustainable use of renewable marine resources and the recovery of depleted species and species of conservation concern. Monitoring is also crucial for evaluating ecosystem-level effects of human activities (Carstensen 2014), and for understanding the consequence of large-scale environmental changes like climate change and ocean acidification, which in turn will inform adaptation planning (Warburton et al. 2013). The inability to maintain time series of relative abundance estimates based on monitoring within protected areas could severely compromise the ability to correctly assess the status of populations (Punt and Methot 2004; Field et al. 2006).

In this section, we review the potential consequences of excluding current ongoing surveys from protected areas. In doing so, we also highlight the benefit to scientific understanding of allowing these surveys to continue in protected areas. We consider consequences for monitoring programs and decision making with respect to the conservation of species and ecological communities, within and outside protected areas, and the conservation of sustainable and prosperous fisheries.

5.1 SCIENCE AND CONSERVATION WITHIN A PROTECTED AREA

The successful application of a protected area depends, among other things, on a thorough evaluation of its efficacy (Hilborn et al. 2004). Most protected areas to date have lacked the data to objectively gauge the performance of the closure, thereby limiting opportunities for adaptability, which in turn compromises success (Jones 2001; Pomeroy et al. 2005; Fox et al. 2014; Stanley et al. 2015). This also risks maintaining an ineffective management measure that potentially limits opportunities for commercial fishing or other human activities.

Considerations for the planning of new monitoring programs within protected areas are reviewed in detail in Stanley et al. (2015). Here we consider specifically the potential role and potential relevance of existing routine surveys for monitoring the efficacy of some, but likely not all, new protected areas that overlap with existing surveys.

Gauging the efficacy of a protected area requires monitoring with sufficient replication, consideration of natural and sampling variability, and a comparison between sites within and outside the area (Hilborn et al. 2004; Stanley et al. 2015). Given low frequency ecological variability and the potentially long lags involved in responses following management actions (reviewed in Stanley et al. 2015 for MPAs; Hutchings and Reynolds 2004), long-term monitoring is particularly valuable. Where spillover effects of the protected area are expected, comparisons outside the area should involve sites at different distances from the area.

The most robust designs for protected area monitoring involve some variant of a before-after-control-impact (BACI) design (Underwood 1992, 1994), where, in the present context, impact refers to changes in properties of interest as a result of the new protection. The BACI design involves representative sampling in the protected area and at unprotected ‘control’ sites, prior to and following the implementation of protection. This design allows for unambiguous conclusions about the efficacy of the management measure, by controlling for larger scale spatial and temporal variability, provided that prior to the management measure, the control sites were similar to the future protected site with respect to the properties that are the target of the conservation effort. A BACI design that includes temporally and spatially replicated sampling before and after the implementation of the protected area, termed Beyond-BACI, can account for temporal trends in variation and spatial variability, and is considered an optimal approach for evaluating the effects of protection (Osenberg et al. 2010; Willis 2013).

In contrast to BACI-type designs, simpler control-impact (CI) designs that only compare properties in the protected area and at control sites, once the protection is in place, lack the ability to unambiguously differentiate between management impacts and natural differences (‘site effects’) that may have existed between sites prior to the implementation of protection (Osenberg et al. 2010). Early meta-analyses of the efficacy of marine protected areas were dominated by studies that were based on CI designs (Halpern and Warner 2002), which overestimated the benefits of protection for a number of situations (briefly reviewed in Stanley et al. 2015). This stands to reason given that the siting of protected areas is often based on one or more significant features that differentiate the area from other sites, such as the aggregation of corals and sponges or the presence of an EBSA. Like CI designs, before-after-impact (BAI) designs also do not allow for unambiguous determination of the efficacy of protection. In the absence of control sites, differences in properties of interest in the protected area before and after the implementation of protection cannot uniquely be attributed to the management intervention, but can be the result of unrelated natural or larger scale variation (e.g., LeBlanc et al. 2015). In cases where protected area boundaries encompass site-specific habitats or features completely or almost completely (e.g., Hecate Strait and Queen Charlotte Sound Glass Sponge Reefs and Endeavour Hydrothermal Vents MPAs), it may not be possible to select matching control sites outside of the protected area and thus BAI design may be appropriate.

Based on the preceding, efficacy monitoring for new protected areas in Canadian marine waters will ideally require monitoring prior to their implementation and covering an area that is broader than the protected areas. Ideally this monitoring will have begun several years prior to and extend several years following closure to maximize the power to detect changes in the protected areas resulting from the protection, in light of natural or background temporal variability. However, the tight timelines along which Canada reached its 2017 interim target and is striving to reach its 2020 target for protected areas are such that adequate new monitoring programs are unlikely to be in place in time for broad spatial scale pre-impact monitoring (but see Dunham et al. 2018a). Existing spatially and temporally replicated monitoring data are likely to be limited to those from existing ongoing surveys.

For a number of protected areas, existing ongoing surveys will be the only source of data from which to establish baseline conditions that can effectively serve as reference points. In most of these instances, the continuation of the surveys would be required to gauge future ecological changes against these reference points. While new, less impactful surveys could be implemented, it may be very difficult to calibrate these surveys to existing ones (see section 4.6 above; Considerations for changing monitoring tools in an existing survey).

There will be benthic taxa or features that are not appropriately or adequately sampled by the existing surveys. These could include encrusting taxa, taxa for which catchability to the survey gear is highly variable, and taxa for which sampling in and around a protected area is too sparse to provide a meaningful measure of density. For these cases, the existing survey data will be of limited or no benefit to monitoring the efficacy of spatial protection measures. However, within these same protected areas there may be other taxa that are well monitored by the existing surveys and for which ongoing monitoring is required as a result of additional related conservation objectives or a desire to monitor overall community and ecosystem dynamics resulting from protection. The relevance of existing surveys for monitoring trends related to new protection for various taxa can already be evaluated before protection is implemented via an analysis of the statistical power available to detect changes in the area, using existing data. Power analysis is an objective means by which to determine, which, if any, of the ecological components in a protected area are sufficiently well monitored by existing surveys. As a result, power analyses would constitute an important element in evaluating whether existing surveys can provide a benefit to monitoring the outcomes of protection that outweighs the harm to valued benthic species and features caused by the surveys.

In addition to providing data on the abundance and distribution of taxa, the existing routine surveys also collect biological samples that can provide data to inform other parameters of interest such as trophic relationships and demographic and genetic characteristics of taxa in a protected area. These may include parameters for which it is not possible to sample sufficiently, representatively or at the correct scale using less invasive means such as ROVs (e.g., samples from mobile species or those requiring sampling of a large number of organisms such as to estimate demographic characteristics). Furthermore, survey designs for the existing surveys are such that inference for studied parameters can correctly be made for a specific area or for the survey area as a whole. That is, the estimated mean and variance (for example) for the properties will be representative for the target area of inference. Consequently, from the samples obtained during the surveys it will be possible to evaluate the extent to which protection results in changes in functional roles among species (e.g., predator-prey relationships) or changes in biodiversity characteristics beyond those sampled macroscopically (e.g., sub-population structure, or spillover effects evaluated using genetics). Again, the benefits associated with this information should be evaluated relative to the harm caused by the survey activities when determining if the activities should continue.

5.2 SCIENCE AND CONSERVATION IN THE BROADER ECOSYSTEM

5.2.1 Survey Bias

For this section, the following definitions are provided.

The survey domain of estimation is the geographical area for which statistical inferences, such as the mean abundance of a species, are targeted. This is also commonly referred to as the survey area.

In statistics, a sampling unit is one of the basic units into which an aggregate (the sampling frame) is divided for the purpose of sampling. In marine resource surveys, the basic sampling unit is a haul and the sampling frame is the ensemble of all possible sampling units, i.e., all distinct locations where a haul can be made. Ideally, the area covered by the sampling frame will be identical to the survey domain, though often it will be smaller due to areas where a haul is not possible due to the habitat or bathymetry.

The target population is defined here as the ensemble of sampling units where a biological stock for which inference (e.g., mean abundance) is desired occurs. Note that the term stock is used in this section to mean a biological population and is used to differentiate from the population of sampling units.

In survey methodology, coverage refers to the relationship between the sampling frame and the target population (Lohr 2010). When the target and sampled population of hauls is the same, coverage is said to be complete. Given correct sampling methods (e.g., random selection of haul locations), estimates of mean density for the stock(s) will be unbiased, i.e., on average they will be equal to the ‘true’ values.

In contrast, undercoverage occurs when a subset of the target population is not included in the sampling frame (Lohr 2010), i.e., the excluded units cannot be part of the sample of chosen survey location. Undercoverage with respect to a stock or species occurs when the survey area does not include places where the stock or species occurs. The exclusion of surveys from areas that were previously included in their sampling domain will generate new, or enhance existing, undercoverage for stocks and species that reside in the newly protected area. The degree of coverage will remain unchanged for stocks and species that reside exclusively outside the protected area.

In the absence of suitable corrections (see 5.2.1.3 below), undercoverage can cause bias in the estimated properties (e.g., mean abundance density), that is, on average these estimates will differ systematically from the true values. The sign (positive or negative) of the bias will depend on the nature of the undercoverage. If density in the excluded subset is like the target population, the undercoverage will have little or no impact on the bias. If densities in the excluded subset are different from the target population, the impact will depend on the difference:

- if the excluded subset is associated with large density values, then the under-coverage will result in a negative bias;
- if the excluded subset is associated with small density values, then the under-coverage will result in a positive bias;
- if the excluded subset is associated equally with small and large values, then there is little or no impact on bias;

- if the excluded subset is associated with middle values, then there is little or no impact on bias.

In the first two cases, the magnitude of the bias will depend on the degree to which abundance densities at sampling units differ between the excluded subset and the new sampling frame, as well as the degree of undercoverage.

Strictly speaking, bias does not cause problems for resource monitoring if its magnitude and sign do not vary in time, unless abundance values from surveys are treated as absolute rather than relative. In fact, catchability to a survey is a bias as regards total abundance. This is why the estimates from the majority of existing surveys are treated as indices of relative abundance. Bias resulting from new or additional undercoverage also does not cause a problem for abundance indices provided that bias does not vary systematically over time. Existing abundance indices can be recalculated, excluding samples that were obtained from the area that is now protected. Abundance trends will remain proportional to true abundance. Of course, if there are certain life-stages of a population that reside exclusively in the closed area, their abundance will not be reflected in the new indices and the tracking of cohorts as a function of stage, size or age, as required by some population assessments, would be affected.

Bias caused by undercoverage becomes problematic when the magnitude and possibly the sign of that bias vary systematically over time, i.e., if it is non-stationary in which case abundance indices will no longer be strictly proportional to true abundance. This is true for existing undercoverage resulting from survey domains that do not circumscribe the distributional area of a stock or species (e.g., Davies and Jonsen 2011), and for new undercoverage that results from the exclusion of surveys from protected areas where they occur. It is worth noting that in some instances it might be possible to relocate sampling stations from a newly protected area to a neighboring formerly unsampled area of similar habitat, effectively compensating for the new undercoverage. Such a strategy has been employed in the International Pacific Halibut Commission survey with respect to sites that fell in the Hecate Strait MPA (pers. comm. J. Amyot, DFO Oceans Program, Pacific Region). The success of such a strategy requires that densities in the two areas be similar in magnitude and vary similarly over time.

There are two principal mechanisms that can generate time-varying bias in estimated relative abundance resulting from undercoverage: shifts in stock spatial distribution and divergent (sub)stock demographic dynamics in the closed area relative to the broader area. These effects can occur throughout a stock's distribution and therefore for many stocks the effects on bias may include contributions from multiple protected areas.

5.2.1.1 Survey bias – shifts in stock spatial distribution

Shifts in the spatial distribution of a stock that result in density changes that are not proportional inside and outside the sampling domain will result in a time-varying bias. There are numerous factors that can elicit a change in distribution, and such changes are very common in mobile marine fauna (MacCall 1990; Pinski et al. 2013; Shackell et al. 2014).

Density dependent shifts are among the most pervasive (reviewed in MacCall 1990; Swain and Sinclair 1994). Optimal foraging theory predicts an expansion of population range into marginal habitats as abundance increases, consequently percent changes in local density are expected to be greater in marginal than in optimal habitat (Fretwell and Lucas1970; reviewed by Rosenzweig 1991). Many stocks display dynamics consistent with the predictions from optimal foraging theory (MacCall 1990; but see Swain and Morin 1996 for a contrasting pattern). For stocks for which an unmonitored area constitutes part or all of their preferred habitat, abundance changes will be greatest in the monitored area and densities estimated by the survey will generally be monotonically but not proportionally related to abundance. As abundance declines from a high level, densities estimated by the survey will decline more rapidly, resulting in what is termed hyper-depletion (Hilborn and Walters 1992). The trends and status for the stock will appear more pessimistic than they truly are (e.g., Davies and Jonsen, 2011). Conversely, for stocks for which the unmonitored area constitutes potential, but less favoured habitat, densities in the survey will remain disproportionately elevated as the stock is declining. This phenomenon, termed hyper-stability, creates an elevated risk that management actions aimed at halting stock decline and promoting rebuilding will be delayed (Hilborn and Walters 1992). This in turn causes added harm to the stock, potentially causing it to reach levels from which rebuilding is difficult (Hutchings 2014; Kuparinen et al. 2014).

The siting of protected areas often involves the selection of sites with properties that distinguish them from alternative areas, such as high densities of taxa of conservation concern, high biodiversity or EBSAs. By the nature of containing unrepresentative habitats, protected areas are likely to include habitats that are favoured by some taxa, and marginal or neutral for others. There is therefore a high potential for survey exclusions from many protected areas to result in hyper-stable survey indices for some stocks and hyper-depletion for others.

Environmental changes can affect the location of preferred habitats and therefore stock distributions. Most notable among environmental changes are those associated with climate change, and its related effects such as ocean acidification and hypoxia. Large scale changes consistent with climate change effects have already been observed in some areas (Perry et al. 2005; Pinski et al. 2013), and more dramatic changes are predicted (Cheung et al. 2009, 2011). Other environmental changes can result in fitness trade-offs that affect the choice of preferred habitat. A notable example is predation risk (Heithaus and Dill 2006; Swain et al. 2015). Overall, environmentally-induced changes are likely to result in disproportionate changes in abundance in monitored and unmonitored areas, resulting in time-varying patterns of hyper-depletion or hyper-stability.

5.2.1.2 Survey bias – divergent demographic dynamics

Even in the absence of distribution shifts, different levels of stock productivity in monitored and unmonitored sites can result in different patterns in abundance and different stock dynamics following perturbation or as a result of environmental variability. Sessile species, those with limited mobility and those with high site fidelity, are most susceptible to experience location-dependent productivity, while highly mobile species are least. This phenomenon may in fact underlie the spatial differences in density that motivates the choice of location of protected areas in many instances. To the extent that this is the case, survey undercoverage due to exclusion from protected areas is prone to result in bias. With the exception of large scale spillover effects, positive effects of protection are further expected to result in different productivity and dynamics inside and near the protected area, compared to further afield, causing problems for population assessments (Field et al. 2006). The magnitude of the resulting time-varying bias is likely to increase with the degree of site specificity, restrictions on mobility of all life stages, the degree of benefit accrued from protection and the degree of undercoverage resulting from survey exclusion in the protected area. Protected areas that effectively protect a portion of a population can further result in life-history characteristics that diverge over time within and outside the area. This can result in a bias in the estimation of life-history characteristics which can be important considerations in understanding population productivity on one hand and indicators of overall fishing pressure on the other (Field et al. 2006; Rochet et al. 2000).

5.2.1.3 Accounting for undercoverage to reduce bias

There are two general approaches for data analysis and statistical inference to dealing with the exclusion of surveys from a newly protected area. The first is to remove the now unsampled area(s) from the survey domain and to base broader scale inferences on a geographic area that excludes the protected area(s). This amounts to taking no action to reduce possible biases. Such an approach is appropriate if mean densities within the protected area and within the sampling domain are always the same. It could also be appropriate for sessile taxa for which the protected area is a sink, where individuals residing within it do not contribute to the dynamics of the overall population nor do they contribute to densities outside the protected area.

The second approach is to maintain the existing survey domain and to attempt to impute (predict) values for sampling units falling inside the protected area. An effective method that imputes values that result in averages that are similar to those that would be obtained by sampling in the area will in effect address any potential biases (Ono et al. 2015), whereas an ineffective method will not, on average. It is also possible that an imputation method will be inappropriate, in that it consistently predicts incorrect values, potentially leading to exacerbated biases. In some instances, it may be possible to validate the accuracy and effectiveness of different imputation methods via a retrospective analysis of existing survey data (for an application, see section 6). The results of this validation will be correct provided that the relationship between values inside and outside the protected area remains stationary. However, should that relationship change in the future, for example as a result of density dependence or changes in movement patterns related to ecological or environmental change, the imputation will cease to be effective and appropriate, likely leading to time-varying bias. Detecting and particularly responding to such a situation will be very difficult and potentially impossible in the absence of renewed sampling inside and outside the area of survey exclusion.

There are numerous approaches for imputation. The most basic is to assume that mean densities are the same within and outside the protected area, producing a result that is equivalent to that obtained by the first approach described above. This assumption is implicit in calculations of mean density over all samples from the sampling frame, without explicitly accounting for portions of the survey domain that are not covered by the sampling frame. Other approaches for imputation use estimated densities in areas neighboring the unsampled area or in locations comprising similar habitats, to provide more refined predictions for the unsampled area. These approaches can range from simple assumptions concerning similarities in density, to the use of more sophisticated geostatistical (spatially and perhaps temporally autocorrelated) models (Shelton et al. 2014; Thorson et al. 2015). However, regardless of the method, if the area for interpolation is relatively large and there are important non-stationary changes in distribution, imputations based on past information may not be reliable.

5.2.1.4 Evaluating the potential bias caused by the exclusion of surveys from more than one protected area

Retrospective analyses as described above can serve to identify potential residual biases that may remain despite attempts at imputation, although it will often not be possible to predict the future magnitude of these biases. Individual surveys may overlap with numerous closed areas, thus it will be necessary to best evaluate the potential overall bias resulting from the exclusion of surveys from many or all of these areas. While the exclusion of a survey from one small area may contribute little to overall bias, the cumulative impact of exclusion from multiple areas could be significant. Decisions to exclude surveys from protected areas should therefore involve a strategic assessment across all existing and planned protected areas.

5.2.1.5 Determining whether impacts on bias and precision matter

The foregoing addresses the potential for bias and how to quantify it. Effects on the precision of estimates may also occur and can likewise be simulated. However, the key consideration is whether these impacts are of sufficient magnitude to matter. Minimally, at which point do they affect scientific advice? Ultimately at which point do they affect the decisions that follow from that advice, and how does that affect the likelihood of achieving long-term fisheries and conservation objectives?

Impacts on advice can be considered by evaluating the statistical power available to reliably conclude that there has been a change in stock status when such a change has occurred (Ellis 2010; see example in Sinclair et al. 2003) and to correctly determine its magnitude and direction or sign (Gelman and Carlin 2014). Simulations of various scenarios of population trends and of time-varying effects of undercoverage on bias and precision can be undertaken to evaluate the power associated with making a correct determination. To further evaluate the downstream impact of increasing bias, one must undertake closed-loop simulations of the science-management-fishery process, via what is termed management strategy evaluation (MSE; Smith et al. 1999; Rademeyer et al. 2007; Punt et al. 2014). An MSE evaluation goes further than demonstrating an effect on biomass indices in terms of bias, precision, or time-varying bias (a necessary step), by assessing risk to fisheries or (non-benthic) conservation objectives of current or alternative assessment and management procedures. While highly useful, MSE processes can be lengthy and resource intensive, and therefore not broadly applicable. They are likely to be most important for key resources where acceptability of surveying in closed areas may be very low and the risk of impacts on the reliability of stock indicators are high.

5.2.2 Providing Scientific Advice When Survey Indices Are Biased

With at least 10% of Canada's oceans protected using spatial management measures by 2020, there will be substantial overlap with many existing surveys. As noted previously, this overlap is likely to be large, given that data from the existing surveys were/are often used in protected area planning. Even in cases where the overall amount of overlap between surveys and protected areas is small, there are likely to be some stocks monitored by multispecies surveys for which the overlap between their distributional range and the protected areas is high.

When protected areas are relatively small relative to stock range and include habitats that are not used or considered very marginal, even at high density, the resulting biases in survey estimates are likely small enough to disregard (Rideout and Ollerhead 2017), provided that those habitats do not become more favored with broad scale environmental change or as a result of the spatial protection. Conversely, for stocks for which the protected areas constitute a non-negligible portion of their range, or for which the areas are disproportionately favored, biases that cannot be eliminated via imputation may be too large to disregard. Unfortunately, unless retrospective analyses can be trusted to provide appropriate estimates of residual bias, it is often difficult, if not impossible, to estimate bias caused by undercoverage in a survey (Lohr 2010). Consequently the potential for bias should be evaluated based on the preceding considerations, on a stock-specific basis and for the suite of stocks of interest in an ecosystem.

In the absence of ongoing additional monitoring that can inform on changes in the magnitude in bias over time, the data from scientific surveys will either need to be disregarded entirely in the provision of scientific advice if the biases are believed to be large, or used with caution, given the risk of generating misleading advice. This can compromise the information or advice for species for which the surveys are the main or only source of information on relative abundance.

While it is difficult to imagine cases where a survey would be excluded from an area, but a fishery using the same type of gear would not, there may be some protected areas in which certain forms of commercial fishing are allowed, but surveys using more impactful gear are not. Catch rates (CPUE) from commercial fisheries are used as abundance indices in many stock assessments worldwide, principally for stocks for which there is no monitoring from fishery-independent surveys (e.g., Atlantic Bluefin tuna). It is possible to use CPUE series instead of survey relative abundance indices in cases where the latter are considered unreliable because of undercoverage. Alternatively the CPUE series could be used as part of the imputation methodology, or the two series could be analyzed jointly to estimate or model the lack of proportionality in the survey series using the CPUE information (for an example for cusk, *Brosme brosme*, see Davies and Jonsen 2011). In all cases, one assumes that the CPUE series is proportional to abundance. This assumption is known to be incorrect in many cases, leading to hyper-stability or hyper-depletion in the CPUE indices (see 5.2.1.1), as a result of factors such as technological and technical changes in the fishery, market demands, gear saturation, and competition among harvesters (Hilborn and Walters 1992). It is for this reason that fishery-independent surveys are considered the 'gold standard' for monitoring abundance trends in species that are well sampled by them.

Fishery dependent data are, however, of little value for tracking the trends in abundance of bycatch species, some of which will be considered species at risk or of conservation concern. Changes in catch rate for bycatch may reflect changes in fishing practices or gear modifications to target another species, or to reduce bycatch of certain species. Without an independent means of monitoring abundance, it is impossible to determine if a reduction in bycatch reflects a decline in the abundance of a species or if it reflects the success of measures meant to mitigate bycatch.

Some options for providing catch advice in the absence of reliable indices of abundance do exist, based on what are collectively called data-limited methods (Carruthers et al. 2014, 2015). These methods use life-history information, catch data and/or catch characteristics (e.g., mean length of fish in the catch) to set catch limits. While the simulated long-term performance of some of these measures appears good in some contexts (particularly when stock productivity does not vary systematically over time), they are outperformed by methods employing an index of abundance (Geromont and Butterworth 2014). In particular, stock assessments supported by fishery-independent surveys (or CPUE series considered proportional to abundance) can be more reliable because stock productivity, and changes therein, can be estimated and dynamics can be modelled (ICES 2012b).

Ultimately, preoccupations with using biased or possibly overly simple methods to inform management decisions concern the likelihood of failing to meet conservation and other management objectives. As indicated above, these preoccupations can be assessed using MSE. In particular the closed-loop simulations can be used to identify management procedures that can meet fisheries management or conservation objectives and that can be expected to perform acceptably under a range of credible scenarios for the future dynamics of the fish stock and imperfect knowledge of the stock. In the present context, under such an approach, the goal will be to identify management measures that are expected to perform well despite degradation in the quality of stock indicators resulting from new undercoverage.

6. CASE STUDY - WHITE HAKE, ATLANTIC COD AND THORNY SKATE IN SEAPEN AGGREGATION AND CONSERVATION AREAS IN THE SOUTHERN GULF OF ST. LAWRENCE

The considerations presented above and the proposed framework were applied to a case study. It was chosen because it illustrates potentially conflicting conservation and management objectives. The case study is not meant to guide decision making for similar management questions or to draw general conclusions, but rather to illustrate how considerations presented above can be practically applied. Significant aggregations of a group of sensitive benthic species, sea pens, have been identified on the southern slope of the Laurentian Channel in the southern Gulf of St. Lawrence (Kenchington et al. 2016). Closures to bottom-contacting commercial fishing gear and possibly bottom-contacting survey gear are in place to protect these areas. The southern slope of the Laurentian Channel is now one of the last remaining areas occupied by the southern Gulf White Hake (*Urophycis tenuis*) population, a species at heightened risk of local extinction (Swain et al. 2015, 2016). The southern slope has also become an important area for Atlantic Cod (*Gadus morhua*) and Thorny Skate (*Amblyraja radiata*), two additional species of heightened conservation concern in the southern Gulf. Exclusion of multispecies trawl surveys from sea pen closure areas could affect the monitoring of these species, thereby potentially compromising recovery efforts.

6.1 CONTEXT

6.1.1 Background

In Canada, sea pen fields are considered Ecological or Biologically Significant Areas (Kenchington et al. 2014), and large dense fields are considered Significant Benthic Areas (SBA; DFO 2017a). Sea pen fields are present in patches along the Laurentian Channel of the Gulf of St. Lawrence at depths below 200 m. These fields contain the highest densities of sea pens in Atlantic Canada and the Eastern Arctic based on trawl survey catches (Kenchington et al. 2016). Three sea pen SBAs along the southern slope of the Laurentian Channel have been identified (Fig. 3; Kenchington et al. 2016). There are four species of sea pens that occur in these patches: *Anthoptilum grandiflorum*, *Halipteris finmarchica, Pennatula aculeata,* and *Pennatula grandis.* Based on catches in the multispecies research vessel (RV) bottom-trawl survey of the southern Gulf of St. Lawrence (sGSL), *H. finmarchica* is the least frequently encountered of the four species and catches are largest for *A. grandiflorum* and *P. grandis* (Table 8).

In 2017, a number of areas in and around the Laurentian channel were closed to commercial fishing to protect sea pen SBAs (Fig. 3). These closure areas, here termed sea pen closure areas (SCA), offer protection to portions of the SBAs from destructive bottom-contacting commercial fishing. There is now a requirement to evaluate whether research surveys employing bottom-contacting gear should be authorized in the SCAs. There are five major surveys that overlap with the SBAs and SCAs in the Laurentian Channel (Table 6): the RV bottom-trawl surveys in the sGSL and northern Gulf, Sentinel bottom-trawl surveys in each area, and a snow crab trawl survey in the sGSL. The sampling frames of these five surveys overlap spatially along the southern slope of the Laurentian Channel (Figs. 3 and 4). There are three SBAs that overlap with this area, as well as three SCAs: Eastern Honguedo Strait Coral and Sponge Conservation Area (Honguedo-East), North of Bennett Bank Coral Conservation Area (Bennett Bank), and Slope of Magdalen Shallows Coral Conservation Area (Magdalen Shallows Slope) (Fig. 4). This case study is focused on these sea pen areas. The impact and potential for harm of the five surveys is evaluated; however, when considering the potential impacts on surveys stemming from exclusion from protected area, only the RV survey in the sGSL is examined for illustrative purposes.

The sGSL RV survey was chosen to illustrate some of the trade-offs involved in the authorization decision making process. This survey has taken place annually in September since 1971 (for details see Hurlbut and Clay 1990 and Chadwick et al. 2007). It follows a stratified-random design, with strata based on depth and geographic region (Fig. 5). Survey sets are roughly attributed in proportion to stratum area, as can be seen in Figure 3. The survey tracks the relative abundance of over 50 species of fish and over 70 invertebrate taxa (Benoît and Swain 2008; Benoît et al. 2009). The data from the survey are the basis for the assessments of a number of groundfish species including Atlantic Cod (Swain et al. 2009), Witch Flounder (Ricard and Swain 2018), American Plaice (Ricard et al. 2016) and Yellowtail Flounder (Surette and Swain 2016), and also provide indices for the assessment of species such as Atlantic Herring (Surette 2016) and Snow Crab (Benoît and Cadigan 2016). Notably, the survey provided the main indices for the recovery potential or pre-COSEWIC assessments for a number of species of conservation concern including sGSL Atlantic Cod, White Hake and Thorny Skate (Swain et al. 2012c, 2012d, 2016). Atlantic Cod and White Hake are at heightened risk of extirpation this century and the RV survey is the principal means of monitoring the status of these populations. Importantly for the present context, adult Atlantic Cod, White Hake and Thorny Skate have experienced important shifts in their distribution, notably to deeper waters where the SBA and SCA occur (Swain et al. 2015; Fig. 6). Should these shifts in distribution continue, exclusion of RV surveys from the SCAs or SBAs could result in biased abundance indices (section 5.2.1), which is evaluated below.

6.1.2 Important Considerations

This case study is not meant to provide the definitive considerations for authorizing the sGSL RV survey or other bottom-contacting surveys. Decision making for any one survey will require joint evaluation of the consequences to the survey and subsequent scientific advice, resulting from exclusion from any, all, or a subset of closed areas that overlap with the survey. Here, only the SCA are considered, while for example the sGSL RV survey also overlaps with the Bank des Américains MPA and another area considered as an 'other effective area based conservation measure'. Permitting decisions may involve making choices on particular surveys to exclude, where multiple surveys overlap spatially and provide potentially very similar information, which is well beyond the scope of this document.

The factors discussed below include evaluations based on the overlap between surveys and the SCAs and the surveys and the SBAs. The latter evaluation illustrates what the considerations might look like when a greater proportion of Canadian marine waters are covered by spatial management measures, as Canada strives to reach its 2020 CBD target. It also illustrates the considerations under a scenario in which SCAs are expanded to provide additional coverage of the SBA. This could occur, for example, if DFO adopts the advice on the management of corals and sponges in the Newfoundland and Labrador region (DFO 2017b). That advice states that under the precautionary approach, 100% of SBAs should be protected and where socio-economic reasons justify less than complete protection, the maximum protection possible should be targeted, with an interim precautionary measure of 70% of the spatial extent of each SBA protected. The current level of protection is below this value.

6.2 OVERLAP BETWEEN THE SURVEY AND SEA PEN AGGREGATION AND CONSERVATION AREAS

The Honguedo-East SCA overlaps with the northern portion of sGSL RV survey stratum 415, the Bennett Bank SCA overlaps with part of stratum 425 and the Magdalen Shallows Slope overlaps with both strata 425 and 439 (Figs. 3 and 5). The percentage overlaps are presented in Table 9. The SCAs cover around 13.7% of stratum 415, 10.3% of stratum 425 and 4.4% of stratum 439. Conversely, the survey strata generally cover much larger proportions of each SCA: 14.2% of the Honguedo-East SCA, 19.9% of the Bennett Bank SCA, and 35.6% of the Magdalen Shallows Slope SCA.

SBA A overlaps with the northwest corner of stratum 415 (Fig. 4). SBA B overlaps with four survey strata: 415, 416, 425 and 426, though overlap with the latter is negligible and not considered further. Together, SBAs A and B cover 39.2% of stratum 415, while SBA B covers 5% of stratum 416 and 44.6% of stratum 425 (Table 9). SBA C covers both strata 425 and 439. Jointly, SBAs B and C cover 53.5% of stratum 425, while SCA C covers 52.1% of stratum 439. In contrast, the percentage of SBA area covered by RV survey strata is generally smaller. Stratum 415 covers 10.7% of SBA, while the survey covers 28.2% of SBA B and 47.0% of SBA C.

6.3 FACTORS TO CONSIDER IN THE DECISION TO AUTHORIZE SURVEYS IN THE CORAL CONSERVATION AREAS

6.3.1 Potential Impacts and Recovery for Sea Pens in the Closures

Sea pens are known to be sensitive to the impacts of bottom-contacting fisheries. Sea pens in commercial fishing areas in the Aleutian Islands were frequently found to be damaged (18% of observed individuals; Heifetz et al. 2009). Malecha and Stone (2009) simulated trawl disturbance and found that while some sea pens have the ability to re-erect themselves after a disturbance, mortality can increase through time due to predation.

Trawling experiments were conducted in sea pen fields just north of the Gaspe peninsula in the sGSL in August 2015, with follow-up monitoring in October 2015 and 2016 and in August 2016 (B. Sainte-Marie, DFO Quebec region, personal communication). The four species of sea pens of the Gulf occur in this area, though the area is most densely populated by *P. aculeata*. The experiments involved four passes of a shrimp trawl in three replicated corridors. Though many sea pens appeared to pass under the trawl footgear undamaged during the first pass, nearly all were removed or had burrowed in the sediment (Langton et al. 1990) after four passes. Following the disturbance, the site was repopulated rapidly by some *P. aculeata*, which may have reemerged from the sediment or have crawled along the bottom, an ability lacking in the other sea pen species. In 2016, the site was found to be at least partially recolonized by both small, presumably recruiting, and large, presumably migrating, *P. aculeata*. The recovery of the other species has yet to be established since analyses of the experiment are ongoing.

Sea pens possess some of the characteristics that make species sensitive to bottom-contacting gear impacts (section 3.3). They are erect and emergent, and are either sessile or have limited mobility (e.g., *P. aculeata*), making them vulnerable to contact by the gear. DFO (2017b) concluded that compared to other sensitive hard coral and sponge taxa, sea pens are often more resilient than other corals due to their shorter life spans and quicker growth rates; therefore they are believed to have the greatest potential for recovery with expected timescales of decades rather than centuries. Neves et al. (2015) estimated the age at maturation and maximum observed age of *H. finmarchica* in the NW Atlantic at 4 and 22 years respectively. Recently, Murillo et al. (2018), estimated the ages for *A. grandiflorum* and *P. aculeata* in the Gulf of St. Lawrence. Presumed ages ranged between 5 and 28 years for *A. grandiflorum.* Based on mean lengths in colonies in the sGSL survey, this would correspond to colonies 15-16 years old. Presumed ages ranged between 2 and 21 years for *P. aculeata*. Mean colony lengths observed in the SGSL survey corresponded to *P. aculeata* colonies younger than 9 years old. The authors modelled the growth patterns of the two species to estimate asymptotic size and therefore maximum age, with estimates that fell within previously published ranges for pennatulids of between 15 and 50 years. However, the authors cautioned that the age determination for the sea pens required additional validation. The results of the two studies are consistent with recovery times on the scale of decades.

RV survey strata 415, 425 and 439 cover a combined area of 5,998 km^2. The average annual number of survey sets in these strata for the most recent 10-year period, 2008 to 2017, is 13.4 per year (from Table 10). With an average swept area of 0.1402 km^2 (Table 6), this results in an annual mean swept area of 1.879 km^2, or 0.03% of the total areas per year. Since the survey sets are distributed randomly, on average 0.03% of the SCA areas that overlap with the sGSL RV survey are trawled annually by that survey. The recurrence time interval for sGSL RV survey sets at locations in the strata is 3,192 years.

The cumulative effect of the various bottom-trawl surveys is greater than for any single survey and is not spatially homogenous within the SCAs and SBAs. All five bottom-trawl surveys overlap in the southwestern portions of the SCAs and SBAs that overlap with the sGSL RV survey sampling frame area (Fig. 3). In contrast, the nGSL RV and Sentinel surveys are the only bottom-contacting ongoing surveys that occur in the northwestern portions of the SCA and SBA that fall outside the sGSL RV sampling frame area. These two portions are hereafter respectively termed the southern and northern portions of the SCAs/SBAs for simplicity. The proportion of areas impacted and survey recurrence time intervals for southern and northern portions of each SBA and SCA are provided in Table 11. The most impacted areas are the southern portions of SBA C and SCAs 2 and 3, with between 0.16% and 0.19% of the areas swept by surveys annually, leading to recurrence time intervals of 540 to 625 years. For all other sea pen areas and portions, recurrence time intervals were greater than 1,450 years. In the northern portion of SCA 3 recurrence time is estimated at close to 16 thousand years. In these calculations we have assumed that the snow crab survey samples independent locations each year, while in effect, the survey largely samples fixed locations. This assumption means that the proportion area affected in the southern portions is actually overestimated and recurrence time underestimated. At the sites where the snow crab survey occurs, impacts may cumulate over the years due to repeated trawling at the same sites, but this is limited to between 0.025 to 0.066 km^2 in the SCAs and to zero km^2 in SBA A, 0.42 km^2 in SBA B and 0.22 km^2 in SBA C.

Based on the available evidence, the surveys:

- involve a single annual pass, which will likely remove only a portion of sea pens in the trawl path based on results from the trawling experiment, and not completely remove sea pens as assumed by the approach proposed in section 4.4.3;
- impact only a very small proportion of the SCA, SBA and neighboring areas annually, resulting in little habitat fragmentation and therefore high potential for recolonization by migrating *P. aculeata* and recruits of that and other species, as a result of a presumably large available pool of local reproductive sea pens; and
- will recur on a timescale (100s to 1000s of years) that is at least an order of magnitude longer than the anticipated recovery time of disturbed sea pens (10s of years).

6.3.2 Mitigation Potential - Alternative Monitoring Methods, Gear/Survey Modifications and Exclusion of Redundant Surveys

The RV and Sentinel surveys are used to monitor the abundance and distribution of a large number of species in broad areas of the Gulf, including the Laurentian Channel where they monitor species of commercial interest including Witch Flounder, Greenland Halibut, redfish, and Atlantic Hagfish (Ricard and Swain 2018; Morin et al. 2017), and species of no commercial interest. Bottom-trawls are well suited for monitoring a large diversity of benthic and demersal taxa. Fixed-gear alternatives are too selective for species and size, while observational methods are not well suited for sampling fish that occur at low densities and do not provide the necessary biological samples required for assessments (section 4.5.1).

The snow crab survey also collects information on a broad suite of species, but is principally used to monitor the relative abundance of crab. The trawl survey was chosen over a fixed gear (trap) alternative because it allows for the monitoring of all sizes of crab, providing recruitment indices that are important for stock forecasts and management decisions (Hébert et al. 2016). Furthermore, trap surveys can be subject to spatially and temporally variable catchability which affects the quality of abundance indices (Cadigan et al. 2017). This results from factors such as gear saturation at higher densities and the effect of local currents which impact the area over which bait is attractive.

The RV and Sentinel surveys have target fishing procedures with respect to tow duration and tow speed. Tow durations that are not less than two-thirds the target are considered acceptable. Consequently the area swept annually by each survey (footprint) could be reduced by a third in the SCAs, without compromising survey quality. However, fishing experiments would be required to establish whether the catch scales linearly with swept area to ensure that biases are not introduced.

Portions of the SCAs and SBAs are overlapped by all five surveys and there is likely to be some redundancy in the information that is collected (e.g., see comparisons for the sGSL RV and Sentinel surveys in Swain et al. 2012b). Following careful examination of potential redundancies and the potential to share data among surveys (for an application to snow crab see Benoît 2012; Benoît and Cadigan 2016), it might be possible to exclude one or more of the surveys from the sea pen areas. This could lead to important reductions in footprints and increases in recurrence time intervals in the southern portions of the sea pen areas.

The survey trawls could be modified to lessen impacts on the seafloor by changing the footgear or reducing the trawl size and door weights. Changes in fishing efficiency resulting from such changes would need to be evaluated prior to the change and for all species monitored by the survey and constitutes a very large endeavor (section 4.6). The benefits gained by such modifications are likely to be small compared to the approaches above for reducing footprint size.

6.3.3 Trawl Survey to Monitor Changes in the Closure

The RV surveys provide the only large scale and long-term monitoring data of sea pen densities within the SCAs and neighboring areas prior to the implementation of the closures. Such data are required if the efficiency of the SCAs is later to be evaluated based on the BACI-type design (section 5.1). There have been other local surveys using video-based methods, but these would be insufficient to provide the necessary information at the correct scale and would not provide information about spatial-temporal variation. That said, trawl surveys are likely not optimal for monitoring sea pen given potentially variable catchability (Kenchington et al. 2011) and variability in data from the RV survey could result in low statistical power for detecting an effect of the closure on sea pens. However, data from the RV survey could be appropriate for monitoring fish and macroinvertebrate taxa that are associated with sea pen fields and which may respond to the closure. Ecological responses with respect to conservation objectives related to these species and communities could therefore be evaluated using the RV data.

6.3.4 Trawl Survey to Monitor the Status of Fish Species of Conservation Concern

The sGSL RV survey has been used to assess and monitor the status of Atlantic Cod, White Hake and Thorny Skate, among other species that occur in the sGSL or on the southern slope of the Laurentian Channel. To date, the information from the other surveys has played a much less important role in the assessments. Here we consider the potential impacts of excluding the sGSL RV survey from the SCAs or the SBAs on the estimated abundance indices for adults of these priority species. For the present purposes the adult classes were based on lengths measured in the surveys, consistent with what was done in the most recent scientific advice for these species: ≥39 cm for Atlantic Cod, ≥45 cm for White Hake and ≥51 cm for Thorny Skate (Swain et al. 2012c, 2012d, 2016).

6.3.4.1 Methods

Set specific catches for each species were first standardized for tow length, past changes in survey vessel or gear, and a change from daylight only survey activities prior to 1984 to 24 hour surveys thereafter (for details see Swain et al. 2012c, 2012d, 2016). The analysis was restricted to survey strata 415-439 (Fig. 5), which is the core group of strata sampled annually since 1971.

Three inshore strata (401, 402 and 403) were added in 1984, but are not included in the analysis in order to have a homogenous series since 1971. Data for 2003 were omitted because there was no sampling in strata 425 and 439 and an uncalibrated vessel was used that year.

The annual mean catch (numbers) per tow with associated standard error was estimated using the standard estimators for stratified-random sampling. Five sets of estimates were made for each species, based on 1) all available survey sets (standard analysis), 2) all sets excluding those in the SCAs, and 3) all sets excluding those in the SBAs. For the latter two, two sets of estimates were made. In the first, the portion of the stratum areas that intersected with the SCA or SBA areas were removed from the sampling domain, thereby reducing its surface area. This simulates undercoverage (as defined in section 5.2.1). In the second, densities in the intersecting areas were imputed and the current sampling domain area was unchanged. Densities were imputed by assuming that they were the same as the mean densities estimated in the remaining (sampled) portion of the stratum. This imputation method is simple and is consistent with the assumption of a stratified-random design that fish densities are homogenous within strata. Nonetheless, more sophisticated imputation methods that account for spatial autocorrelation might be possible.

A minimum of two sets per stratum and year is required to estimate a stratum mean density and variance. The elimination of sets that fell in the SBAs resulted in a number of instances in which this minimum was not achieved (Table 10). To get around this problem, strata were combined in certain years in the estimation. Most often strata 425 and 439 were combined, and this was done for 1971, 1972, 1975, 1983, 1984, 1986, 1988, 1992, 1995, 1998, 2000, 2004, 2007, 2009, 2010-2012, 2014, and 2017. Strata 415, 425 and 439 were combined to achieve sufficient sample sizes in 1973, 1977 to 1982 and 2013. For the analyses dealing with the SCAs, there was only one instance in which strata needed to be combined, strata 425 and 439 in 1979. It is important to note that the above issues with sample size are germane to the present simulations and would not reflect the situation if survey hauls were excluded from SCAs or SBAs in the future, since these hauls would almost certainly be redistributed elsewhere within the strata.

6.3.4.2 Results

The exclusion of survey sets from the SCA areas produced abundance indices that were very comparable to the standard index for the three species, when considered over the 1971 to 2017 period (Fig. 7, left column). However differences between the series were more pronounced during the last 25 years for Thorny Skate and White Hake (Fig. 7, right column). The differences are most evident when examining the log of the ratio of the index for excluded catches over the standard index (Fig. 8). Trends in log-ratio as a function of year were evaluated using general additive models assuming Gaussian errors (GAM; Wood 2006). Over time, the abundance indices for White Hake and Thorny Skate that excluded the areas of survey and SCA overlap were increasingly smaller compared to indices which used the standard survey, though the trend was only significant for Thorny Skate. This reflects a shift in the distribution of these two species into some of the areas that would now be excluded from the survey domain (Fig. 6). In contrast, the abundance indices for White Hake and Thorny Skate that involved imputation of densities for the overlapping area were increasingly larger over time relative to the standard survey, except in the most recent years for Thorny Skate (Fig. 8, open green symbols). Although the species have shifted into deeper waters over time, the shift has been less pronounced in the deepest parts of the survey, where the overlap with the SCAs occurs. As a result, densities in the sampled portions of the strata are greater than densities in the unsampled portion, while the imputation assumes they are the same. For Atlantic Cod, which are more broadly distributed throughout the sGSL and which do not occur in the deepest areas of the survey, excluding these sets had only a minor impact on the survey series (Fig. 8), with a slight positive bias in the series that excluded the SCAs from the survey domain. This was because cod densities in the excluded area are low or zero, which slightly brings down the mean in the standard estimation.

For indices that excluded survey sets in the areas where SBAs occur, the results are generally similar in pattern but more pronounced compared to the results above for SCAs (Figs. 9 and 10). This stands to reason because the SBAs are much larger in surface areas compared to the SCAs but occur at similar mean depths. For White hake, there was a significant negative trend in log ratios when the conservation areas were removed from the survey domain, but not when values were imputed. For Thorny Skate, there were significant negative trends in log ratios for both methods.

Overall, the results indicate that excluding the sGSL RV survey from either the SCAs or the SBAs did not adversely affect the estimates for Atlantic Cod, but could result in abundance series for Thorny Skate and White Hake that are increasingly biased or possibly more variable with time. Excluding the area of survey and SCA/SBA overlap from the survey domain results in indices that overestimate declines over time. Meanwhile imputing values for the area of overlap can result in underestimated declines in time (e.g., White Hake and Thorny Skate for 1990 to about 2014 in the SCA analysis). Such time-varying biases could compromise recovery monitoring and therefore recovery action efficacy for these species. Alternative imputation methods may not result in such a bias, though this would need to be more thoroughly evaluated.

7. PUBLISHED ECOLOGICAL FRAMEWORK ON PERMITTING DECISIONS (SAARMAN ET AL. 2018)

Saarman et al. (2018) published in June 2018 an ecological framework that was developed for informing permitting decisions on scientific activities in Californian marine protected areas. Like the Canadian framework agreed upon at the January 2018 peer review (DFO 2018), their framework uses indicators to inform decision making, and does not prescribe decisions. Both frameworks seek to evaluate the risks associated with planned scientific activities in the context of protected area objectives, but also in light of the potential scientific and management benefits of those activities. However, the Californian framework is more general in its intended application, covering scientific activities affecting all biological components in a protected area and physical habitat, not just benthic ecosystems. Proximate, ultimate and cumulative impacts on an area’s populations, ecological assemblages and physical habitats are evaluated using semi-quantitative measures, typically based on expert opinions.

Proximate ecological impacts on populations or assemblages are generated as the sum of additional annual mortalities (expressed as proportions) caused by the scientific activities, accounting for sampling inefficiencies that increase mortality, multiplied by the proportion of the population/assemblage affected. Mortalities are almost exclusively based on expert opinion due to the lack of published estimates. By summing mortalities over all scientific activities, the approach accounts for cumulative impacts of scientific sampling.

In the Californian framework, ultimate impacts are calculated as the product of proximate impacts and recovery time divided by two, further multiplied by an index of the ecological importance of the population/assemblage. The reason for dividing recovery time in half is not well explained in Saarman et al. (2018). Notably, recovery time is taken as the inverse of natural mortality (M) for individual populations or from the species with the lowest M (i.e., the least productive and therefore most sensitive) for an assemblage. Values for M are by and large taken from the review by Hoenig (1983) for different taxonomic groups, with values estimated from the following empirical relationship with maximum age (longevity):

- $\log(M)=1.44 - 0.982*\log(\text{max age})$

The use of this equation results in recovery times that are much shorter than those employed in the Canadian framework. For example, an organism with a lifespan of 1,000 years would be assumed to have a recovery time in the Californian framework of 209.2 years (which is further divided by two). Nonetheless, Saarman et al. (2018) claim, without supporting references, that their recovery time proxy would allow for “replacement of the abundance (density or percent cover) and size-structure of individuals removed, to reflect the lost density and size-dependent functional roles of impacted species”. The Canadian framework assumes that this process could take much longer, up to an order of magnitude longer than longevity.

The calculation of ultimate impacts in the Californian framework includes a qualifier for ecological importance of a species or assemblage (termed interaction index). Examples of species with ecologically important roles include habitat forming, trophically important and keystone species. Through a number of steps informed by expert opinion, the authors derive an ‘interaction index’ ranging from 1 to 3 that is included in the calculation of ultimate impacts as described above. The resulting calculated ultimate impacts are then assessed against a threshold value established a priori by managers. Saarman et al. (2018) propose an ultimate impact threshold of 0.1 as “a conservative starting point for setting impact threshold levels”.

With respect to assessing potentially acceptable ultimate impacts of research activities, the Californian and Canadian frameworks are almost mathematically equivalent. The Californian framework proposes that acceptable scientific activities should generally respect the following inequality:

$$(\textstyle\sum_{source} Z_i) \cdot Prop.Impact \cdot \frac{T_{recovery}}{2} \cdot interaction < 0.1 \tag{5}$$

Where $\sum_{source} Z_i$ is the sum of mortalities (proportions) caused by scientific activities, *Prop.Impact* is as described in section 4.4.3, $T_{recovery}$ is recovery time and *interaction* is the interaction index. In comparison, the Canadian framework proposes that acceptable scientific activities should generally respect the following inequality:

$$(\textstyle\sum_{source} Z_i) \cdot R > 10 \cdot lifespan \;\; \Leftrightarrow \;\; (\textstyle\sum_{source} Z_i) \cdot Prop.impact \cdot lifespan < 0.1 \tag{6}$$

Where R is the recurrence time interval (eqs. 3 & 4), $R^{-1} = Prop.Impact$ (eq. 3), and the multiplier 10 on the left-hand side represents the order of magnitude buffer. The Canadian framework further assumes, as a precaution, that direct mortality is complete (100%), even though there is evidence that true values may often be less (section 3). Therefore $\sum_{source} Z_i$=1. As with the Californian framework, the species with the longest lifespan (or recovery time) is used to assess impacts for the assemblage.

Comparing the two frameworks, and noting that in the Californian framework:

- $\sum_{source} Z_i$ will typically have a value <1;
- $\frac{T_{recovery}}{2} \ll lifespan$, as noted above; and
- the term *interaction* can only comprise values ∈ $(1,2,3)$,

It appears that the Canadian framework will likely lead to more precautionary decision making in most instances.

There are two key elements which distinguish the Californian and Canadian frameworks. First, recovery times of disturbed physical habitat are considered only in the former. The assessment is based on expert opinion, but values are capped at 20 years for undescribed ‘pragmatic considerations’ (Saarman et al. 2018). It is doubtful that incorporating such considerations in the Canadian framework in the future would provide beneficial protection to habitats in protected areas in addition to the implicit protection afforded by evaluating the acceptability of impacts on sensitive and structure forming taxa.

Second, the Californian framework does not explicitly account for the broader risks to marine conservation resulting from modifications to long-term scientific monitoring programs to reduce or eliminate harm caused by them in the protected areas. Instead, that framework only considers whether a proposed scientific activity is appropriate to be conducted in a protected area. Appropriate activities are those that are relevant to the area's protections, needed to maintain the integrity of long-term monitoring programs, not feasible to conduct elsewhere or important and of sufficiently low impact to not interfere with protected area goals (Saarman et al. 2018). These activities are then evaluated with respect to their impacts in the protected area only. The authors only further state that there is "a recognized need to continue established surveys and the collection of time series data to inform resource management in and outside of protected areas" as well as "studies and environmental monitoring required to meet mandates of governmental agencies". There is no discussion about how those needs might affect the acceptable risk for impacts of scientific activities in protected area. In contrast, the Canadian framework is explicit about the broader conservation risks associated with modifying long-term monitoring programs and identifies factors that should be considered, though it is not explicit about how managers should balance risks to conservation outside and inside protected areas.

8. DISCUSSION

The elements presented here and in the framework that ties them together (DFO 2018) do not jointly lead to a prescription for decision making, but rather are a series of important considerations in the process of permitting research surveys in protected areas. These elements lead to an evaluation of risks of harm to benthic species and features, and risks associated with the statistical quality of monitoring data with respect to conservation objectives and management within protected areas and in the broader ecosystem (sustainable fisheries, species at risk, adaptation planning, etc.).The value placed on these different risks is not a scientific question, but rather one of conservation values and priorities, and trade-offs will be inevitable. It is therefore important that these trade-offs be made explicit to decision makers. On one hand, blindly stopping trawl surveys in closed areas potentially represents a large-scale transfer of risk from the benthic habitats and taxa onto numerous species of finfish and mobile invertebrates subjected to fishing mortality, many of which are at very low abundance levels due to historically intense fishing. On the other hand, failing to adequately protect highly vulnerable benthic taxa could severely hinder the achievement of conservation objectives. There will be opportunities to reduce the impact of individual surveys by reducing their footprint within protected area and by reducing the collective footprint of largely redundant surveys, while preserving the integrity of the data that are collected. In other instances the potential harm within protected areas may be sufficiently high, and the scientific value of sampling sufficiently low that decisions should be relatively straight forward. However given the extent of spatial protection targeted by the government, there are likely to be many cases in which the trade-offs in conservation risks are such that difficult decisions will be required.

Key to informed decision making will be ensuring that the evaluation of risks is as complete as possible. Within a given protected area, the harm caused jointly by multiple independent sampling programs must be evaluated with respect to recovery capacity. The estimation of recurrence times as described earlier is amenable to the estimation of such cumulative effects. Similarly, many surveys will overlap with more than one protected area, which may differ in their respective conservation objectives. It will therefore be important that the effect of cumulative proposed modifications to survey designs and sampling frames be evaluated based on simulations or data re-sampling, as in the case study, and on the basic principles known to induce bias and affect precision as described above. Initial assessments of survey harm and acceptability are therefore likely to be somewhat long and tedious. However investments early on in the process will minimize the chances that erroneous decisions are made that may be difficult to reverse or for which the adverse consequences only become evident once harm to features, populations and ecological communities inside and outside protected areas is advanced.

Ideally, permitting decisions would be based on case specific information on the harm caused by sampling gear and on recovery rates. This will not be possible for the majority of surveys in Canadian waters. The proxies developed here are intended as general indicators of potential long term harm for use in these instances. While these proxies are very similar to those developed independently by Saarman et al. (2018), a higher degree of precaution is proposed here than in the Californian approach. The cautious approach is perhaps warranted, given important uncertainties in gear impacts and factors affecting recovery rates of benthic taxa and features following disturbance. Furthermore, an important application of the Canadian framework will be for vulnerable taxa that may require a greater degree of precaution. In contrast, the Californian framework is intended for broader application including to a range of taxa differing in their sensitivity to disturbance by scientific activities.

Documenting the basis for decisions in each protected area will be critical given the divergent conservation values and priorities that are likely to exist among sectors within DFO, and certainly among the various external stakeholders. Furthermore, following best practices for risk-based decision making, it will be imperative that the basis for decisions be periodically reviewed as scientific evidence changes. Such changes include:

- new and more precise information on the degree of harm caused by bottom-contacting surveys and the recovery potential of affected taxa and species;
- evidence concerning the adequacy of monitoring within protected areas to ensure conservation objectives are met, which could lead to alterations in monitoring type, design and intensity;
- the development of sampling methods that can substantially reduce harm to benthic ecosystems while providing data that can appropriately and adequately replace the information presently collected by bottom-contacting surveys; and
- evidence suggesting that biases in broader ecosystem indices are larger or smaller than anticipated, motivating a need to revisit the decision to exclude or allow a survey in a protected area.

Of course periodic review of the basis for decision making will also be motivated by possible changes in conservation priorities within and outside protected areas.

The proposed framework should also be open to periodic review to ensure its efficacy and relevance. Experience gained in applying it may identify new considerations that should be contemplated. Experience may also identify elements or information that are recurrent across groups of protected areas and which may motivate a streamlined decision making process akin to class environmental assessments.

The new Canadian framework is founded on risk-based decision making, with many considerations that are often at best semi-quantitative. The reliability of decision making using the framework depends in part on the quality of the various conservation objectives that must be considered. Objectives that are specific, measurable, achievable, relevant, timely, evaluated and reviewed periodically (SMARTER objectives) provide a quantitative basis against which to more rigorously evaluate the various trade-offs involved in survey permitting in closed areas.

Notably, specificity and measurability allow for the determination of acceptable harm with respect to direct impacts of surveys on protected features, impacts of monitoring quality on the conservation of those features, and impacts on broader ecosystem components. Achievability is important in the context of protected area objectives as there will be no benefit to restricting science activities if the protection is unlikely to produce the anticipated conservation benefits. Structured evaluations and periodic review, as noted above, are required to ensure that scientific programs are meeting their objectives to support science-based decision making, while avoiding unnecessary or unacceptable harm.

This report has dealt principally with existing sampling activities. While many of the elements described here are also relevant to sampling in new areas, these warrant particular considerations. New areas include frontiers and pristine offshore locations, as well as locations within protected areas that are presently not part of survey sampling frames. Initial sampling in new areas does not involve considerations about possible biases to long term monitoring activities and therefore the acceptability of scientific activities depends only on a trade-off between possible harm and the benefits gained by the activity. In many instances this should motivate the use of sampling tools that restrict harm. Such pilot studies should reveal whether there are sensitive taxa that warrant enhanced precaution and whether there may be advantages to using more harmful sampling methods for further monitoring and surveying.

Key to correctly evaluating all conservation risks inherent in scientific permitting decisions is the evaluation of biases in survey indices resulting from the exclusion or modification of survey activities in protected areas. These can only be quantified retrospectively or from simulation. There will be uncertainty associated with assumed or simulated bias over time in cases in which surveys have been excluded from protected areas. Uncertainty can become high in the face of environmental change given likely impacts on species distributions. Assessments of the consequences of exclusion to the integrity of survey indices should recognize this uncertainty. A review of methods for imputation and their robustness to species distributional change would improve our understanding of potential time-varying bias in survey indices.

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11. TABLES

Table 1. Examples of fisheries closures with benthic conservation objectives.

Management measure	Description
Rockfish Conservation Areas	Bottom fisheries restricted to support replenishment of depleted Pacific stocks (Lancaster et al. 2015)
Strait of Georgia and Howe Sound Glass Sponge Reefs	Bottom-contact fishing prohibited on the reefs and within a 150m buffer zone (DFO 2017a)
Disko Fan Conservation Area (Davis Straits)	Closed to protect concentrations of large gorgonian corals, including high-densities of bamboo corals (Hiltz et al. 2018)
Hawke Channel Box (Newfoundland Shelf)	Gillnet, longline and trawl closure to sustain the crab fishery and conserve cod aggregations (Kincaid and Rose 2017; Mullowney et al. 2012)
Division 3O Coral Closure	Transboundary VME closure established by NAFO on the southern Grand Banks (Rideout and Ollerhead 2017)
Scallop Buffer Zones (SFA 21, 22 and 24)	Dragging prohibited along Gulf of St. Lawrence coastlines to protect juvenile lobsters and nursery habitats
Gulf of St. Lawrence Coral & Sponge Conservation Areas	Placed off-limits to bottom fisheries to protect sea pen and sponge dominated communities
Northeast Channel Coral Conservation Area	Dense octocoral colonies east of Georges Bank protected from all fishing impacts (Bennecke and Metaxas 2017)
Lophelia Coral Conservation Area (Laurentian Channel)	Bottom fisheries excluded to protect eastern Canada's live *Lophelia pertusa* coral reef (Buhl-Mortensen et al. 2017)
Emerald Basin and Sambro Bank Vazella Closures	Protection for unique concentrations of the barrel-shaped sponge *Vazella pourtalesi* off the coast of Nova Scotia

Table 2. MPAs with descriptions of benthic conservation objectives.

Area	Description
Endeavour Hydrothermal Vents	Conserves vent processes and unique structural habitats including chimney-like smokers that support endemics plus a diversity and abundance of microbes and invertebrates
Hecate Strait / Queen Charlotte Sound Glass Sponge Reefs	Bottom fisheries removed from core zones to protect the largest living examples of these fragile and extremely vulnerable reef systems that also provide structural habitat
SGaan Kinghlas-Bowie Seamount	Closure to all bottom-contact fisheries (January 2018) provides ecosystem protection for 3 submarine volcanoes and the rich communities of fish, plants and benthos that thrive in this highly productive setting
Tarium Niryutait	Covers portions of the Mackenzie estuary and shallow parts of the Beaufort Sea where the conservation of belugas, their habitats and the supporting ecosystem are prime objectives
Anguniaqvia niqiqyuam	Extending from Darnley Bay into the Amundsen Gulf, the site has broad ecosystem objectives intended to support key species, with benthic protection an implicit component
Gilbert Bay	Protects a coastal embayment in southeast Labrador, home to a resident population of genetically distinct cod
Eastport	Protection for lobsters and lobster habitat in two areas of central Bonavista Bay where the community proposed and supported a complete removal of commercial fisheries
Basin Head	Fish extraction and plant harvest are prohibited from a coastal lagoon system in eastern PEI to protect a unique strain of Irish moss found only in the MPA
Bank des Américains	High relief feature east of the Gaspe Peninsula with complex seabed habitats and diverse invertebrate communities; regulations would limit mobile gear throughout and exclude commercial and recreational fishers in a core zone
Laurentian Channel	Proposed regulations foreclose on fisheries access to a variety of channel seafloor habitats and lend protection for offshore conservation priorities, including what may be Canada’s most extensive sea pen fields
St. Ann’s Bank	Bottom-contact fisheries are prohibited in 75% of the MPA to conserve and protect benthic habitat, distinctive physical features, biogenic structural habitat, and high fish diversity off eastern Cape Breton
The Gully	Massive submarine canyon east of Sable Island National Park Reserve that contains the country’s highest known diversity of corals (Mortensen and Buhl-Mortensen 2005) and where all fisheries are excluded from depths >600m
Musquash Estuary	Protects benthic biodiversity and the varied habitats of a unique coastal embayment on the Bay of Fundy; exceptional for its natural condition in southwest New Brunswick

Table 3. Summary of biological and ecological traits that contribute to the acute or immediate sensitivity of taxa and communities to disturbance by mobile bottom-contacting gear. Entries represent a composite of traits identified in four studies: Hewitt et al. 2011; Bolam et al. 2014, 2017; Clark et al. 2016.

Trait	Trait modality	Response to disturbance
Living habit	Erect or emergent	Liable to breakage or to being dislodged
Living habit	Other	Vulnerability affected by living position
Living position	Sediment surface	Vulnerable to all bottom-contacting gear
Living position	Shallow in sediment (0-5 cm)	Vulnerability depends on the depth of the disturbance
Living position	Deep in sediment (>5 cm)	Vulnerability depends on the depth of the disturbance
Mobility	Sedentary	Highly vulnerable to the gear
Mobility	Limited	Vulnerable to the gear
Mobility	High	Able to escape the gear horizontally or by digging in sediment
Fragility (morphology)	Exoskeleton or rigid structure	Very fragile, will be damage or killed by contact with the gear
Fragility (morphology)	Other forms	Fragility dependent on factors such as softness
Feeding mode	Scavengers and predators	Known to benefit from trawl disturbance
Feeding mode	Other feeding modes	Sensitivity dependent on the nature of the disturbance
Body size	na	Can affect vulnerability to the gear and retention by the gear

Table 4. Summary of the prognosis for recovery of structure-forming coldwater sponge species according to various disturbance types associated with fishing activities. Recovery assessment is individual-based as opposed to community-based; based on Table 3 in Boutillier et al. (2010).

Disturbance Type	Comments	Prognosis for Recovery
Mechanical Damage: Minor tearing of body wall	Sponges showing tissue repair have been collected; increased risk of infection; distal wounds appear to heal faster than wounds on lateral surfaces	Excellent
Mechanical Damage: Large wounds relative to body size	Incomplete regeneration; increased risk of infection; impaired reproduction and growth	Moderate
Mechanical Damage: Breakage at base	No signs of recovery after 1 year during experimental trawling in Alaska	Very Poor or No Recovery
Dislodgement: Minor change to orientation, position relative to currents not strongly affected	Sponges can lay new growth down to adapt to minor change in current direction	Unaffected
Dislodgement: Significant change to orientation, position relative to currents strongly affected	Sponges likely to die if food availability is restricted as a result of dislodgement	Poor
Dislodgement: Sponge dislodged on bottom, free-floating	NA	No Recovery
Dislodgement: Sponge brought up on deck and returned	When the aquiferous system is drained very few sponges can fill it up again; air in the chambers cause the sponges to float	No Recovery
Dislodgement: Crushing	Turning over of substrate commonly seen in trawl tracks	No Recovery
Sedimentation: Light accumulation of sediments in incurrent aquiferous system, no serious damage to aquiferous system	Ability to clear sediment; sediment accumulation can be viewed in cross sections with concentrations near ostiole	Very Good
Sedimentation: Repeated accumulation of sediments in incurrent aquiferous system	Sponge death or impairment	No Recovery

Table 5. Sensitivity categories for deep-sea benthic taxa to disturbance by mobile fishing gear (Clark et al. 2016).

Sensitivity	Expected response to disturbance	Traits
High	Mortality of individuals in the swept area	Fragile, sessile, erect and emergent forms
Intermediate	Mortality of some individuals in the swept area	Fragile forms with no or limited mobility. Surface dwellers
Low	Mortality of a few individuals in the swept area	More robust or small erect forms, dwellers in the top layer of the sediment with limited mobility
Tolerant	No response	Robust or mobile surface dwellers or subsurface dwellers with high burrowing capacity
Favoured	Individual may move into the area	Mobile scavengers and predators

Table 6. Inventory of ongoing bottom-contacting surveys undertaken in marine coastal, shelf and slope waters off Canada. Surveys are identified by DFO region (C&A-Central and Arctic, Gulf, MAR-Maritimes, NL-Newfoundland and Labrador, PAC –Pacific, and QC- Quebec), survey name, targeted species, location and gear employed (OTB – otter trawl or bottom trawl, LLS – bottom set longline, GNS – bottom set gillnet, DRB – bottom dredge, DRH- hydraulic dredge , pots and traps). The sampling design employed in the survey (F- fixed station, R – random, SR- stratified random, T-transect), the survey frequency (Freq: A-annual, B-biennial, R-rotational, and O-occasional), the mean number of hauls per complete survey in recent years (Hauls), the length of gillnet and longlines (Length, in meters), configuration of pots and traps (Config), estimated swept area per average haul (Haul swept area; in km2, see text for methods) and based on door-spread for trawls (Haul swept doors), the area of the survey domain (in km2), annual survey swept area (in km2)and the recurrence interval (Recur. Interval; in years, for random and random-stratified surveys only, see text for methods) are indicated where the information was available to the report authors. In the table, "-" means information not available to the authors, "na" means not applicable and "nd" means not derived.

DFO Region	Survey	Spec.	Location(s)	Gear	Design	Freq	Hauls	Length	Config.	Haul swept area[9]	Haul swept doors[10]	Survey domain	Survey swept area	Recur. interval
C&A	Arctic char fishery-indep. sampling	Arctic char & bycatch	Cambridge Bay, NU	GNS	F	A	-	46	na	-	na	0.20	nd	nd
C&A	Greenland halibut longline survey	Grld halibut	Cumberland Sound	LLS	SR	A	30	800	na	0.0800	na	7,002	2.4	2,918
C&A	Northern shrimp trawl survey	N. shrimp	Arctic, SFA 1	OTB	SR	A	360	na	na	0.0250	0.0667	185,541	24.0	7,730
C&A	Northern shrimp trawl survey	N. shrimp	Hatton Basin	OTB	SR	A	-	na	na	-	-	-	nd	nd
C&A	Inshore exploratory surveys	Various	NAFO 0A, OB - inshore to shelf break	LLS	SR	A [1]	45	800	na	0.0800	na	8,136	3.6	2,260
C&A	Multispecies trawl survey	Various	NAFO 0A, OB	OTB	SR	A	150	na	na	0.0700	0.3500	49,129	52.5	936
C&A	Inshore exploratory surveys	Various	NAFO 0A, OB - inshore to shelf break	OTB	SR	A [2]	80	na	na	0.0231	0.0833	5,424	6.7	814
C&A	Canadian Beaufort Sea Marine Ecosys. Assess.	Various	Beaufort Sea	OTB	T	A [3]	60	na	na	-	-	nd	nd	nd
GULF	Herring index multimesh gillnets (fall)	Atl herring	southern Gulf St. Lawrence (coastal)	GNS	R	A	20	50	na	0.0050	na	865	0.1	8,650
GULF	Sentinel longline	Atl. cod	southern Gulf St. Lawrence	LLS	F	A	300	1250	na	0.1250	na	~ 5,000	37.5	nd

DFO Region	Survey	Spec.	Location(s)	Gear	Design	Freq	Hauls	Length	Config.	Haul swept area[9]	Haul swept doors[10]	Survey domain	Survey swept area	Recur. interval
GULF	Sea scallop dredge survey	Sea scallop	southern Gulf St. Lawrence	DRB	SR	R: 5 yr	500 [4]	na	na	0.0004	na	23,520	0.2	534,545
GULF	Snow crab trawl survey	Snow crab	southern Gulf St. Lawrence	OTB	F	A	395	na	na	0.0028	*0.0083*	57,840	3.3	nd
GULF	Multispecies trawl survey	Various	southern Gulf St. Lawrence	OTB	SR	A	180	na	na	0.0405	0.1402	73,182	25.2	2,900
GULF	Northumberland Strait multispecies survey	Various	southern Gulf St. Lawrence	OTB	SR	A	110	na	na	0.0104	0.0347	11,925	3.8	3,122
GULF	Sentinel mobile	Various	southern Gulf St. Lawrence	OTB	SR	A	170	na	na	0.0362	*0.1085*	73,182	18.4	3,967
MAR	4Vn sentinel survey (September)	Atl cod	NAFO 4Vn	LLS	SR	A	56	3900	na	0.3900	na	13,750	21.8	630
MAR	4W Inshore sentinel survey (September)	Atl cod	coastal 4W	LLS	SR	A	18	3000	na	0.3000	na	4,350	5.4	806
MAR	Atlantic halibut industry survey	Atl. halibut	Nfld & Scotian Shelf	LLS	SR-F	A	150-230 [5]	4500-5500	na	0.5000	na	420,000	95.0	4,421
MAR	Lobster trawl survey	Lobster	western Scotian Shelf	OTB	nd	A	100	na	na	0.0211	0.0569	na	5.7	nd
MAR	Shrimp survey	N. shrimp	Scotian shelf	OTB	SR-F	A	44SR, 16F	na	na	0.0403	*0.1209*	5,499	7.3	758
MAR	Sea scallop dredge survey (Inshore)	Sea scallop	Western Scotian Shelf and Bay of Fundy	DRB	SR	A	700	na	na	0.0044	na	12,650	3.1	4,117
MAR	Sea scallop dredge survey (Offshore)	Sea scallop	Eastern Scotian Shelf and Western Scotian Shelf and Georges Bank	DRB	SR	A	600	na	na	0.0024	na	10,663	1.5	7,282
MAR	Snow crab trawl survey	Snow crab	Scotian shelf	OTB	F	A	400	na	na	0.0038	*0.0115*	125,000	4.6	nd
MAR	Surf clam survey [8]	Surf clam	Banquereau, Grand Banks	DRH	R	O	505	na	na	0.5560	na	59,583	280.8	1,061
MAR	Multispecies trawl survey (summer)	Various	Scotian shelf (4VWX)	OTB	SR	A	245	na	na	0.0405	0.1402	225,396	34.3	6,562
MAR	Multispecies trawl survey (winter)	Various	Scotian shelf (4VW5Z)	OTB	SR	A	180	na	na	0.0405	0.1402	199,297	25.2	7,897

DFO Region	Survey	Spec.	Location(s)	Gear	Design	Freq	Hauls	Length	Config.	Haul swept area[9]	Haul swept doors[10]	Survey domain	Survey swept area	Recur. interval
MAR	Multispecies trawl survey (GB - US NMFS)	Various	Georges Bank	OTB	SR	A	55	na	na	0.0211	0.0569	49,500	3.1	15,817
NL	Sentinel gillnet	Atl. cod	Nfld shelf	GNS	F	A	2100	550	na	0.0550	na	na	115.5	nd
NL	Sentinel longline	Atl. cod	Nfld shelf	LLS	F	A	238	1200	na	0.1200	na	na	28.6	nd
NL	Redfish unit 2 industry survey	Redfish		OTB	SR	B	-	na	na	-	-	-	nd	nd
NL	Scallop dredge survey	Scallop	Nfld coast - 3Ps	DRB	SR	R: 2-3 yr	200 [4]	na	na	0.0022	na	2,100	0.4	11,986
NL	Northern Shrimp Research Foundation (NSRF) survey	Shrimp	Shrimp fishing areas 2, 3(WAZ), 3(RISA), 4	OTB	SR	A	330	na	na	0.0235	0.0684	nd	22.6	nd
NL	Snow crab trap survey	Snow crab	NAFO 2J3KLNOPs4R	pots	F	A	12500	na	450m line of 10	0.0037	na	>500,000	45.8	nd
NL	Snow crab trap survey	Snow crab	Coastal Nfld (various bays)	pots	SR	A	1600	na	360m line of 8	0.0029	na	nd	4.7	nd
NL	Multispecies trawl survey (fall)	Various	Nfld & Labrador shelves	OTB	SR	A	600	na	na	0.0235	0.0684	566,758	41.0	13,810
NL	Multispecies trawl survey (spring)	Various	NAFO 3LNOP	OTB	SR	A	375	na	na	0.0235	0.0684	377,872	25.7	14,732
NL	Multispecies trawl survey (EU-Spain & Portugal)	Various	Flemish Cap (NAFO 3M)	OTB	SR	A	181	na	na	0.0454	0.2917	55,119	52.8	1,044
NL	Multispecies trawl survey (EU-Spain)	Various	Nose (3L) and tail (3NO) of Grand Bank	OTB	SR	A	222	na	na	0.0472	0.1389	35,472	30.8	1,150
PAC	Dungeness crab survey	Dung. crab	Fraser River delta & Vancouver Harbour	trap	F	B	55	na	300m ground line	0.003	na	467	1.8	nd
PAC	Prawn trap survey	Prawn	Howe Sound	trap	F	B	45	na	300m ground line	0.005	na	333	1.6	nd
PAC	Sablefish trap survey [6]	Sablefish	Pacific coast	trap	SR	A	91	na	1,200m line of 25	0.0013	na	21,668	0.12	180,567
PAC	Scallop trawl survey	Scallop	nd	OTB	SR	O	20	na	na	0.001	*0.0030*	4	0.1	nd

DFO Region	Survey	Spec.	Location(s)	Gear	Design	Freq	Hauls	Length	Config.	Haul swept area[9]	Haul swept doors[10]	Survey domain	Survey swept area	Recur. interval
PAC	Inshore shrimp survey	Var. shrimp spec.	Multiple inshore areas of BC	OTB	F	A&B	90	na	na	0.02	*0.0600*	1,767	5.4	nd
PAC	Halibut hook and line survey	Various	Pacific coast	LLS	F	A	170	2750	na	0.2753	na	nd	46.8	nd
PAC	Rockfish hook and line survey (charter)	Various	Pacific coast	LLS	SR	A	195	1100	na	0.1100	na	23,816	21.5	1,110
PAC	Rockfish hook and line survey (research vessel)	Various	Pacific coast	LLS	SR	A	70	550	na	0.0550	na	5,632	3.9	1,463
PAC	Small-mesh multispecies bottom trawl survey	Various	SW Vancouver Isl. & Queen Charlotte Sound	OTB	F	A	190	na	na	0.063	*0.1890*	6,195	35.9	nd
PAC	Multispecies trawl survey [7]	Various	Pacific coast	OTB	SR	A	350	na	na	*0.0210*	0.066	53,988	22.1	2,448
QC	Atlantic halibut industry survey	Atl halibut	Gulf of St. Lawrence	LLS	SR	A	125	1000-3500	na	0.2250	na	~115,000	28.1	4,089
QC	Sentinel gillnet	Atl. cod	northern Gulf of St. Lawrence (4RS)	GNS	F	A	5700	91	na	0.0091	na	122,913	51.9	nd
QC	Sentinel longline	Atl. cod	northern Gulf of St. Lawrence (4RS3Pn)	LLS	F	A	160	1200	na	0.1200	na	130,074	19.2	nd
QC	Post-season trap survey	Lobster	Gaspé, Magdalen Isl.	pots		A	na	na	na	na	na	na	na	nd
QC	Scallop dredge survey	Sea scallop	Magdalen Islands	DRB		B	na	na	na	na	na	na	na	nd
QC	Snow crab trawl survey	Snow crab	Estuary and northern Gulf of St. Lawrence	OTB	SR-F	B	135SR, 22F	na	na	0.0023	*0.0069*	na	1.1	nd
QC	Snow crab trap survey	Snow crab	Estuary and northern Gulf of St. Lawrence	pots	F	A	335	na	single	0.0001	na	~50,000	0.04	nd
QC	Gillnet groundfish survey	Various	Saguenay Fjord	GNS	F	B	60	274	na	0.0274	na	na	1.6	nd
QC	Sentinel mobile	Various	northern Gulf of St. Lawrence (4RS3Pn)	OTB	SR	A	287	na	na	0.0362	*0.1085*	129,221	31.1	4,149

DFO Region	Survey	Spec.	Location(s)	Gear	Design	Freq	Hauls	Length	Config.	Haul swept area[9]	Haul swept doors[10]	Survey domain	Survey swept area	Recur. interval
QC	Multispecies trawl survey	Various	northern Gulf of St. Lawrence	OTB	SR	A	180	na	na	0.0235	0.0684	125,780	12.3	10,216

[1] annual (5 year program at each site; currently 3 sites with annual longlining)
[2] annual (5 year program at each site; typically trawl near 2 communities each year)
[3] annual, episodic
[4] hauls for one complete round of surveys
[5] switching from a fixed station to random stratified design with 150 stations by 2020
[6] excludes inlet work (20 fishing events per year) that follows a fixed station design
[7] This entry covers four separate surveys, two of which are completed annually and excludes Strait of Georgia survey. The domain and swept area calculations reflect an annual average.
[8] last surveys in 2009-2010 but there are plans to resume; here assumed survey every 5 years
[9] values in italics were estimated from the swept area between the doors
[10] values in italics were estimated from the wingspread swept area

*Table 7. Summary of the advantages, disadvantages and potential biases of different survey sampling methods based on the review by Murphy and Jenkins (2010). Original sources references for these considerations are available in that paper and are not repeated here. Additional considerations from the authors of the present report are indicated by *.*

Methodology	Advantages	Disadvantages	Potential Biases
Observational: Underwater visual methods (UVM)	-Data on fine-scale fauna-habitat associations -Potential for high sampling density at a small scale -No post-survey image processing required	-SCUBA restrictions (depth, dive duration, sea state, timing with respect to tides) -Requires certified staff -Spatial coverage per site is limited; very large effort required to survey a large area -Limited to conspicuous taxa * Likely inappropriate for accurately sampling larger bodied mobile taxa (fish and squid) * Not possible to obtain physical samples of mobile taxa	-Under and over estimation of size and density of macrofauna. -influence of diver on fish behaviour
Observational: Underwater imagery including remotely operated vehicles (ROV)	-Survey at depths and times dangerous for divers. -Reduced impact on fish behavior compared to UVM -Permanent record of survey -Potential for high sampling density at a small scale -Data on fine-scale fauna-habitat associations -Possibility of targeted (limited) biological sampling using ROV	- Post-survey image processing generally required (can be extensively time consuming). - Restricted field of view - Requires high water clarity - Fish need to be perpendicular to the camera for measurement -Spatial coverage per site is limited; very large effort required to survey a large area - The deployment and retrieval of gear can be time consuming, particularly for ROV -Limited to conspicuous taxa * Towed-body cameras can impact the seafloor and epibenthic taxa. Impact expected to be limited to the swept area, with some harm due to the resuspension of sediments. * Not possible to obtain physical samples of highly mobile taxa	-Potential underestimation of density, including high frequency of zero counts
Observational: Hydroacoustics	-Non-destructive -Absolute population size can be estimated -Results obtained with only a short delay -Can survey large areas, with high resolution at a range of spatial scales -Already routinely used for pelagic and bathypelagic species	-Differentiation of species and sizes is limited; additional biological sampling using a trawl is required. -Cannot detect demersal species near the seafloor and therefore not suitable for flatfish and many other demersal groundfish species (e.g., wolfish, cusk, monkfish).	-Boat avoidance -Fish with swimbladders have higher target strengths -In the absence of additional biological sampling, length and biomass estimates can be biased.
Extractive: Trawling	-Samples demersal and bottom-dwelling species -Survey large areas in a short time; sampling at a site integrates a large surface area compared to visual or video monitoring. -Survey trawls that use a fine-mesh in the cod end sample a broad range of species and fish sizes *Little or no post-survey processing is required for the data (with the exception of information collected from the processing of biological samples post-survey) * Physical samples allow for the differentiation of conspicuous species	-Impacts on the seafloor and associated biota. Footprint is related to the area swept by the net, the trawl doors and often the sweep-lines. Resuspension of sediments. -Requires generally flat habitat -High mortality of the fish captured by the net; likely some (more limited) mortality of fish escaping the trawl. -Typically not appropriate for accurate sampling of pelagic species	-bias against highly mobile species *limitations on the areas in which trawling is possible means that survey areas may not cover the distributional areas of some species, leading to potential biases.

Methodology	Advantages	Disadvantages	Potential Biases
Extractive: Fixed fishing gear (gillnets, longlines, pots, traps)	- Samples demersal and bottom-dwelling species - Can be used effectively in areas that are inaccessible to survey bottom-trawls - Used in tagging studies to capture animals with minimal harm - All else being equal, potentially the lowest cost option * Little or no post-survey processing is required for the data (with the exception of information collected from the processing of biological samples post-survey) * Physical samples allow for the differentiation of conspicuous species	-Impacts on the seafloor and associated biota. Footprint is related to that of the gear and any sweeping of the seafloor during deployment and retrieval. -More species and size selective than bottom-trawls *Lost gear contributes to seafloor litter and may cause ghost fishing in the case of gillnets and pots/traps. -Can be very labour intensive if there is a need to cover a large area synoptically	-biases with respect to selectivity

Table 8. Summary of the catches of the four species of sea pens in the multi-species research vessel (RV) survey of the sGSL for 2014-2017: number of occurrences, mean catch weight, and maximum catch weight.

Species	Number of occurrences	Mean catch (kg)	Max. catch (kg)
Halipteris finmarchica	14	0.129	0.51
Anthoptilum grandiflorum	55	2.158	30.30
Pennatula aculeata	58	0.085	1.07
Pennatula grandis	43	3.072	35.00

Table 9. Percentage overlap between the sGSL RV survey strata (Str) and the sea pen significant benthic areas (SBA) or sea pen conservation areas (SCA). Refer to figure 3 for the SBA and SCA locations and figure 5 for the stratum locations.

SBA or SCA	% of stratum overlapping the area				% of area overlapping the stratum			
	Str 415	Str 416	Str 425	Str 439	Str 415	Str 416	Str 425	Str 439
SBA A	17.16	0	0	0	10.74	0	0	0
SBA B	32.08	4.99	44.61	0	11.63	2.46	14.11	0
SBA C	0	0	8.89	52.11	0	0	11.76	35.22
SCA 1	13.67	0	0	0	14.24	0	0	0
SCA 2	0	0	7.34	0	0	0	19.86	0
SCA 3	0	0	2.99	4.44	0	0	20.64	14.99

Table 10. Annual number of survey sets in each stratum (415, 416, 425, 439) that were included and excluded from the calculation of abundance indices based on overlap between the survey and sea pen significant benthic areas (SBA) or sea pen conservation areas (SCA).

	Included (SBA)				Excluded (SBA)				Included (SCA)			Excluded (SCA)		
Year	415	416	425	439	415	416	425	439	415	425	439	415	425	439
1971	2	6	1	1	1	0	1	1	3	2	2	0	0	0
1972	2	6	1	2	1	0	1	0	3	2	2	0	0	0
1973	1	6	1	1	2	0	1	1	3	2	2	0	0	0
1974	3	4	2	2	0	0	0	0	3	2	2	0	0	0
1975	3	4	1	2	0	0	1	0	3	2	2	0	0	0
1976	2	6	2	2	0	0	0	0	2	2	2	0	0	0
1977	3	5	1	0	0	0	1	2	3	2	2	0	0	0
1978	1	5	2	1	1	0	0	1	2	2	2	0	0	0
1979	1	6	1	1	2	0	1	1	3	1	2	0	1	0
1980	2	6	1	0	1	0	1	2	3	2	2	0	0	0
1981	1	6	2	0	2	0	0	2	3	2	2	0	0	0
1982	1	5	1	0	2	0	1	2	3	2	2	0	0	0
1983	2	5	1	1	1	0	1	1	3	2	2	0	0	0
1984	2	7	2	1	1	0	1	1	3	3	2	0	0	0
1985	5	13	2	3	4	0	2	2	8	4	5	1	0	0
1986	3	9	2	1	3	0	1	1	5	3	2	1	0	0
1987	3	9	2	2	3	0	1	1	5	3	3	1	0	0
1988	3	7	2	1	1	0	1	1	4	2	2	0	1	0
1989	3	6	4	2	3	1	0	2	6	4	4	0	0	0
1990	5	8	2	2	0	0	1	1	5	2	3	0	1	0
1991	5	9	2	2	2	0	3	2	5	4	3	2	1	1
1992	3	8	3	1	2	1	1	2	4	3	3	1	1	0
1993	5	8	2	3	0	1	3	2	5	5	5	0	0	0
1994	3	8	2	2	2	0	2	1	4	4	3	1	0	0
1995	2	7	1	2	3	1	4	2	4	5	3	1	0	1
1996	3	9	2	3	4	0	4	2	5	5	5	2	1	0
1997	3	7	3	2	4	1	2	2	6	5	4	1	0	0
1998	3	8	3	1	3	0	2	3	6	5	3	0	0	1
1999	3	8	2	2	3	0	3	2	6	4	4	0	1	0
2000	4	8	3	1	2	0	2	3	6	5	4	0	0	0
2001	2	6	2	2	3	1	2	2	4	3	4	1	1	0
2002	5	9	2	2	1	0	3	2	6	5	4	0	0	0
2003	2	3	0	0	1	0	1	0	3	1	0	0	0	0
2004	7	11	0	2	2	1	5	2	8	3	3	1	2	1
2005	4	16	2	4	2	0	6	2	6	8	6	0	0	0
2006	3	7	3	2	3	1	2	2	4	5	4	2	0	0
2007	3	7	1	2	3	1	4	2	6	4	4	0	1	0
2008	6	7	2	3	1	1	4	2	7	6	5	0	0	0
2009	3	7	1	3	3	1	3	1	4	4	4	2	0	0
2010	3	8	2	1	2	0	1	2	5	3	3	0	0	0
2011	3	6	1	2	1	0	3	1	4	3	3	0	1	0
2012	3	5	2	1	2	0	3	2	4	4	3	1	1	0
2013	3	5	0	1	1	1	3	2	4	2	2	0	1	1
2014	3	8	3	1	3	0	2	2	5	4	2	1	1	1
2015	2	8	2	2	4	0	3	3	6	4	4	0	1	1
2016	3	8	2	2	3	0	3	1	5	5	3	1	0	0
2017	3	6	2	0	3	0	2	3	5	3	2	1	1	1

Table 11. Average annual proportion of the areas impacted by trawl surveys and the average recurrence time (years) of the survey for the portions of the sea pen significant benthic areas (SBA) or sea pen conservation areas (SCA) falling respectively in the southern and northern portions of the Gulf of St. Lawrence. Refer to figure 3 for the SBA and SCA locations and figure 5 for the stratum locations.

SBA or SCA	Surface area (km^2)	Proportion impacted		Recurrence time (yr)	
		southern	northern	southern	northern
SBA A	3,892	0.00043	0.00031	2,354	3,241
SBA B	10,153	0.00069	0.00020	1,458	5,017
SBA C	1,638	0.00175	0.00031	571	3,185
SCA 1	2,338	0.00033	0.00026	2,997	3,783
SCA 2	821	0.00160	0.00025	625	4,031
SCA 3	335	0.00185	0.00006	540	15,763

12. FIGURES

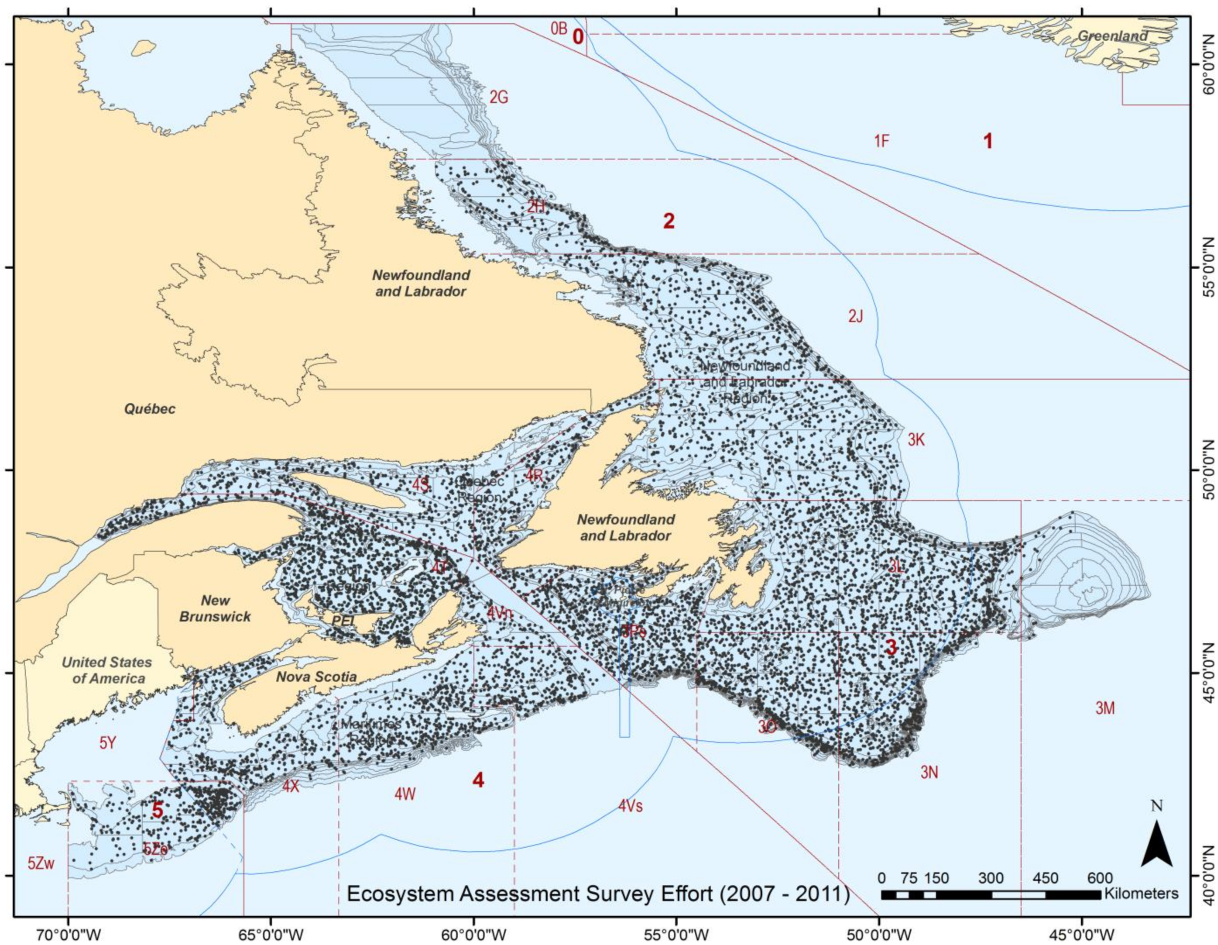

Figure 1.The distribution of individual survey hauls for the DFO multispecies bottom-trawl surveys in Atlantic Canada for the five-year period 2007 to 2011. The map does not include multispecies bottom-trawl survey hauls on the Flemish Cap and the nose and tail of the Grand Bank undertaken by Spain and the European Union, nor those on Georges Bank and the western Scotian Shelf undertaken by the U.S.A., and in the Northumberland Strait by the DFO inshore survey.

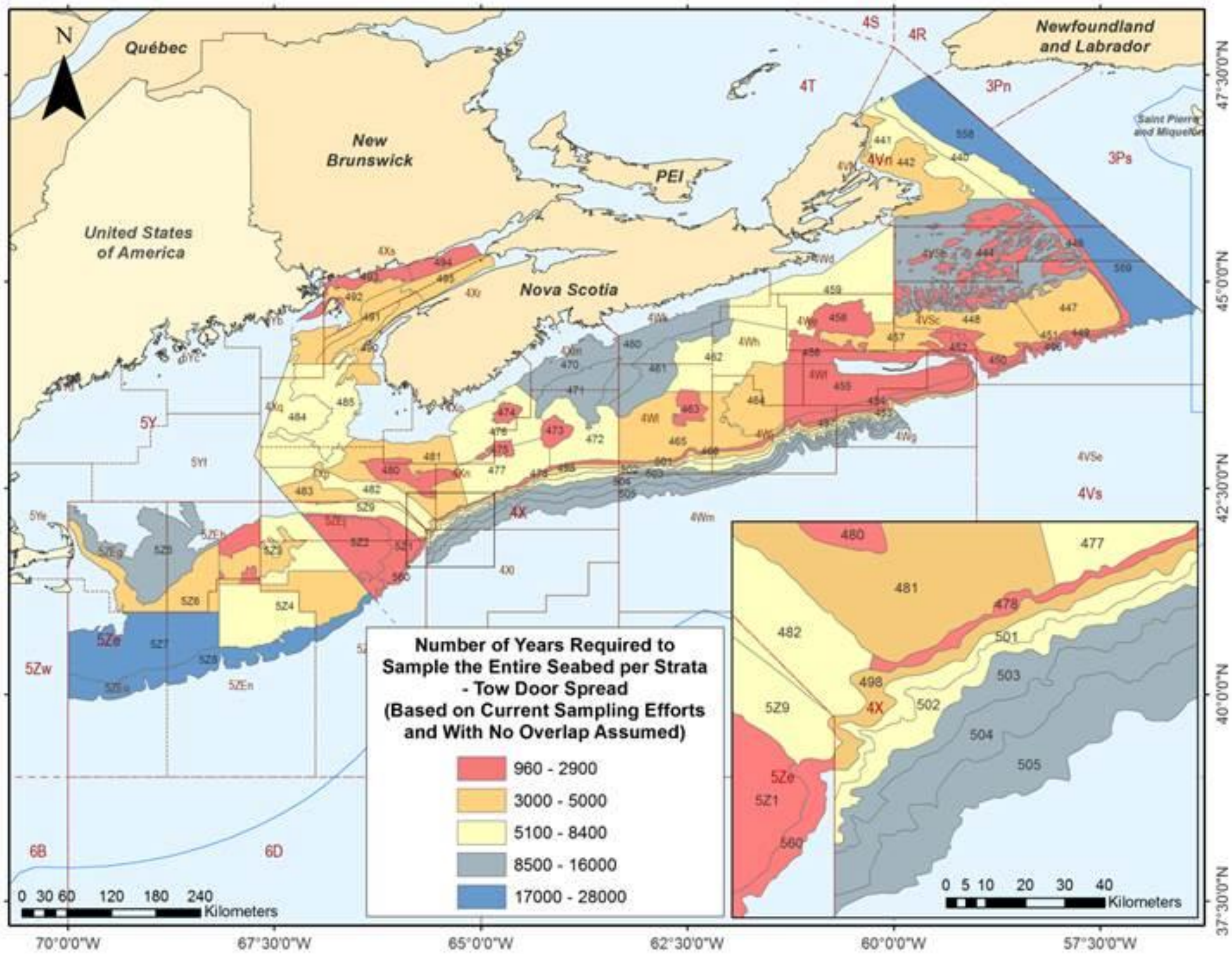

Figure 2. Mean survey disturbance interval (years) for individual survey strata in the multispecies bottom trawl surveys undertaken by the DFO Maritimes region.

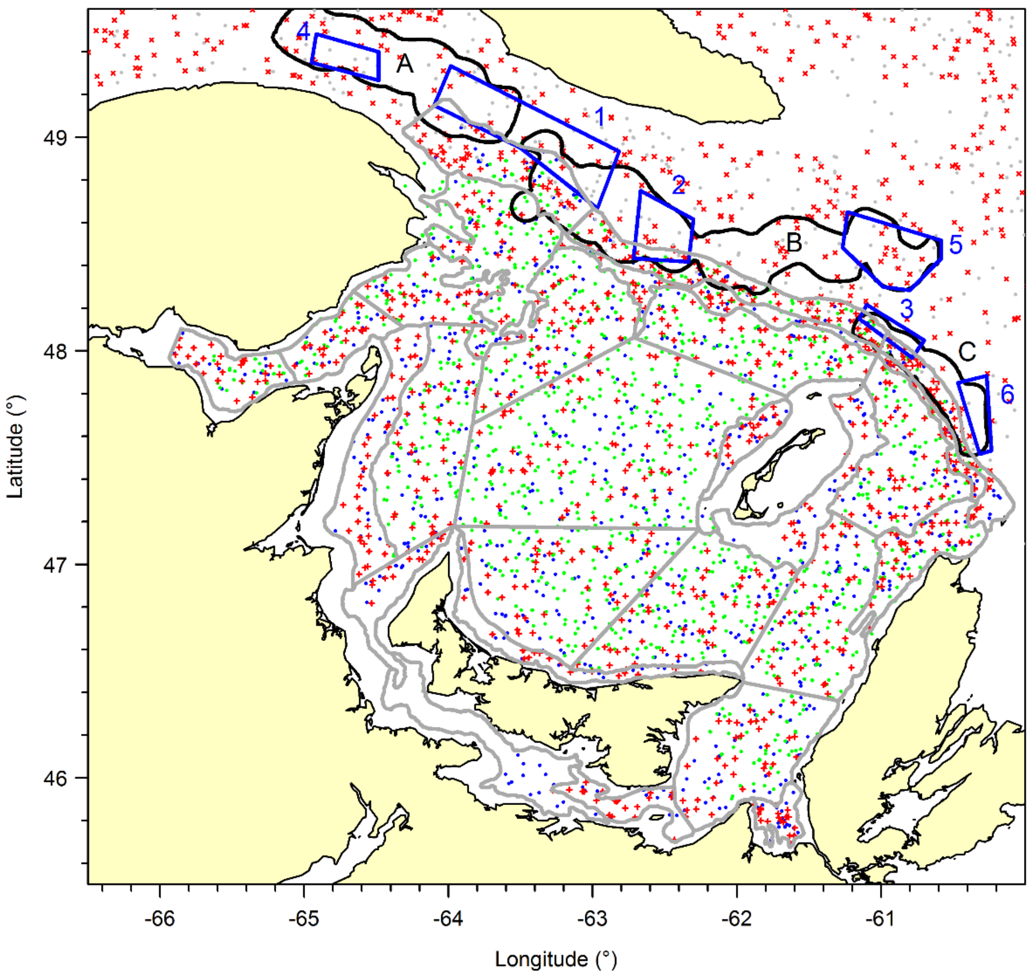

Figure 3. Major sea pen significant benthic areas (SBA; black polygons; from Kenchington et al. 2016), sea pen conservation areas (SCA; blue polygons) and RV survey strata (grey polygons) in the southern and central Gulf of St. Lawrence along with the location of trawl hauls (2011-2015) for the five bottom-trawl surveys that occur in these areas (points). The SBAs are labelled with capital letters, whereas the SCAs are labelled with numbers, corresponding to the following areas: 1-Honguedo-East (Eastern Honguedo Strait Coral and Sponge Conservation Area), 2-Bennett Bank (North of Bennett Bank Coral Conservation Area), 3- Magdalen Shallows Slope (Slope of Magdalen Shallows Coral Conservation Area), 4- Honguedo-West (Western Honguedo Strait Coral Conservation Area), 5- Gulf-Centre (Central Gulf of St. Lawrence Coral Conservation Area), 6- Gulf-East (Eastern Gulf of St. Lawrence Coral Conservation Area). The survey haul locations are identified for each survey: southern Gulf RV (blue dots), northern Gulf RV (grey dots), southern Gulf Sentinel (red +), northern Gulf Sentinel (red x) and snow crab (green dots).

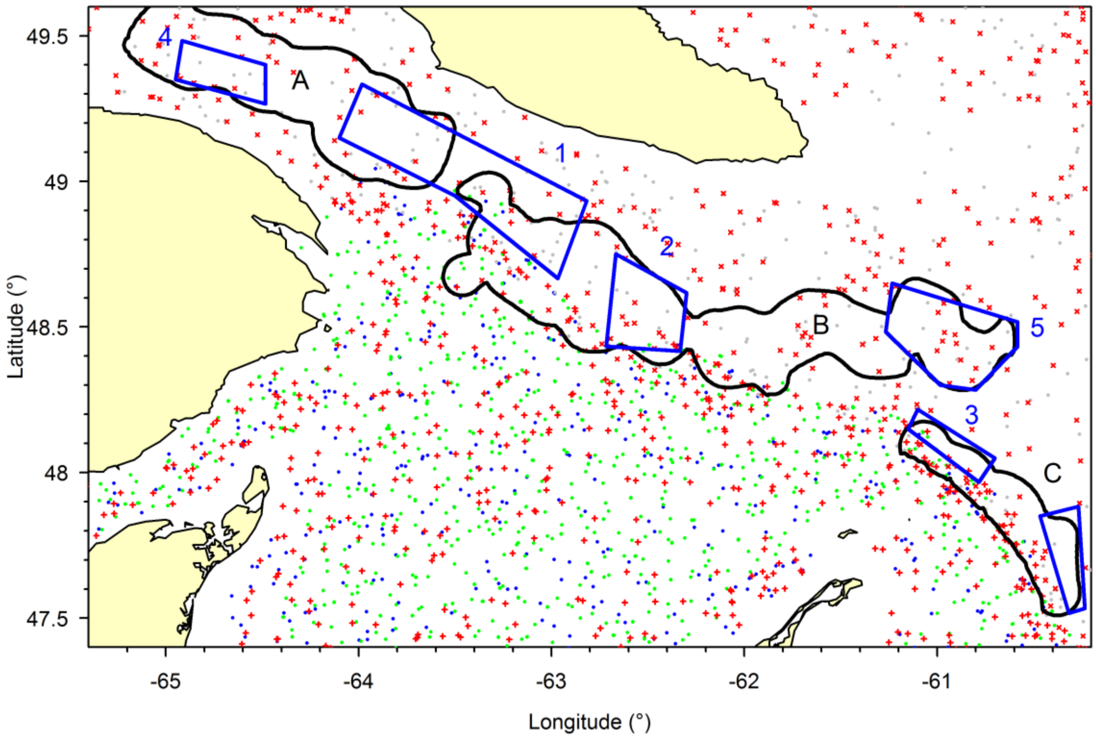

Figure 4. Identical to Figure 3, but showing a close-up view of the SBAs and SCAs, and excluding the southern Gulf RV strata boundaries for enhanced clarity.

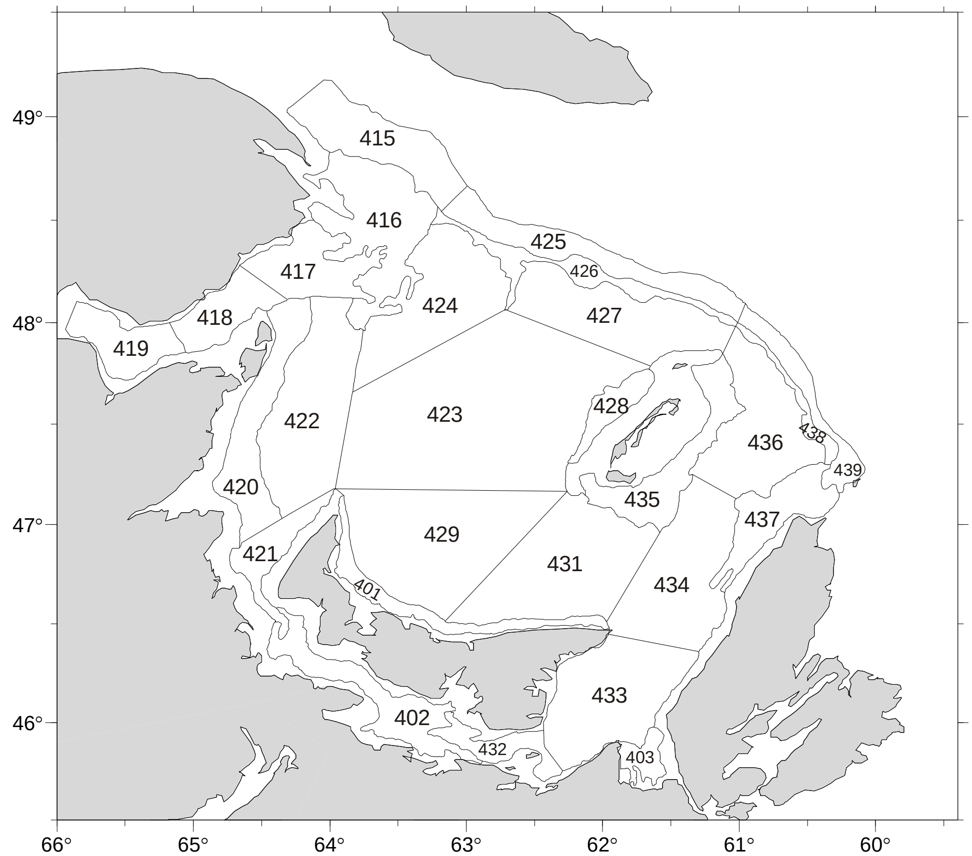

Figure 5. Stratum boundaries for the southern Gulf of St. Lawrence September RV survey.

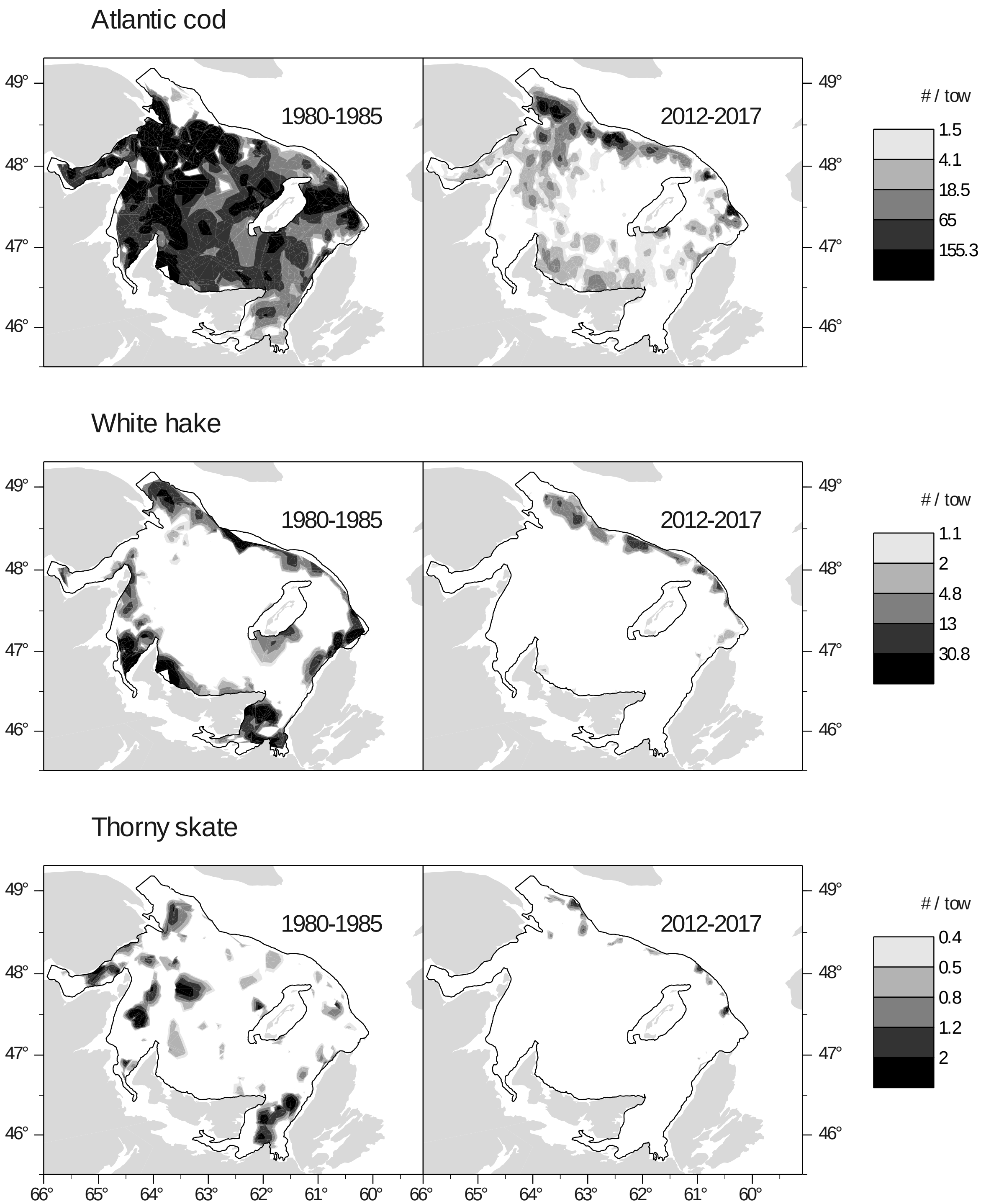

Figure 6. Spatial distribution of catch for adult Atlantic Cod, White Hake and Thorny Skate (rows) in two time periods, 1980 to 1985 and 2012 to 2017 (columns) in the southern Gulf of St. Lawrence bottom-trawl survey. Shading levels for each species are based on five quantiles for non-zero catch over the 1971-2017 period: 0.10, 0.25, 0.50, 0.75 and 0.90. For the present purposes the following sizes of fish were used to define adults: ≥39 cm for Atlantic Cod, ≥45 cm for White Hake and ≥51 cm for Thorny Skate (Swain et al. 2012c, 2012d, 2016).

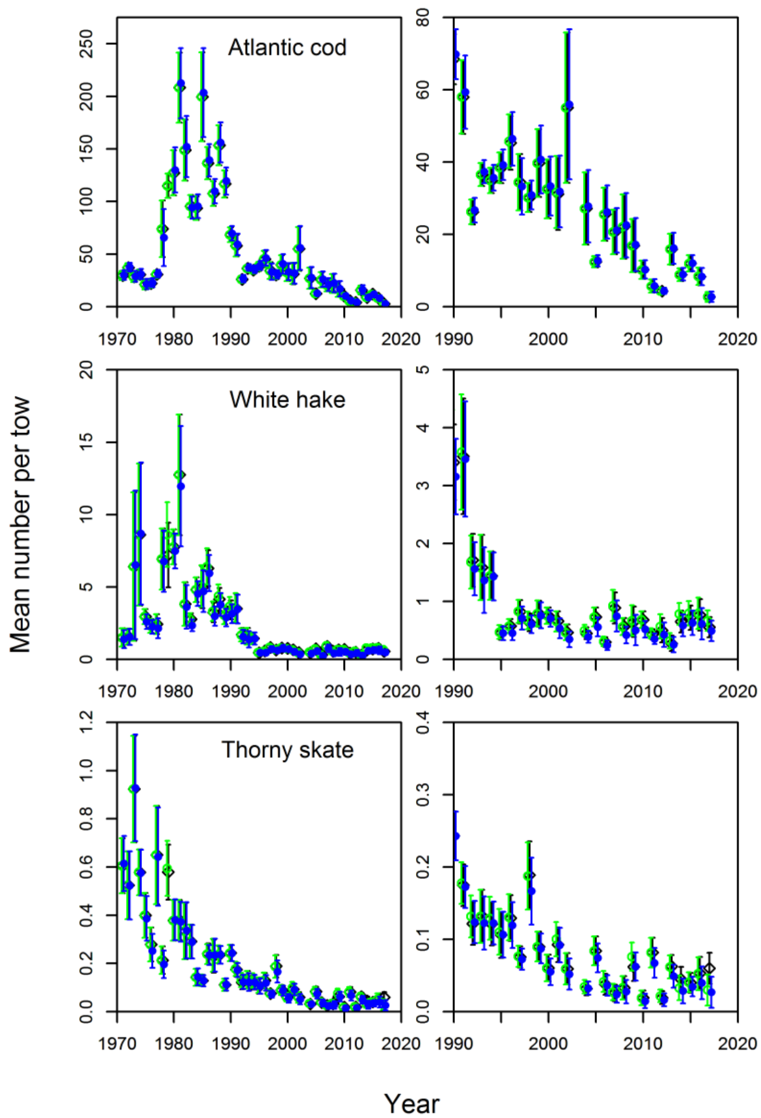

Figure 7. Estimated mean catch per tow (±SE) of adults of the three fish species (rows) based on the full survey data (black symbols), and the survey excluding sets in the three overlapping SCAs, either excluding the SCAs from the survey domain (blue closed symbols) or imputing catch for those areas and maintaining the existing survey domain (green open symbols). The abundance indices are presented for the 1971-2017 period (left column) and in more detail for the 1990-2017 period (right column). Note the different y-axis scales between the plots.

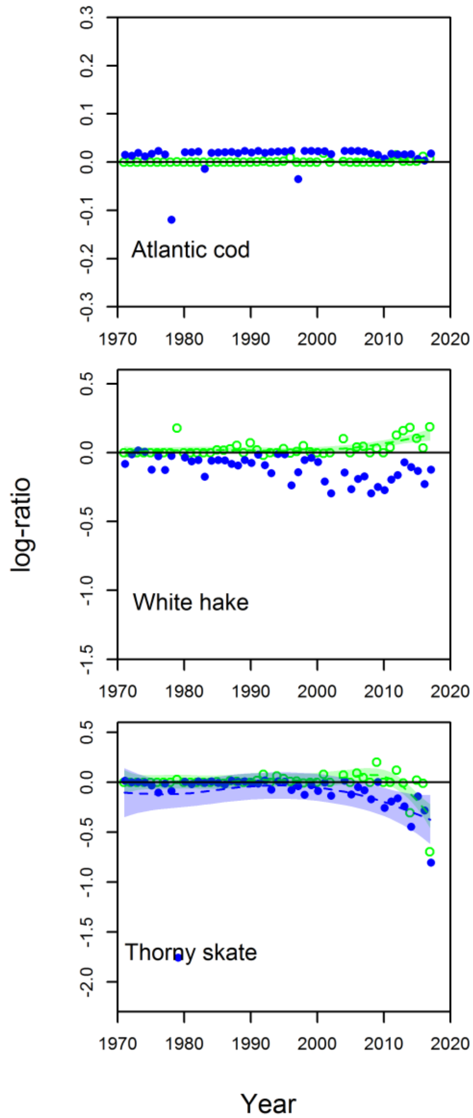

Figure 8. Difference in the log-mean catch per tow of adults of the three fish species (rows) for estimates based on the full survey data and for estimates based on excluding sets in the three overlapping SCAs, either excluding the conservation areas from the survey domain (blue closed symbols) or imputing catch for those areas and maintaining the existing survey domain (green open symbols). Predicted differences and their confidence interval (dashed line and shaded area) are shown for those cases for which a statistically significant long term trend in differences ($p < 0.05$) was inferred using GAM analysis. Note the different y-axis scales between the plots.

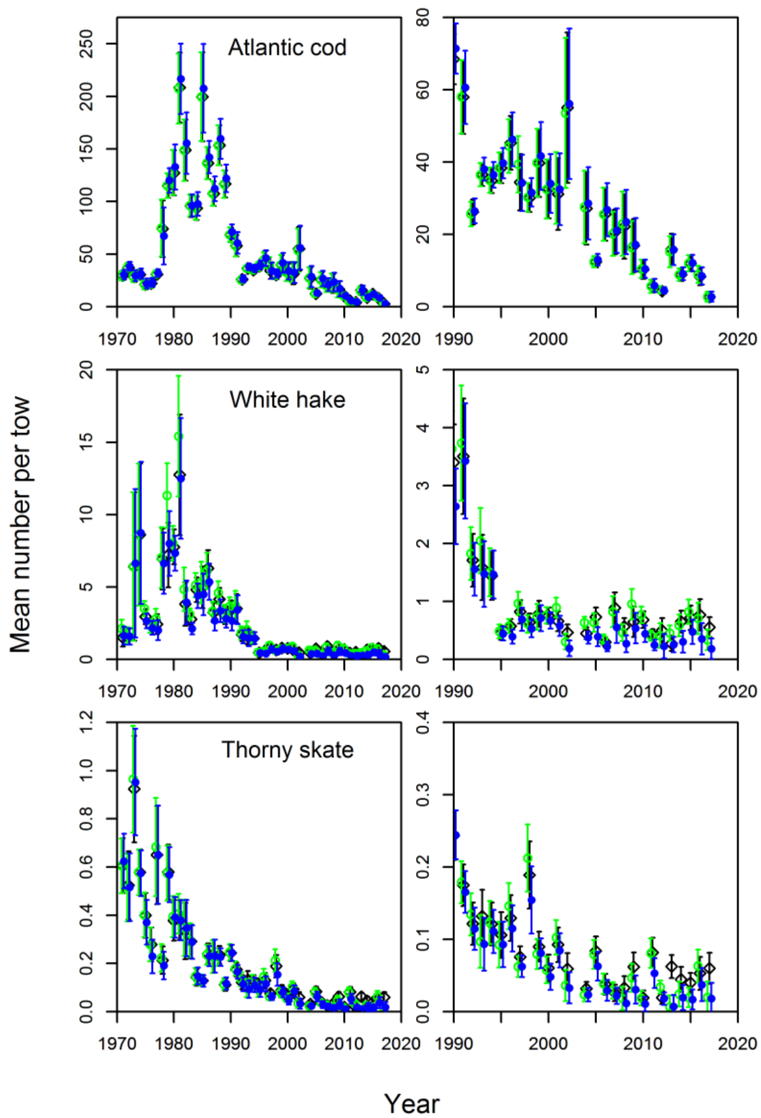

Figure 9. Estimated mean catch per tow (±SE) of adults of the three fish species (rows) based on the full survey data (black symbols), and the survey excluding sets in the three overlapping SBAs (Kenchington et al. 2016), either excluding the SBAs from the survey domain (blue closed symbols) or imputing catch for those areas and maintaining the existing survey domain (green open symbols). The abundance indices are presented for the 1971-2017 period (left column) and in more detail for the 1990-2017 period (right column). Note the different y-axis scales between the plots.

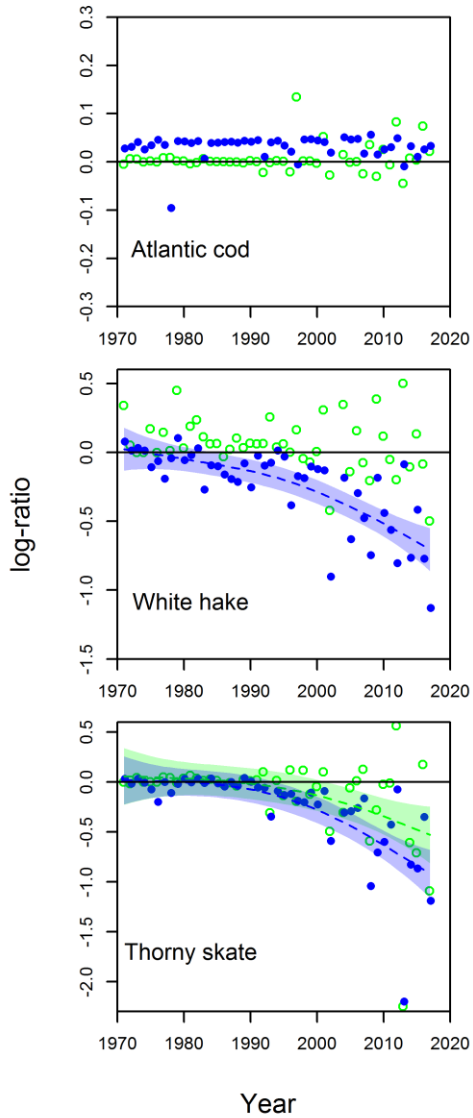

Figure 10. Difference in the log-mean catch per tow of adults of the three fish species (rows) for estimates based on the full survey data and for estimates based on excluding sets in the three overlapping seapen aggregation areas, either excluding the conservation areas from the survey domain (blue closed symbols) or imputing catch for those areas and maintaining the existing survey domain (green open symbols). Predicted differences and their confidence interval (dashed line and shaded area) are shown for those cases for which a statistically significant long term trend in differences ($p < 0.05$) was inferred using GAM analysis. Note the different y-axis scales between the plots.