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A Critical Review of Monitoring Needs
and Strategies for Marine Protected Areas
and Areas of Interest in Newfoundland
and Labrador



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A Critical Review of Monitoring Needs and Strategies for Marine Protected Areas
and Areas of Interest in Newfoundland and Labrador

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Section 1 MPAs as a global approach to marine management.

Sustaining ocean health requires a broad management focus that extends beyond stock assessments and considers how to maintain ecosystem components necessary for ecological processes. Maintenance of ocean resources requires healthy, intact ecosystems to support them. The implementation of Marine Protected Areas (MPAs) represents one key management strategy to address a variety of issues that affect marine ecosystems. MPAs, as defined by the International Union for Conservation of Nature (IUCN), are “*clearly defined geographical spaces, recognized and managed, through legal or other effective means, to achieve the long-term conservation of nature with associated ecosystem services and cultural values*” (IUCN-WPCA 2008). Anthropogenic activities increasingly threaten marine ecosystems, and no management system guarantees success in attaining specific conservation goals; however, MPAs offer a potential buffer to unknowns inherent in marine ecosystems (Christie & White 2007), and also provide numerous potential benefits to multiple species, including those not targeted by fisheries (Staub & Hatzios 2004, Stewart et al. 2009). In particular, MPAs offer a pragmatic management solution to help mitigate threats posed by ocean industries (e.g. fishing) that can adversely affect biodiversity (Cook et al. 2013) and modify ecosystem structure and function (Olsgard et al. 2008, Garcia et al. 2012).

Currently MPAs protect less than 2.5% of the world’s oceans in some capacity (Spalding et al. 2013), of which a smaller subset (<0.5%) restrict all extractive activity. While this coverage represents a global increase from earlier estimates of ~ 0.65% (Wood et al. 2008, Jessen et al. 2011), a relatively few very large (>100,000 km²) marine protected areas account for much of that growth (Spalding et al. 2013, Toonen et al. 2013, Devillers et al. 2014). Despite this growth, and given a widely noted goal of 10-30% ecoregion coverage for global MPAs (e.g. 10% by 2020: Convention on Biological Diversity), much work remains to achieve that objective.

Studies published almost weekly support MPA use, indicating widespread recognition for the application of MPAs as a management tool. Particularly for protecting biodiversity and marine habitat, MPAs likely represent one of the best tools available (IUCN-WPCA 2008). While many recognize the potential utility of MPAs, they represent just one of many management tools that could be applied to marine systems (Hilborn et al. 2004). The efficacy of an MPA

as a management approach hinges on balancing protection for the ecosystem, the magnitude of change evoked by the intervention (i.e., reduction in fishing effort) (Devillers et al. 2014), stakeholder support (Roberts et al. 2003b), and the associated conservation objectives. MPAs, though conceptually simple, may not be the best or most pragmatic management solution to all problems (Boersma & Parrish 1999, Hilborn et al. 2004). Directly incorporating MPAs into existing management plans (e.g. egg conservation in Eastport Lobster Management Area; Section 3.3.2) is a suitable first step. Indeed the most pragmatic approach to achieve management objectives will likely be through integration of MPAs into existing management approaches, such as total allowable catches (TACs), individual transferable quotas (ITQs) and seasonal restrictions. For example, conventional management through effort restrictions (e.g. season and gear) might well produce equivalent or better results than MPAs for fisheries targeting highly mobile species by limiting by-catch and habitat impact (Hilborn et al. 2004). In Newfoundland and Labrador, the Eastport MPA (*see section 3.3.2*) is among several key management actions employed to manage the lobster fishery. Gear restrictions, minimum size requirements, v-notching and the MPA itself all seek to conserve egg production to mitigate recruitment limitation in the Eastport area (DFO 2013a). Given the small size of the Eastport MPA and its potential contribution relative to the management area, its utility as a singular conservation approach will invariably fail in achieving the objective of maintaining a viable population of lobsters for the Eastport management area. Successful establishment of MPAs will be predicated on a robust and clear policy framework to integrate MPAs within existing regulations based on science-based decision making.

In short, marine protected areas, coupled with traditional management tools, can achieve a broad range of management objectives in marine systems. The successful application of MPAs requires careful planning and, importantly, thorough evaluation of their efficacy (Hilborn et al. 2004).

Section 1.1 Important considerations for the application and monitoring of MPAs

Effective management based on marine spatial closures depends on thorough feedback of information to gauge biological response relative to conservation objectives. Management effectiveness can be defined broadly as the degree to which management actions achieve the goals and objectives of the closure (Roberts et al. 2003a). The establishment of MPA systems worldwide has sparked growing interest in developing and implementing objective evaluation frameworks by which to measure management effectiveness. Through critical evaluation, managers can assess future needs and adapt current practices to improve management effectiveness and optimize human and financial resources (Christie & White 2007). The placement of suitable and measurable objectives ultimately constrains the ability of science to gauge efficacy of marine spatial management interventions. Indeed, the success and acceptance of MPAs hinges upon clear articulation of the management objective the MPA was intended to address (Agardy et al. 2003). To date, most MPA evaluations lack sufficient data to gauge performance objectively (Jones 2001), thus limiting potential for adaptability and long-term success (Pomeroy et al. 2005, Fox et al. 2014).

A multitude of factors define the most suitable monitoring framework for a given marine protected area, beginning with the specific conservation objectives of the MPA. Spatial-temporal scales and the biological attributes of the area chosen for management intervention should be considered when developing the best indicator(s) for a given objective and when framing the expected and potential outcomes of the management action.

Section 1.1.1 Spatial scale considerations

The residency of an organism within MPA boundaries constrains the protection a MPA can potentially offer. MPAs offer less protection to species with ranges beyond the MPA boundaries than sessile species or those with small home ranges. In this sense, design and monitoring must consider movement patterns of focal species (*see Table 1 for a review of monitoring and dispersal scales*) and the objectives of the MPA. MPAs are established for a wide diversity of objectives, ranging from protecting specific populations or habitat features to protecting a representative area of a larger ecosystem. For the purposes of this review we will discuss MPAs under two broad categories; 1) those implemented for

the conservation of ecosystems, habitat, and biodiversity (*conservation based*); and 2) those implemented as a tool for fisheries enhancement (*fisheries based*). Although these two types of MPAs can be synonymous, with potentially mutual benefits, differences in framing their objectives drive changes in the appropriate spatial scales, and thus, how they are managed and evaluated.

Table 1. Review of literature documenting the relative scales of movement (represented as a spatial extent) for larval, juvenile and adult life history stages within MPA systems. Citations are sorted by life history stages (white – larval; light grey - Juvenile; blue – Adult). We also included movement studies that complement the focal species of NL MPAs but are not specifically focused on MPA management areas (bold).

Location	Effect Measured	Spatial Extent	Reference
Isle of Man fisheries closed areas (UK)	Sprat settlement by distance of great scallop (<i>Pecten maximus</i>)	< 1 km	(Beukers-Stewart et al. 2004, Beukers-Stewart et al. 2005)
Exuma Cays Land and Sea Park (Bahamas)	Abundance of Queen conch (<i>Strombus gigas</i>) larvae in water column	2 km	(Stoner & Ray 1996)
Dwesa MPA (South Africa)	Dispersal of larval eye limpit (<i>Cymbula oculus</i>)	6 km	(Branch & Odendaal 2003)*
Locally managed marine areas (Verata Fiji)	Dispersal of larval blood cockle clam (<i>Anadara</i> sp.)	2-10 km	(Tawake & Aalbersberg 2008)
La Laguna MPA (Spain)	Dispersal of larval China limpet (<i>Patella aspera</i>)	2-10 km	(Hockey & Branch 1994)
Gaztelugatxe Marine Reserve (Spain)	Density and biomass of goose barnacle (<i>Pollicipes pollicipes</i>)	10-12 km	(Borja et al. 2006)
George’s Bank closed area (US)	CPUE & settlement of scallop (<i>Placopecten magellanicus</i>) downstream of reserve	20 km	(Murawski et al. 2000, Murawski et al. 2005)
Goukamma & Dwesa	Dispersal of larval brown	25-27 km	(Pelc et al. 2009)

MPAs (South Africa)	mussel (<i>Perna perna</i>)		
Great Barrier Reef (Australia)	Parentage analysis of juvenile reef fish species	30 km	(Harrison et al. 2012)
Gulf of St. Lawrence (Canada)	Dispersal distance of larval American lobster (<i>Homarus americanus</i>)	10-90 km	(Stanley et al. in preparation)
Puerto Peñasco reserve (Mexico)	Larval dispersal and recruitment (model) of rock scallop (<i>Spondylus calcifer</i>) & black murex snail (<i>Hexaplex nigritus</i>)	100-200 km	(Cudney-Bueno et al. 2009)
Gulf of Maine (US) & Southwestern Nova Scotia (Canada)	Dispersal distance of larval American lobster	50-250 km	(Incze et al. 2010)
Coastal Newfoundland (Canada)	Otolith chemistry of juvenile Atlantic cod (<i>Gadus morhua</i>)	10-80 km	(Stanley et al. submitted)
Saltmarsh habitat (UK)	Salt marsh fish species	< 1 km	(Green et al. 2012)
La Laguna MPA (Spain)	Density of juvenile China limpits (<i>Patella aspera</i>)	1.4 km	(Hockey & Branch 1994)
Banyuls, Carry-le-Rouet, Endoume, Portofino, Elba Island, Meda Gran & Blanes MPAs (Mediterranean)	Mortality rate of <i>Diplodus</i> (<i>puntazzo</i> , <i>sargus</i> & <i>vulgarus</i>)	<2 km ²	(Macpherson et al. 1997)
Terra Nova National Park (Canada)	Acoustic tagging of juvenile Greenland cod (<i>Gadus ogac</i>)	0.29 – 3.57 km ²	(Shapiera et al. 2014)
Apo Islands MPA (Philippines)	Diver survey of reef fish diversity	0.5 km	(Russ & Alcala 1996, Abesamis et al. 2006,

			Russ & Alcala 2011)
Cerbère-Banyuls (France) Marine Reserve	Mean squared displacement of dusky groupers (<i>Epinephelus marginatus</i>)	<0.5 km ²	(Pastor et al. 2009)
Waikiki Marine Life Conservation District (US)	Acoustic tracking of unicornfish (<i>Naso unicornis</i>)	< 0.5 km ²	(Meyer & Holland 2005)
Barbados Marine Reserve	Abundance of various reef fish species	< 1 km	(Chapman & Kramer 1999)
Rock fish conservation areas (Canada)	Diver surveys of rock fish abundance	< 1 km	(Cloutier 2010)
Marine Extractive Reserve of Corumbau (Brazil)	Abundance and biomass of various reef fish	< 1 km	(Francini-Filho & Moura 2008)
Monte da Gula Marine Reserve (Portugal)	Home range of dusky groupers	< 1 km ²	(Afonso et al. 2011)
Kvernskjær lobster reserve	Movement of adult European lobster (<i>Homarus gammarus</i>)	1.6 km	(Huserbraten et al. 2013)
Eastport Marine Protected Area	Adult American lobster tagging	1-5 km	(Rowe 2001)
Carry, Cerbère - Banyuls, Medes, Cabrera, Tabarca, and Cabo de Palos MPAs (Western Mediterranean)	Biomass of various reef fish species	2.5 km	(Goni et al. 2008)
Mombasa Marine National Park (Kenya)	Diver survey of reef fish abundance	< 2.5 km	(McClanahan & Kaunda Arara 1996)

Columbretes Islands Marine Reserve (Spain)	Commercial landings (biomass) of spiny lobster	< 5 km	(Goni et al. 2006, Goni et al. 2010b)
Kåvra MPA (Sweden)	Abundance vs distance to MPA of European lobster (<i>Homarus gammarus</i>)	7 km	(Moland et al. 2013b)
Gilbert Bay Marine Protected Area	Acoustic tagging of adult Arctic charr (<i>Salvelinus alpinus</i>)	Up to 25 km	(Morris & Green 2012)
Leigh Marine Reserve (New Zealand)	Commercial fishing CPUE of spiny lobster (<i>Jasus edwardsii</i>)	< 1 – 30 km	(Kelly et al. 2002)
Columbretes Islands Marine Reserve (Spain)	Egg production per unit area of spiny lobster (<i>Palinurus elephas</i>)	>50 km	(Diaz et al. 2011)
Gilbert Bay Marine Protected Area	Acoustic tagging of adult Atlantic cod	40 km or 270 km ²	(Morris et al. 2014)
Californian coast fisheries closure areas	Mark-recapture of striped marlin (<i>Kajikia audax</i>)	~ 1000 km	(Jensen et al. 2010)
Gulf of Maine	Pop-up satellite tags for spiny dogfish (<i>Squalus acanthias</i>)	~ 1200 km	(Carlson et al. 2014)
Atlantic ocean	Tagging studies of leatherback sea turtles (<i>Dermochelys coriacea</i>)	~ 5000 km	(James et al. 2007)

*** No reserve effect found**

The adult home range, or ‘neighborhood’ (Palumbi 2004), of species varies greatly from 100-1000 km (e.g. sea turtles, tuna, and marine mammals; James et al. 2007), to 10-100 km (e.g. American lobster (*Homarus americanus*), Comeau & Savoie 2002), to <10 km (e.g. species with demersal eggs; Bradbury et al. 2008). The “neighborhood” of the target species or community is critical because mismatches in MPA scale can lead to gaps in protection and potential

mortality (Palumbi 2004), especially when fishing effort concentrates at the borders of the MPA (Kellner et al. 2007, Gruss et al. 2011). Species whose ecological ambit most closely resembles the size of the MPA should respond most effectively to protection (Palumbi 2004), a foundational principle for MPAs focusing on conservation of ecosystems.

In a general review of marine home ranges, Palumbi (2004) noted that, despite high variability, the adult home ranges for benthic and commercially exploited fishes typically fall within a 10-100 km range (Table 2). This generality provides a potential spatial scale template for marine protection. Globally, only 20% of individual MPAs fall into this range of protection, with a global median MPA size of 4.7 km² (Wood et al. 2008; Figure 1). These numbers suggest that a high percentage of global MPAs are designed at a scale that favours emigration. Aggregation of data from the world protected area database (IUCN 2015), clearly shows that the objectives for MPAs in the 10-400 km² globally favour conservation-based objectives (limited activity; IUCN categories I-III; Supplemental Table 1), whereas very large (<400 km²) and very small (>5 km²) designs prioritize multiple usage and production (IUCN categories IV-VI; Supplemental Table 1) (Figures 1-2), noting recent notable exceptions of large strictly no-take areas (Supplemental Table 2).

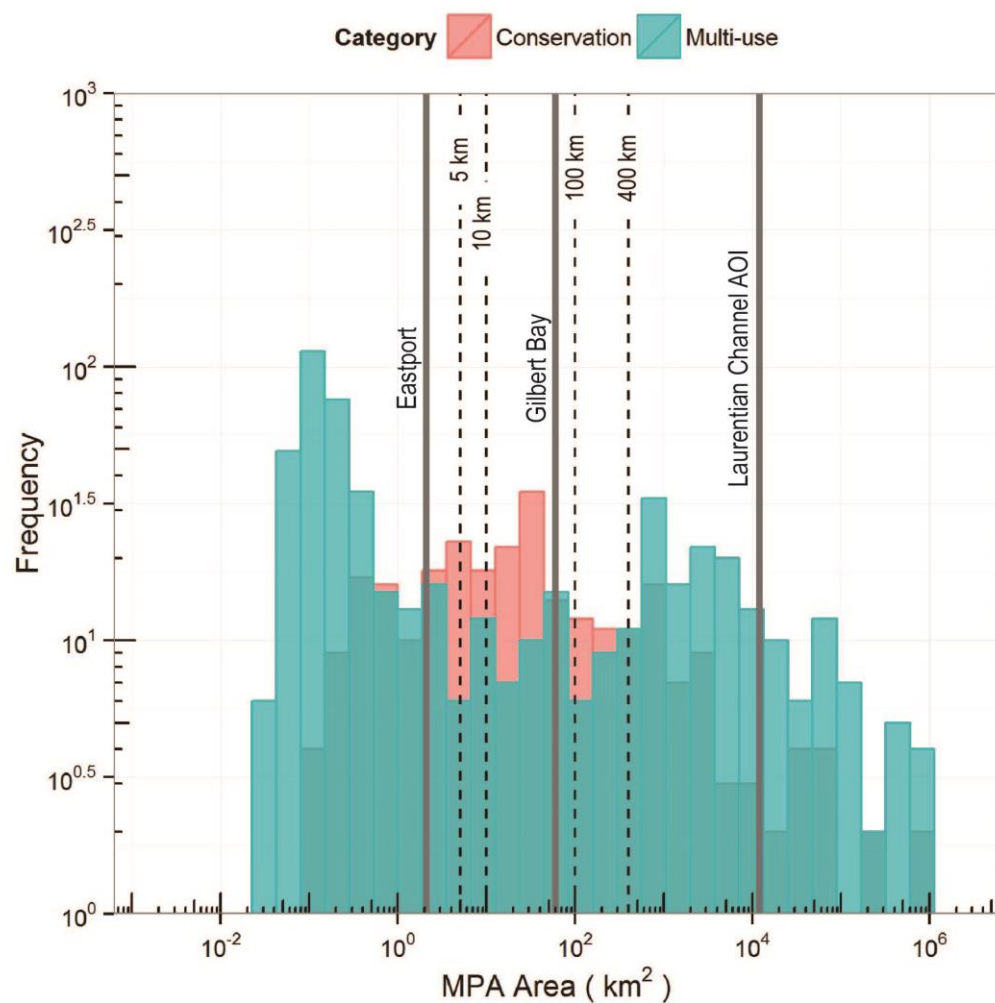


Figure 1 Distribution of MPA size according to the World Protected Area Database¹. Data is aggregated into two broad levels of activity: those which limit human activity (IUCN Categories I-III) and those that permit some (IUCN Categories IV-VI). Vertical dashed lines delineate the size classifications used in this review; very small (0-5 km), small (5-10 km), medium (10-100 km), large (100-400 km) and very large (400 km). Solid grey lines correspond to the size of the current and proposed MPAs in the Newfoundland region.

Table 2. General trends in adult and larval spatial ranges for contrasting life histories* (excerpt from Palumbi 2004).

Scale (km)	Adult	Larval
>1000s	Large migratory species	Intermittent gene flow, many

¹ Data subsetting to management areas in the ocean classified as marine reserves, monuments, sanctuaries, parks or protected areas. Refer to Supplemental R-Code for analysis details. Only MPAs with IUCN categorization were used (n=325).

		species
100s-1000s	Large pelagic fish	Some fish
10-100s	Most benthic fish	Most fish
1-10s	Small pelagic fish and many benthic invertebrates	Algal spores, planktonic direct developers
<1	Sessile species and species with highly specialized habitat requirements.	Benthic direct developers

* Note these represent general trends based on emerging data for which there are significant exceptions and data limitations.

A recent trend globally favours the establishment of larger MPAs than in the past (Spalding et al. 2013). The belief that larger areas increase the potential to conserve and improve larger-scale ecological attributes (Toonen et al. 2011) motivates this move towards larger MPAs, greater than the 1-200 km² typically designated in coastal waters (Wood et al. 2008). Larger-sized MPAs encompass more unique habitats and span the home range of multiple species, and therefore have greater potential to protect ecosystems (Sheppard et al. 2012) and, importantly, the links among ecosystems (Toonen et al. 2011). Larger areas thus provide a tool to help maintain intact ecosystem services directly (Snelgrove 1999, Toonen et al. 2013) (Supplemental Figure 1 c, e).

Though smaller than some of the recently established large pelagic MPAs (Supplemental Table 2), the Laurentian Channel AOI falls within the delineation of ‘very large MPAs’ in that it greatly exceeds the median MPA size typical for coastal waters (4.7 km²; Wood et al. 2008). Using data from the IUCN ‘Protected Planet’ database² (IUCN 2015) the Laurentian Channel AOI would rate among the largest 5% of MPAs globally and among the largest which is completely designated no-take (Figure 2 and Supplemental Figure 2). Conservation objectives for MPAs of

² Data subsetting to management areas in the ocean classified as marine reserves, monuments, sanctuaries, parks or protected areas. Refer to Supplemental R-Code for analysis details.

this size typically focus on larger-scale ecological processes as well as protection of individual species or populations. For example, the coastal Al Yasat MPA (~2020 km²) in Qatar was designed to protect species within its boundaries, particularly reef fish (*Porites* and *Acropora sp.*) at various life history stages, through large-scale mitigation of threats (fishing) relative to home ranges, and protection of the variety of habitat types required for these life histories (Moati et al. 2008). At an even larger scale, Papahānaumokuākea Marine National Monument (PMNN; ~360,000 km²) is a component of the larger network of Marine Life Conservation Districts (MLCDs) which focuses on conserving and replenishing natural marine resources around the state of Hawaii (Friedlander et al. 2010, Toonen et al. 2011). Management, through MLCDs in Hawaii, expressly acknowledges the integrated ecosystem services provided by intact ecosystems; PMNN is thus designed to span intact ecosystems and associated ecosystem services such as availability of diverse food fisheries (Toonen et al. 2011, Toonen et al. 2013). Similarly the government of Palau established the Palau National Marine Sanctuary (PNMS ~500,000 km²) which has a mixed management framework. The overall objective of the PNMS is to maintain a healthy ecosystem through the protection of its associated components and functions (Palau 2015). Following the successful designation of large Pacific MPAs, the Government of New Caledonia committed in 2012 to integrating its existing network of medium (>100 km²) MPAs into a larger managed seascape that encompasses the entire territorial waters of the small Pacific nation. When established, the New Caledonia MPA will become the world's largest MPA, applying an integrated management approach to over 1.4 million km² of the Coral Sea (Toonen et al. 2013). Like the PMNN, the New Caledonia reserve seeks to conserve ecosystem functioning of the marine environment. Though several key habitats, such as the Entrecasteaux Atoll (~2,000 km²) and a variety of large lagoons that support threatened dugongs, are afforded special attention, the objectives and monitoring are much broader than those areas, scaling with the size of the MPA.

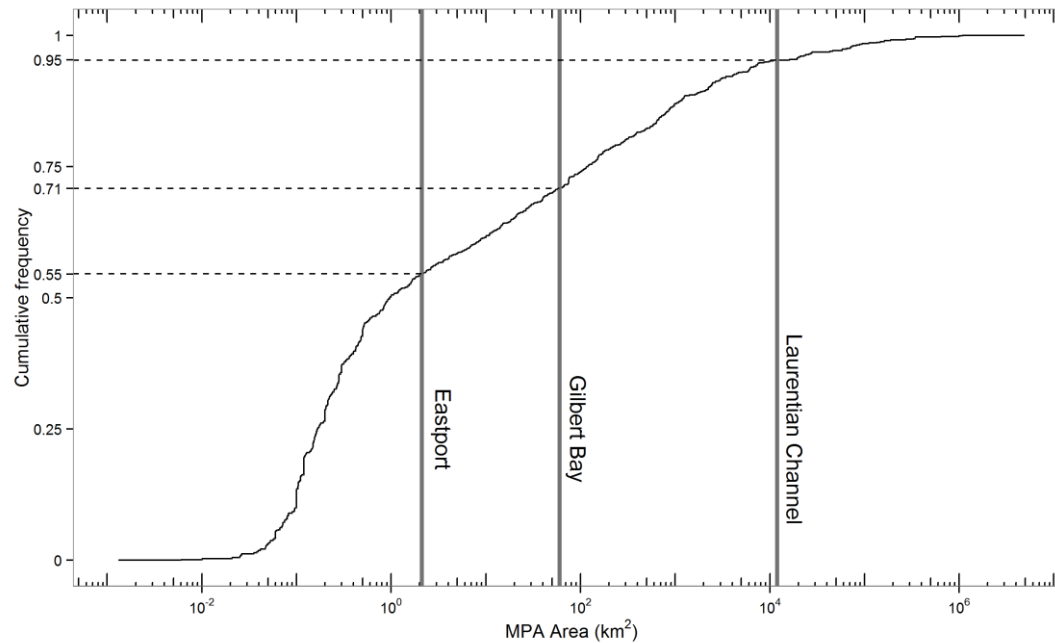


Figure 2. Cumulative frequency distribution of MPA size according to the World Protected Area Database. Vertical lines correspond to the size of established and proposed protected areas in the Newfoundland region. Note the x-axis is presented on a \log_{10} scale.

Often the evaluation of MPAs, relative to their conservation objectives, will require feedback on the biological status of key or focal organisms (e.g. changes in abundance of northern wolffish for the Laurentian Channel AOI). Increasing MPA scale adds challenges for monitoring, especially for species typically monitored with fisheries dependant indices (Pala 2013). As scales increase, key monitoring indices become more diffuse and, in some cases, less involved because the threats posed to species within the MPA decrease with the larger area of protection³. For example, monitoring within the Al Yasat MPA focuses primarily on habitat maintenance and less on direct measures of focal species abundance because it assumes that habitat protection will yield positive biological response for constituent species (Moati et al. 2008). Often, large-scale MPA monitoring focuses on biological attributes, primarily habitat, within the MPA and suggests that management interventions lead to stable communities and generally positive

³ Note this is a general supposition. Protection afforded by MPAs will vary among life cycles and species.

biological responses (Moati et al. 2008, Friedlander et al. 2010, Pelletier et al. 2011; for Al Yasat, PMNN and New Caledonia respectively). Monitoring for large MPAs generally focuses less on comparing MPAs to unprotected areas; empirical data comparing the protected area to outside areas are typically unavailable, largely because of the large scale of the MPA. Choosing reference areas for the proposed Laurentian Channel AOI, for example, will need careful consideration of representivity, given its relatively large size compared to the potential assessment region (reviewed in Sections 1.2.3 and 3.2.1).

Managers often assume that large MPAs that protect key habitats over large scales will yield positive results. Although others question the wisdom of assuming success instead of actually quantifying and testing that assumption (notably Hilborn et al. 2004, Agardy et al. 2011, Pala 2013), advocates for large MPAs ($>1000 \text{ km}^2$) (e.g. Toonen et al. 2013) assert that benefits of synoptic protection outweigh the logistical issues of monitoring or redistribution of fishing effort (e.g. Kelleher & Recchia 1998, Agardy et al. 2003). For wide-ranging species directly or indirectly influenced by marine industry (i.e. tuna fishing), the efficacy of the MPA is predicated on its size relative to the range of the species (Mullowney et al. 2012, Pala 2013).

MPA size and scale of monitoring should reflect MPA objectives, or vice versa, with specific consideration of the life history strategy of the protected species. In a MPA designed to protect a local stock, such as golden cod in Gilbert Bay, the MPA should scale with the neighborhood size of the target organisms, or at a minimum, provide refuge for vulnerable life history stages (i.e., juvenile nursery areas or spawning stocks). Nonetheless, recent studies detail a scale mismatch leading to commercial fishing mortality (see section 2.1.6). The ability to gauge unambiguously the efficacy of an MPA depends upon the intersection between MPA objectives and the spatial scale of biological response (Agardy et al. 2003). For example, the broad-scale objective of the Eastport MPA is to “*maintain a viable population of American lobster in the Eastport Lobster Management Area (EPLMA)*”. Despite the clear conservation objective for Eastport, which follows designs from other successful MPAs (e.g. Mombassa Marine Park, Kenya: McClanahan &

Mangi 2000, Leigh Marine Reserve: Kelly et al. 2002), the small scale of the MPA relative to the ambit of lobster may confound efforts to evaluate whether the objective has been achieved. Particularly for Eastport (see sections 2.2 and 5), the scale of surplus production provided by the MPA (larger and more fecund females), the scale of larval dispersal, and the scale of the effect relative to other sources of variability (i.e., environment, local fishing effort, and oceanographic conditions) all confound the detection of any clear signal of recruitment spillover (Buxton et al. 2014). In a decadal assessment of the Columbretes Island Marine Reserve in Spain, Goni et al. (2010b) demonstrated a clear biomass export from the MPA through emigrating adults of spiny lobster (*Panlinurus elephas*). However, Goni et al. (2010b) were unable to detect any signal of larval recruitment, likely because of large temporal variability in larval survival and settlement, further confounded by extended pelagic duration and dispersal.

Detecting recruitment subsidy (larval movement) is difficult, and in some cases impossible, because differentiating a small proportional change in recruitment relative to the average temporal and spatial expected variation in recruitment (~ 150-200% Pelc et al. 2010). This problem represents one of the largest obstacles in evaluating potential linkages between surplus production within the MPA and increased recruitment outside (Sale et al. 2005), despite numerous theoretical models describing the relationship (Botsford et al. 2001, Botsford et al. 2003, Hastings & Botsford 2003). A sufficiently large differential in production (e.g. Kelly et al. 2000) might produce a detectable signal, but dispersal distance and an increasingly diffuse settlement pattern further compound the problem. Given the high degree of variability expected in recruitment, the power to detect any significant change in recruitment depends on the magnitude of change in mean recruitment. The variability and potentially low statistical power to detect an effect demands extensive temporal and spatial replication, both before and after implementation (Pelc et al. 2010). Indeed, undescribed ocean circulation patterns (Palumbi 2004, Goni et al. 2010a) and diffuse effects relative to natural variability (Pelc et al. 2010), obscure empirical evidence for recruitment spillover.

Though limited in number, there are notable exceptions to the paucity of empirical studies documenting larval spillover. One such example arises from the closure and subsequent recovery of the Georges Bank scallop (*Placopecten*

magellanicus) fishery. Marked increases in adult abundance were documented in zones closed to fishing activity. Following several years of closure, fishing activity resumed with landings demonstrating a close association with spatial predictions of larval spillover from closed areas by recruitment models (Murawski et al. 2000, Lewis et al. 2001). Similarly, rock scallop (*Spondylus calcifer*) and black murex snail (*Hexaplex nigritus*) in the Puerto Peñasco reserve network increased threefold in juvenile density following the establishment of protection, with high spatial overlap outside the reserve (spillover) with predictions generated by a biophysical dispersal model (Cudney-Bueno et al. 2009). Recent work with genetic parentage analyses on the Great Barrier Reef represents one of the only studies to demonstrate direct ‘unambiguous’ links between protected and non-protected areas. Harrison et al. (2012) calculated that no-take areas can produce disproportionate numbers of recruits for a variety of coral reef fish species and demonstrated this influence extended ~30 km from the reserve. Targeted sampling using genetic approaches or biophysical models, combining dispersal residency, behaviour, and circulation patterns, can help narrow the sampling range to avoid wasted effort in areas with low probability of larval recruitment. Collectively, these studies highlight how the extent of larval dispersal will define the range of recruitment subsidy and therefore the design of monitoring.

Considering the difficulty of measuring recruitment spillover, managers should be mindful of gauging the success of the MPA based on a metric that is unlikely to be adequately measured. In cases where measuring larval spillover is difficult, indirect measures of production could be used. For example, comparing the relative production of the MPA to the surrounding areas would provide an indirect indicator of MPA performance contributing towards recruitment spillover, which is measurable (see Section 3.2.5 for more detail).

Detecting spillover by emigrating adults from protected areas to unprotected areas is much more direct and, predictably, many more documented approaches and examples demonstrate spillover. Monitoring spillover may take various forms but like the design (escapement vs. protection) must expressly consider spatial scale and life history. Early work on adult spillover focussed on small MPAs supporting largely artisanal fisheries in the Philippines (Apo Island MPAs 6.9 km²; Russ & Alcala 1996, 2011) and Kenya (Mombasa Marine National Park 10 km²; McClanahan & Kaunda Arara 1996, McClanahan & Mangi 2000), where spillover occurred within 10 km of the defined reef habitat

managed by the MPAs. Documentation of catch per unit effort (CPUE) and diversity provided two monitoring indices to evaluate the MPA, detailing the potential influence of protection and the scale at which detection may be possible. For example, Russ and Alcala (2011) document increased diversity of predatory reef fish with proximity to the Apo Island MPA and a scale of benefit restricted to within 0.5 km of the MPA, scaling with the movements of adult fish in the area. Tag-recapture surveys for spiny lobsters (*Palinurus elephas*) over a decade at the Colombrete Islands Marine Reserve documented increased biomass-weighted CPUE near the MPA boundaries (< 5 km), which contributed an estimated surplus of > 10% to mean annual yields (Goni et al. 2010b). Similarly, biomass-weighted CPUE for rock lobsters (*Jasus edwardsii*) in the New Zealand Leigh MPA suggested emigration of adults into unprotected areas from the reserve, thus providing a positive contribution to the fishery (Kelly et al. 2002). Kay et al. (2012) reported similar benefits for Californian spiny lobster (*Panulirus interruptus*) in the Santa Barbara Channel Islands reserve network, where trap surveys documented a positive reserve effect for CPUE of legal sized lobsters. The ‘effect’ of protection scaled linearly with proximity to the centre of the nearest MPA; ~ 2.43 fewer legal lobsters per unit effort for every km further from the centre of the reserve. The movement range of adult migrants constrained the scale of the measured effect in these examples.

In contrast to studies that document a net-positive spillover of adult fish, the Kenyan Mombassa MPA produced limited increase in catch per unit effort (CPUE) radiating from the MPA. The increase did not compensate for the loss in fishing grounds and thus failed to meet the key objective of producing a net benefit to the fishery, despite higher abundance and density of fish within the MPA (McClanahan & Mangi 2000). These selected examples of adult spillover effects illustrate two primary points: 1) not all MPAs successfully achieve a net positive effect; and 2) evaluation of key objectives requires monitoring at spatial scales mirroring the movement of focal species.

The spatial scale of how the MPA and its objectives match the scale of the proposed effect, the biology of the protected organisms, and the proposed monitoring indicators all merit careful consideration. In general, the size of the

biological response will scale with the magnitude of change evoked by management action, and therefore dictate the ability of management to detect the response. MPA objectives and associated monitoring should explicitly consider the magnitude and spatial scale of the biological response. Large mismatches in the size of the MPA relative to the range of the focal species, or, MPAs that offer little change from the ‘status quo’, will likely result in diffuse change, if any (Agardy et al. 2003). Detecting diffuse benefits will require increased monitoring, both spatially and temporally, because smaller incremental benefits take longer to accrue (see Section 1.1.2) and therefore be detected. Appropriately-scaled MPAs produce benefits which will likely accrue faster and will be more spatially focused. For example, acoustic tagging in the Azores’ Monte da Guia Marine Reserve showed that dusky grouper (*Epinephelus marginatus*) occupied even small protected areas for the majority of the observational period (Afonso et al. 2011), conforming to previous home range estimates (Pastor et al. 2009). Similarly, Cloutier (2010) documented higher densities and measurable spillover from a network of small ($< 1 \text{ km}^2$) rockfish conservation areas along the coast of British Columbia. The relatively restricted home range of dusky grouper and rockfish resulted in clear benefits from protection and associated spillovers within 1-5 km of the protected areas (Pastor et al. 2009, Cloutier 2010).

The spatial scale of a biological response influences the probability of detecting an effect (diffuse vs. direct responses), the extent of monitoring required to measure a response, and how well the MPA matches the relevant spatial extent of the protected species. In this way, consideration of the spatial scale of the biological response should guide the development of efficient monitoring and define the expected outcome of a management intervention. Effective monitoring at the appropriate spatial scale will provide information needed to inform adaptive management, and thus potentially improve the long-term effectiveness of the MPA.

Section 1.1.2 Temporal scale considerations

Over time, with the cessation of fishing and other pressures generally results in changes in biological communities, many of which are positive (e.g. Halpern & Warner 2002, IUCN 2004, Lester et al. 2009). MPAs often

encompass multiple species, each with a distinct, time-sensitive life history pattern, which assessments must consider over multiple temporal scales (Figure 3; Table 3). The positive benefits of a marine reserve may take time to accrue, particularly for long-lived and late-maturing species, relative to faster growing taxa (Halpern & Warner 2003). Studies in temperate and tropical systems demonstrate that the full benefit of a marine reserve might take several years and, in some cases, decades to fully accumulate (Stewart et al. 2009, Vandeperre et al. 2011) even though short-term changes may occur (Halpern & Warner 2002). For example, spillover from larval settlement only becomes detectable once a cohort recruits to the fishery. In Mombasa Marine Park (Kenya) McClanahan and Mangi (2000) estimated that a detectable signal in the population would require a decade of larval spillover. In a meta-analysis of blue cod (*Parapercis colias*) and rock lobster (*Jasus edwardsii*) size and abundance data from New Zealand marine reserves, Pande et al. (2008) detected consistent positive biological responses relative to unprotected areas 6.5 and 8.5 years after protection for each species, respectively. In applying a meta-analytic approach to evaluate the potential influence of MPAs on adjacent fisheries in southern Europe, Vandeperre et al. (2011) found that biological response to protection increased yearly but depended on the interaction between species and MPA size. For marketable fish, the species for which management action invoked the largest change, response was generally larger and quicker than in non-target species. Indeed, it could take up to 30 years to fully realize the complete effect of the management action. Similarly, the abundance of large predatory reef fish only peaked after 20 – 40 years of protection in marine reserves in the Philippines (Russ & Alcala 2010).

Table 3. Review of literature detailing the time to achieve a monitored biological response These examples represent the subset of literature reviewed for this report that provided information about the time to detect a positive effect of protection relative to the MPA establishment.

MPA	Effect Measured	Time Lag	Reference
Puerto Peñasco reserve (Mexico)	Larval dispersal and recruitment (model) of rock scallop (<i>Spondylus calcifer</i>) & black murex snail (<i>Hexaplex nigritus</i>)	2 yrs	(Cudney-Bueno et al. 2009)

Currambene Creek no-take area (Australia)	Abundance of benthic invertebrates and suspension feeders	2 yrs	(Winberg & Davis 2014)
Arrábida Marine Park (Portugal)	Abundance of demersal fish and invertebrates	4 yrs	(Costa et al. 2013)
Skagerrak coast experimental fisheries closures (Norway)	Size and abundance of Atlantic cod (<i>Gadus morhua</i>) & European lobster (<i>Homarus gammarus</i>)	4 yrs	(Moland et al. 2013a)
Bolong de Bamboung (Sénégal)	Trophic structure and abundance of fish communities	3-5 yrs	(Colleter et al. 2012)
Santa Barbara Islands Reserve Network (US)	Abundance of spiny lobster (<i>Panulirus interruptus</i>)	5 yrs	(Kay et al. 2012)
Marine reserves in the Tuscan Archipelago and northwestern Mediterranean sea (Italy)	Fish community (inside vs. outside)	5 yrs	(Micheli et al. 2005)
Meta-analysis of MPAs in New Zealand, Australia, US, and Kenya	Response of fish species previously targeted by fishing	5.1 yrs	(Babcock et al. 2010)
Marine Extractive Reserve of Corumbau (Brazil)	Abundance and biomass of various reef fish	6 yrs	(Francini-Filho & Moura 2008)
New Zealand marine reserves	Abundance of blue cod (<i>Parapercis colias</i>) and rock lobster (<i>Jasus edwardsii</i>)	6.5-8.5 yrs*	(Pande et al. 2008)
Fishing closures (Iceland)	Abundance of demersal fish species	7 yrs	(Jaworski et al. 2006)
Tasmanian marine reserves (Australia)	Abundance decreases in sea urchin (<i>Heliocidaris erythrorammina</i>) and increases in rock lobster (<i>Jasus edwardsii</i>)	7 yrs	(Barrett et al. 2009)
Closed areas (US Virgin Islands)	Size, abundance, & sex-ratio of Nassau grouper (<i>Epinephelus striatus</i>)	9 yrs	(Beets & Friedlander 1999)
Barbados Marine Reserve	Abundance and size of various reef fish species	9 yrs	(Rakitin & Kramer 1996)
Kokomohua Marine Reserve (New Zealand)	Blue cod (<i>Parapercis colias</i>)	9 yrs	(Davidson 2001)
George's Bank fisheries closure (US)	Abundance through time and space	10 yrs	(Murawski et al. 2000, Murawski et al. 2005)

Columbretes Island Marine Reserve (Spain)	CPUE of spiny lobster (<i>Panlinurus elephas</i>)	10 yrs	(Goni et al. 2010b)
Scandola Marine Reserve (Italy)	Various fish species	13 yrs	(Francour 1991)
Meta-analysis of MPAs in New Zealand, Australia, US, and Kenya	Response of fish species not previously targeted by fishing	13.1 yrs	(Babcock et al. 2010)
Columbretes Islands Marine Reserves (Spain)	Reef fish community composition	16 yrs	(Stobart et al. 2009)
Leigh, Tawharanui and Hahei Marine Reserves (New Zealand)	Density of snapper (<i>Pagrus auratus</i>)	7-18 yrs	(Willis et al. 2003a)
Columbretes Islands Marine Reserves (Spain)	Reproductive capacity of spiny lobster (<i>Panlinurus elephas</i>)	20 yrs	(Diaz et al. 2011)
Kåvra MPA (Sweden)	CPUE of European lobster (<i>Homarus gammarus</i>)	20 yrs	(Moland et al. 2013b)
Hopkins Marine Life Refuge and Point Lobos State and Ecological Reserve	Length of fish species associated with giant kelp (<i>Macrocystis pyrifera</i>) habitat	12-20 yrs *	(Paddock & Estes 2000)
Marine reserves near Catalina California (US)	Biomass and egg production of common fish species	10-24 yrs	(Tetreault & Ambrose 2007)
Leigh Marine Reserve (New Zealand)	Benthic community structure	20-25 yrs	(Shears & Babcock 2003)
Mimiwhangata and Tawharanui marine parks	Abundance and biomass of various fish species	29 yrs	(Shears et al. 2006)
Meta-analysis (temperate MPAs)	CPUE (various species)	~30 yrs *	(Vandeperre et al. 2011)
European MPAs meta-analysis (Mediterranean and Canary Islands)	Fish abundance using effect size analysis	5-40 yrs *	(Claudet et al. 2008)
Apo Island MPA (Philippines)	CPUE (various fish species)	20-40 yrs*	(Russ & Alcala 2010)
Dynamic food-web model of the northeast Atlantic	Recovery of fish community size structure	50-100 yrs	(Fung et al. 2013)

*species dependant

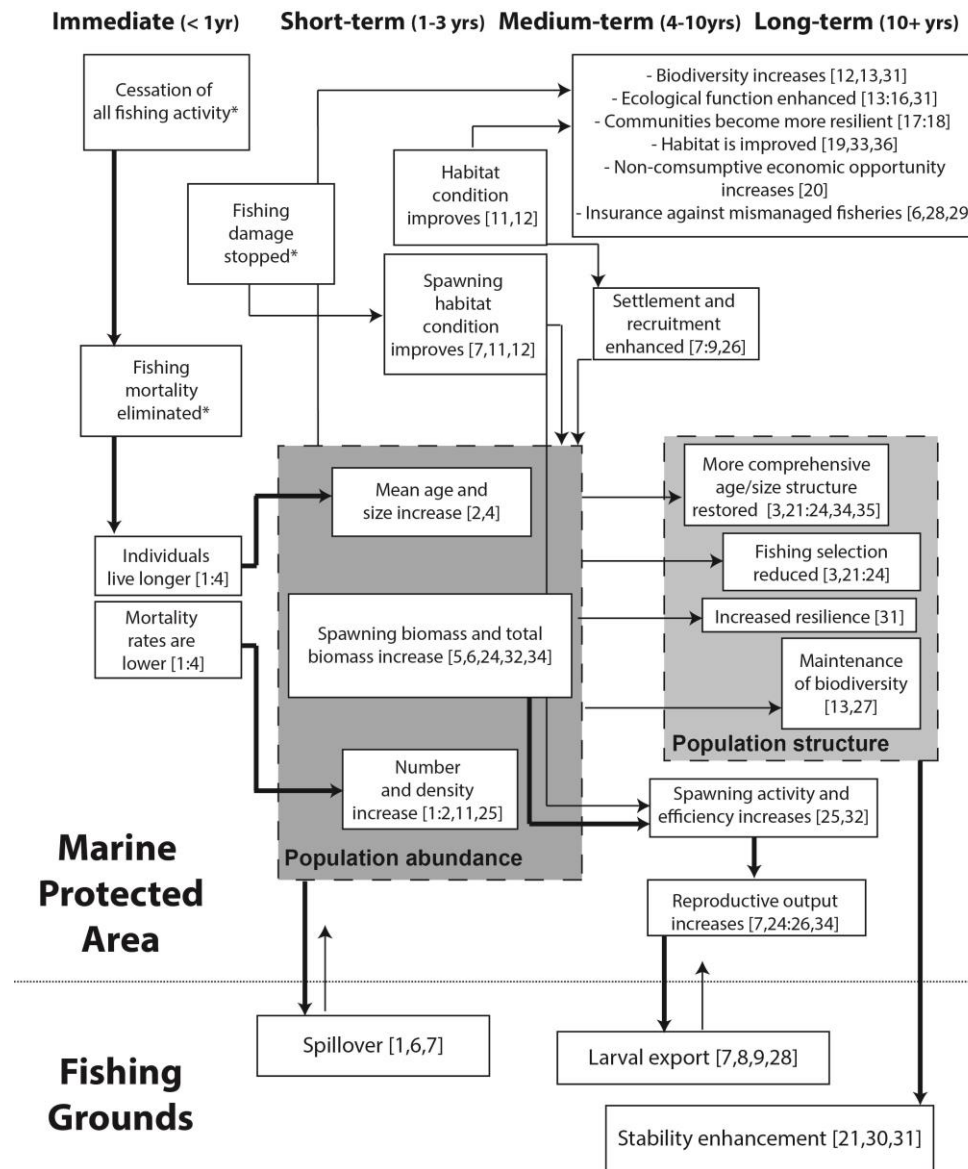


Figure 3: Conceptual map diagramming pathways leading to positive environmental effects, and the relative timescales at which impacts will accrue both within and outside MPA boundaries. The weight of arrows roughly corresponds to the importance of the pathway to the MPA’s capacity to enhance fisheries or provide a biological benefit (adapted from a similar design in Ward and Hegerl (2003). Supplemental Table 3 lists studies that provide empirical data supporting the biological effect.

Prior exploitation rates and density-dependant processes determine the magnitude and rate of biological response (biomass, size, etc.) to protection, which is expected to vary among species (Palumbi 2004). Differences in response time leads to successional stages in community recovery. Evaluations of fisheries or conservation objectives should

therefore incorporate the expectation of time-transient states in community structure and lags in response that depend on exploitation and life history characteristics (Micheli et al. 2004). For example, long-term monitoring of Tasmanian and Australian MPAs showed persistent community-level changes (trophic cascades and feedback loops) even decades after implementation (Edgar et al. 2009). In an empirical analysis of diver census data from southern Australian MPAs, species not targeted by fishers prior to protection and those that did not directly interact with fished species showed no immediate response to protection (Edgar & Stuart-Smith 2009). Similarly, a study of demersal fish and invertebrate species in the Arrábida Marine Park, Portugal, found that only those species targeted by fishing prior to establishment demonstrated significant responses to protection (Costa et al. 2013). Extrapolating this idea further, the timescale for detecting an effect of a management action generally correlates with the magnitude of change evoked by the management action for a given species.

MPA protection and spatial fishing closures generally produce a biological response at some spatial or temporal scale. Large-scale effects, such as community succession or demographic shifts (Green et al. 2014), accumulate over a longer time period (e.g. Russ & Alcala 2010), and are therefore more difficult to evaluate than small-scale effects such as increased densities (Halpern & Warner 2002) or increased recruitment (Cudney-Bueno et al. 2009). Evaluation of MPAs should therefore match sampling effort to the relevant temporal scale of the biological effect (Stewart et al. 2009). For example, if re-establishing community structure is the primary conservation objective, then sampling might be better focused on evaluating fewer spatial replicates over a long period of time than many replicates over a shorter time period. However, adaptive management might require early evaluation and management response. Widespread sampling (e.g. CPUE around the MPA) would therefore prove more effective in detecting small-scale effects that take less time to accumulate.

As in other locations, the success of MPAs in the NL region hinges on sustained monitoring and adaptive management. For current and proposed MPAs in NL, DFO proposes 5- year re-evaluation cycles to adapt monitoring

and MPA policy measures/conservation objectives. The temporal scale at which benefits might occur should be considered within this re-evaluation template. For example, the benefits of MPAs for highly migratory species may be more variable and take longer to fully accumulate (Agardy et al. 2003). In this light, the capacity of the Laurentian Channel AOI to ‘*promote the survival and recovery*’ of leatherback sea turtle (*Dermochelys coriacea*) might only produce a detectable biological signal, and therefore be objectively evaluated, over an extended period of time, especially given the highly migratory behaviour and long lifespan of this species (James et al. 2005a). In contrast, the same objective could be achieved for groundfish species within the MPA over a shorter time period, given their much smaller home range and lifespan (see Section 2 for biological reviews of focal species).

Design and review of monitoring programs must consider the temporal scale over which a biological response could potentially occur. Efficient monitoring programs require optimizing sampling frequency and acknowledging the time required to objectively gauge efficacy. Explicit consideration of the species biology, ecosystem settings, and change in condition evoked by management can help guide the evaluation of temporal scales and thus development of cost-effective and productive monitoring programs.

Section 1.1.3 Location considerations

The location selection of marine protected areas strongly influences the expected outcome and appropriate monitoring. Placement of MPAs in degraded areas, low diversity habitats, or sites with limited historical anthropogenic disturbance can limit the biological response and value of a management intervention. Roberts et al. (2003b) used the analogy, “*a reserve with little biological value will provide few benefits, just as a bank account with little money will yield almost no interest*”, to highlight the importance of prior condition of the location in gauging the expected outcome, and thus the monitoring approach. A review of Tasmanian MPAs confirmed this supposition; distinct change after protection generally occurred only in fish species directly or indirectly (via bycatch) influenced by fishing activity (Edgar & Stuart-Smith 2009). Similarly, the strongest response to protection in the Scandola Marine Reserve near

Corsica, as well as in protected areas along the East African and Californian coasts, occurred in large-bodied fishes either directly or indirectly (bycatch) influenced by prior fishing activity (Francour 1994, McClanahan & Arthur 2001, Tetreault & Ambrose 2007, respectively). The establishment of no-take areas in the Marine Extractive Reserve of Corumbau on poor quality habitat relative to unprotected areas limited spillover effects (Francini-Filho & Moura 2008). The placement over poor quality habitat not only impacted spillover potential, but also hindered the application of a Before-After-Controls-Impact analysis (see Section 3.2.1). Biological response in these examples correlated with the magnitude of change associated with the management action and the relative quality of the protected habitat.

The increased implementation of MPAs as a management strategy globally brings risk of systemic gaps in protection (Devillers et al. 2014). Systemic gaps in protection often follow demand for rapid establishment of MPAs, via national and international commitments, and in seeking compromise with existing marine activities (i.e., commercial fisheries and oil and gas development). Indeed, much of the literature on effective MPA design and planning (e.g. Ban et al. 2011, Delavenne et al. 2012) focuses on striking politically defined balance between existing use and conservation need. While pragmatic as a management and design principle, this objective can potentially misplace protection, in that appeasing stakeholder demands for access may omit species and ecosystems most associated with extractive uses, and presumably those most in need of protection and at risk for further degradation (Devillers et al. 2014). Marine policy that implicitly or explicitly seeks to minimize opportunity loss for current marine stakeholders adds risk of ‘residual’ protection whereby management intervention becomes ancillary to the extractive use of the system (Margules & Pressey 2000, Devillers et al. 2014). For example, major western nations including the United States (US), France, and the United Kingdom (UK), have substantially increased marine protection by establishing large offshore MPAs (see Supplemental Table 2), but placing this protection in areas largely devoid of fishing activity (Watson et al. 2004), while simultaneously leaving relatively heavily exploited coastal ecosystems, and associated biodiversity, at risk.

For the MPAs established and proposed in the Newfoundland region, location characteristics (size, harvesting level, and surrounding habitat) should be considered in the context of conservation objectives, monitoring, and framing of expected response. Areas where the management intervention offers little to change current activity will likely offer smaller and more protracted change relative to those areas where management intervention has significantly changed permitted activities in the area (see section 1.1.2). Protection of relatively ‘pristine’ habitats and/or ecosystems should still be a priority, however, the expected response relative to protecting highly disturbed systems should be considered when implementing monitoring plans.

The Eastport MPA is situated within the fishing grounds of the Eastport Lobster Management Area and LFA 5. The area was selected based on fishermen’s reports of relatively high densities of ovigerous female lobsters (DFO 2014a). Setting aside potential issues with the scale of this MPA and its potential influence to the EPLMA (see sections 1.1.2 & 2.2.5), the management intervention nonetheless directly impacted exploitation of this area. In the years since establishment, several monitoring indicators show positive biological response, including increased abundance and larger size distributions within the MPA (DFO 2014a). The rapid change observed in Eastport, and suggested earlier by Halpern and Warner (2002), was likely driven in large part by the relative impact of change in activity. The location of Eastport, nested within a larger mosaic of similar habitat subject to fishing activity, offered a reasonable comparison location. Although direct observations of the system before implementation was not planned explicitly, long-term trends in the region provide a contextual picture of the system prior to MPA implementation, regardless of the potential to evaluate an effect (see Section 2.2.5). Most importantly, the location and management intervention both fit in principle with the main conservation objective of the MPA.

The Laurentian Channel AOI (LC) provides a striking contrast to the Eastport MPA in location selection. The LC is significantly larger than Eastport and, for the most part, lacks the small size issues (but see Section 2.4.3). The availability of significant background baseline data from trawl surveys for LC (Figure 4), can be used to define

reference points for temporal trends and within before-after-controls-impact analysis (see section 3.2.1). However, for the most part very limited fishing and oil and gas exploration activity characterize the area proposed for the LC MPA (see Supplemental Figures 3-6), relative to surrounding waters. Thus, although the AOI significantly overlaps the distributional areas of the species outlined in the conservation objectives (sections 2.3-2.8), we would expect smaller and more protracted changes in abundance than in other regional MPAs (e.g. Eastport) or in areas where MPA placement significantly reduced anthropogenic mortality. For example, a fisheries closure to bottom trawling in the Gulf of Maine excluded about 22% of the fishing effort targeting the scallop, *Placopecten magellanicus*, in the area; within 10 years (~ 2 generation times for scallop) settlement increased in the area, concomitant with larval spillover (Lewis et al. 2001, Murawski et al. 2005). In a recent synthesis of fisheries-based MPAs Buxton et al. (2014) found little empirical evidence of actual improvement, and those studies that found an effect (Goni et al. 2006, Stobart et al. 2009, Vandeperre et al. 2011) generally focussed on ecosystems under greatest fishing pressure. Overall, the implementation of MPAs, particularly large ones like the Laurentian Channel AOI, is considered long-term management strategies. While some results might manifest over short time frames, the cumulative and multifaceted impact is likely to manifest over a longer, decadal timescales.

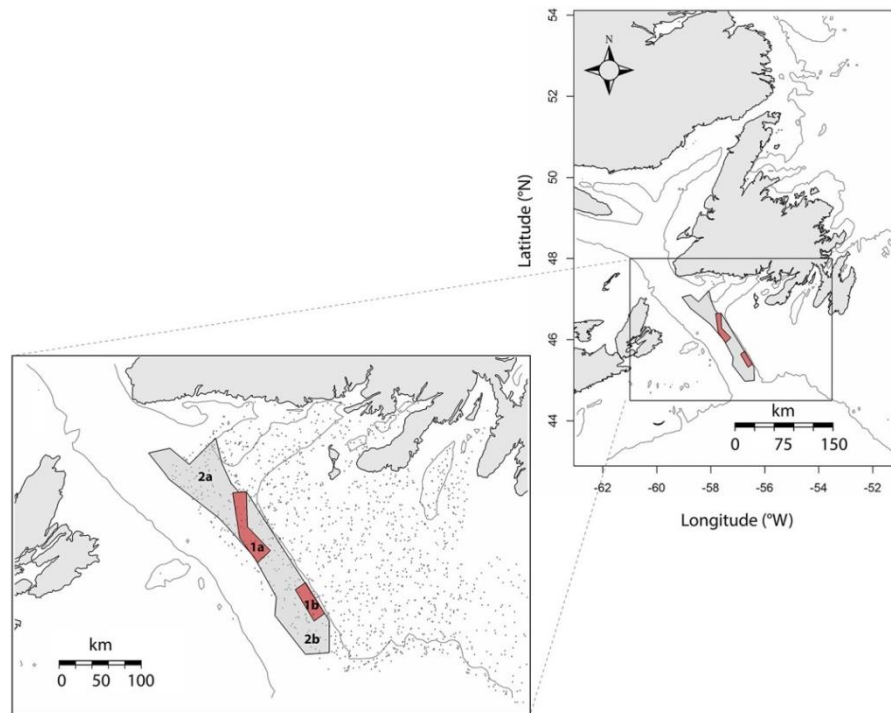


Figure 4. Map depicting the proposed Laurentian channel AOI and the province of Newfoundland and Labrador. Inset shows AOI boundaries and associated groundfish survey sampling locations from 2005-2012 for NAFO divisions 3Ps and 3NO. Solid lines denote the 200-m bathymetric contour. The Laurentian Channel AOI is divided into 2 management levels (1,2) each with two sub-zones (a,b).

Evaluation of MPA efficacy requires both an internal and external process that may include measurements of biological indicators within and outside the MPA, depending on the MPA's objectives. The conservation objective of Eastport, for example, implicitly requires sampling outside the MPA to determine how lobsters in the MPA might contribute to the population in the larger management area. Given the placement of Eastport within a contiguous coastal rocky habitat, the challenge of finding suitable reference sites relates more to scale (see Sections 1.1.1 & 2.2.3) than in finding a suitable comparison. The main conservation objective for Gilbert Bay, in contrast, implies a more internal approach to monitoring not predicated on gauging relative efficacy. Gilbert Bay focusses on a local endemic population of Atlantic cod and the scale of the MPA was designed to encapsulate the majority of movement across all life history

stages (Morris & Green 2002, DFO 2010c, Morris & Green 2014, Morris et al. 2014); the most pragmatic indicators and monitoring needs therefore reside at the scale of the population within the MPA.

In terms of broad objectives, the LC AOI falls between Eastport and Gilbert Bay in that it seeks to protect particular species from “human harm” (see Sections 2.3-2.8). To address this objective requires some degree of external validation to determine whether the MPA is achieving this objective relative to environments with no management intervention. The LC AIO broadly encompasses three coarse habitat types: benthic plains of sediment (400-450 m) and rock outcrops (40-400 m) (Figure 4 & Supplemental Figure 7), and the pelagic waters. These broad habitat types differ in their biological communities, environmental histories, exploitation histories, and interaction with external (outside the MPA) stressors, leading to relative differences in their response to protection (Miller & Russ 2014). MPA monitoring and selection of reference sites must therefore differentiate between the effects of habitat and management intervention when evaluating efficacy. In an analysis of catch data surrounding the Santa Barbara Channel Islands reserve network, Kay et al. (2012) reported a significant effect of habitat on whether protection increased California spiny lobster (*Panulirus interruptus*) abundance, highlighting the importance of incorporating habitat type when evaluating the effect of protection. In a recent global assessment of MPA monitoring, Miller and Russ (2014) note that despite an implicit assumption that contrasting habitat types produce equally contrasting responses, less than 30% of studies reviewed explicitly tested the influence of variable habitat on response.

Availability of systems comparable to the LC AOI also influences the application of external measures of efficacy, in terms of similarity in habitat and exploitation history. In order to objectively gauge how a management intervention contributes to conservation objectives (IUCN 2004, Pomeroy et al. 2005, Fox et al. 2014), designs should incorporate similar reference and MPA sites as much as possible. Historical fishing levels around the LC AOI, *high outside – low within*, (Supplemental Figures 3-6) could hinder selection of suitable non-protected reference locations. Identifying a reference site with a similar history of fishing activity might require moving beyond the immediate

boundaries of the AOI. Although DFO surveys extend north and east to shelf break habitats in NAFO divisions 2J3KLNO, increasing distance from the AOI adds dissimilarity in environmental conditions, thereby further complicating the application of external reference sites. Furthermore, the AOI encompasses much of the deep strata (bottom depth > 400 m) surveyed in the region (Figure 4 & Supplemental Figure 7), thus limiting candidate reference sites (Miller & Russ 2014).

The specific location of the MPA strongly influences the potential biological response and the ability to monitor that effect objectively. In particular, monitoring programs in review or in development should consider the location of the MPA when evaluating the sampling effort required to detect a response given the location (i.e., whether or not the management intervention improved conditions for survival) and how to reasonably compare this biological response to non-protected areas (i.e., reference sites).

Section 1.1.4 Framing objectives

MPA monitoring provides a framework in which to evaluate MPA efficacy. MPAs can potentially help protect multiple species with diverse life histories which, in turn, may exhibit divergent responses to protection. In this sense, MPAs can potentially achieve a broad range of objectives for stakeholders with different perspectives and priorities for protection. MPA planners and advocates must work together to define targeted objectives for MPA networks and individual MPAs. These objectives, first and foremost, must be clearly articulated, achievable, and contextualized. Unrealistic targets or expectations add unnecessary pressure on MPA managers, threaten the continued existence of the MPAs, and even influence future designations (Agardy et al. 2003, Pomeroy et al. 2005); Canada must consider these concerns as it works to implement its MPA strategy (Canada 2011).

Objectives and associated monitoring should consider whether the MPA and/or management action itself will support the conservation objectives or whether it is predicated upon the viability of adjacent unprotected areas in the context of the conservation objectives. For example, the primary conservation priority of the Eastport lobster MPA is to

maintain a *viable population of American lobsters* within the Eastport Lobster Management Area (EPLMA). This objective is clear but lacks a broader context. If populations of lobsters in the EPLMA decline, as seen over the past decade, how does this decline reflect the efficacy of the MPA, especially considering its small size (2.1 km²) versus the broader management area in which it is assessed (EPLMA of ~400 km²) (DFO 2014a)? In reality, the viability of the lobster population in the EPLMA depends more on fishing pressure as a function of population size, local fisher's conservation efforts (v-notching and minimum legal size), and environmental conditions (success of larval recruitment). As a management intervention, the MPA is unlikely to maintain a viable population of American lobsters in isolation; however monitoring indicators suggest some adult migration, increased abundance, and increased size within the MPA, all of which contribute to EPLMA productivity. Nonetheless, the likelihood of separating the influence of the MPA against the backdrop of other sources of population variability remains doubtful.

The Laurentian Channel AOI conservation and research objectives prioritize its potential as a refuge for long-lived and highly migratory species such as leatherback turtles (*Dermochelys coriacea*). Efforts to monitor the survival and recovery of leatherbacks must recognize that individual turtles spend a significant portion of their life history outside the protection of the MPA. Any declines in the leatherback population over time cannot be fully attributed to a failure of the MPA to meet its objectives, particularly if other impacts on the species are not considered. Similarly, simulations of North Sea population responses to various levels of spatial closures to Atlantic cod (*Gadus morhua*) fishing suggested a 25% reduction in fishing pressure would produce almost negligible changes in spawning stock biomass (Horwood 2000). Because Atlantic cod, like leatherback turtles, are wide ranging, extrapolating the influence of protection beyond the boundaries of any closed area would be extremely difficult (Kaiser 2005).

The primary objective of the Gilbert Bay MPA is to protect the local population of Atlantic cod and its habitat. The Oceans Act officially delineated the MPA boundary early in the MPA process. The MPA design protects key stages of the population's life history, and encompasses about 90% of the populations' entire range and thus habitat. Interim

protection measures prior to MPA designation clearly protected the population from directed fishing pressure at spawning and overwintering areas (Morris & Green 2010). After MPA designation, acoustic tagging revealed that some fish moved beyond the MPA boundary where they were harvested in the local commercial fishery (Morris et al. 2014). The scientific monitoring program of the Gilbert Bay MPA captured population dynamics and described the system well over several generations of cod, however, a relatively small mismatch in coverage has led to fishing mortality large enough to influence the Gilbert Bay population. Relatively small and targeted changes to the coverage of the MPA could bridge the gap in protection and potentially aid in future rebuilding of the stock, as could temporal fishing closures in adjacent unprotected areas (Morris & Green 2014). A shift in focus of the conservation objectives to consider rebuilding as a component of maintenance would provide a mechanism to guide adaptive management while maintaining current monitoring frameworks.

The development of suitable monitoring indicators to inform MPA performance in meeting the conservation objectives will depend in part on the wording and intention of the conservation objective. For example, the Gilbert Bay and Eastport MPAs conservation objectives (Section 2.1.6 and 2.2.6 respectively) focus on specific populations (Gilbert Bay cod and lobster) and, in the latter case, Management Areas (Eastport Lobster Management Area). These objectives are precise in that they identify specific biological targets and unambiguously guide the monitoring program. The proposed COs for the Laurentian Channel AOI are more general. For example, corals, smooth skate, black dogfish, and porbeagle shark (Sections 2.3, 2.6, 2.7 and 2.8, respectively) are all to be “protected from human induced mortality in the Laurentian Channel MPA”. Does this CO intend to convey that elimination of human-related mortality within the MPA is sufficient in itself, irrespective of whether the MPA contributes to population status either inside the MPA or more broadly in the region? Additionally, the MPA should ‘promote the survival and recovery’ of northern wolffish and leatherback sea turtles (Sections 2.4 and 2.5, respectively). While the objectives of the Laurentian Channel AOI are specific in their target species, approaches to monitoring could be more general in nature. For example, would monitoring of human activity (i.e. fishing exclusions and boat traffic) within the MPA suffice to gauge MPA efficacy in

the context of protection from human induced mortality, or, would monitoring require an in-depth census of population fluctuations of focal species at some temporal scale? Similarly, can the success of ‘promoting survival and recovery’ of wolffish and sea turtles be gauged by simply measuring how the MPA creates conditions that favour survival (i.e. reduced industrial activity and reduced destruction of critical habitat), or, would monitoring require an explicit measuring of mortality and population demographics through time? While the generality of the conservation objectives for the Laurentian Channel MPA provide flexibility for monitoring, the precise wording of these objectives should be carefully considered prior to formal MPA designation to convey the desired outcome of the management intervention. Creating precise vs general conservation objectives will significantly influence the flexibility or ambiguity of how monitoring can inform success.

Indicators that do not scale with the biological potential of the management action, or, are too general to unambiguously guide monitoring can lead to inefficient use of human and financial resources. Recognizing the conservation objectives within the limitations of the MPA (e.g. Eastport ~0.5 % of assessment area) will lead to more realistic (and ultimately) successful monitoring indicators (Agardy et al. 2003).

Section 1.2 Robust monitoring

The current legislation of two MPAs, and movement towards a third in the Newfoundland region, offers an opportunity to develop a robust and integrated monitoring framework. Monitoring and evaluation indicators considered in isolation limit the potential for successful MPA networks (Roberts et al. 2003b, Edgar et al. 2014). Although specific conservation objectives have been identified for the MPAs in the NL region, the implementation of broadly applicable indicators can strengthen Canada’s objective of a National Network of MPAs (Canada 2005) and help to facilitate their use in larger-scale questions regarding MPA efficacy in Canada.

In critically reviewing monitoring needs and strategies for Gilbert Bay (GB), Eastport (EP), and the Laurentian Channel AOI (LC), we focus on key features that define the conservation objectives of each MPA and review whether

current or proposed monitoring indicators are likely to succeed. Through an extensive review of current literature, and examination of comparable MPAs, we highlight contemporary approaches to monitoring and highlight areas of opportunity to improve existing and proposed monitoring frameworks.

Our review focuses on providing a broad template to identify requisite focal areas to consider when developing monitoring plans and gauging the expected response. In particular we consider species biology, relevant spatial-temporal scales, MPA location, and the intention of the conservation objectives (reviewed in Sections 2, 1.1.1 1.1.2, 1.1.3 and 1.1.4, respectively).

Section 2: Species Biology and conservation objectives (*Objective 1*)

Numerous peer-reviewed meta-analyses examine the potential impact, and expected response, of marine biota to protection provided by MPAs (e.g. Rowley 1994, Cote et al. 2001, Halpern & Warner 2002, Gell & Roberts 2003, Pande et al. 2008, Lester et al. 2009, Davis et al. 2012). Commonly these analyses ascertain generalities about the expected biological response of a given species or community to protection, using meta-analyses to bridge differences among data sources using statistical methods such as effect sizes, which outline the magnitude of a treatment effect. These studies attempt to predict biotic response to specific management interventions, but unfortunately must employ disparate data comprised of different survey types (e.g. CPUE vs. visual surveys), biological systems (e.g. temperate vs. tropical), and species and life history strategies (e.g. migratory fish vs. resident invertebrates). Large-scale reviews risk including studies and data lacking the same critical rigour expected of the analysis itself (Stewart 2010), potentially masking any issues associated with data sources in the larger meta-analysis. For example, an early review of marine protected area effects by Halpern and Warner (2002) suggests that impacts of marine reserves can be rapid and long-lasting, however, longer-term studies and more recent meta-analyses suggest much more varied responses, temporally,

and among species and location (e.g. Willis et al. 2003a, Russ et al. 2005, Guidetti & Sala 2007, Goni et al. 2010b, Russ & Alcala 2010).

Conservation objectives and monitoring frameworks were developed for the current and proposed MPAs in the Newfoundland region. Conservation objectives guide monitoring and ultimately provide managers with tools to gauge MPA efficacy. Predicting the response of a given species to a management action is difficult and we typically lack sufficient knowledge about marine ecosystems to predict reliably specific biological response to a reduction in fishing or other marine activity (Christie & White 2007). However, consideration of the biology of focal species, including variability-magnitude of species range, life history durations, and potential response to protection (discussed in sections 1.1.1, 1.1.2, and 1.1.3 respectively), can focus expectation, inform adaptive management, and ultimately guide the development of realistic objectives.

Section 2.1 Atlantic cod (*Gadus morhua*) – Gilbert Bay MPA

Section 2.1.1 Biology

Atlantic cod (*Gadus morhua*) are among the best-studied groundfish species in the world (Kurlansky 1998, FAO 2012). They have been prized commercially for many centuries, and can reach extremely high levels of abundance. Individuals generally live less than 3 decades and can grow to 2 meters in length. Large individuals can broadcast spawn millions of eggs in one year, typically in spring and summer, and are reproductive for many years. Atlantic cod prefer to spawn at temperatures between 4 and 7 °C (ICES 2005, Righton et al. 2010), but developing eggs tolerate a temperature range from -1.5 to 12 °C (Pepin & Helbig 1997, Jordaan & Kling 2003, ICES 2005, Geffen et al. 2006). During the first year of life, eggs develop in the water column and larvae remain pelagic until they reach 20-40 mm in length, after which juveniles settle to a benthic habitat, usually in summer and fall. Atlantic cod typically reach sexual maturity at sizes greater than 35 cm and 4-6 years of age, however, growth and maturity vary greatly among populations and over time as ecosystems change. As cod reach maturity, many undergo annual feeding migrations that range greatly in

duration and distance (from 10s to 1000s of km) among both individuals and populations (see Robichaud & Rose 2004 and references therein).

Gilbert Bay cod are adapted to local environmental conditions (Morris et al. 2014); the sub-zero temperatures they experience for 6 months of the year affect growth rate, timing of spawning, and migration. Gilbert Bay cod grow at slower rates than individuals from other cod populations (Morris & Green 2002). Spawning occurs in May and June after land-fast ice clears from Gilbert Bay headwaters where spawning occurs. Oceanographic conditions retain eggs and larvae near these inner areas of the bay. Gilbert Bay cod usually reach sexual maturity between 32 and 40 cm, and 5-7 years of age. Although Gilbert Bay cod are considered resident with a home range limited to a few hundred square kilometers, the population migrates seasonally from the inner portions of the bay to coastal feeding areas ~40 km away (Morris et al. 2014).

Section 2.1.2 Distribution and habitat

Atlantic cod are population rich and widely distributed through northern temperate Atlantic waters. Atlantic cod eggs and early larval dispersal depend on hydrographic conditions (Pepin & Helbig 1997, Ciannelli et al. 2010). Generally, the dispersal of eggs and larvae vary with population range (Sinclair 1988). Throughout the range of Atlantic cod, geographic features such as continental shelves, continental slopes, specific coastal areas and bays, and semi-enclosed fjords help define heterotypic groups of fish (i.e. populations, sub-populations, and races; Robichaud & Rose 2004, Skjaeraasen et al. 2011), that often differ genetically (Knutsen et al. 2007, Bradbury et al. 2011) and behaviourally (Salvanes et al. 2004, Morris et al. 2014). Some populations migrate greater distances than others.

Gilbert Bay is fjord-like, characterized by a long narrow inlet with shallow sills and an island archipelago at the headlands that restrict connectivity to the adjacent Labrador Sea. Local physical features and hydrographic conditions, such as reduced salinity at the surface, restrict dispersal and define connectivity. Older Gilbert Bay cod move primarily within the Gilbert Bay MPA boundaries; however some individuals move a short distance outside the MPA during

summer to forage on capelin and other fishes. From September through June, most Gilbert Bay cod remain within the MPA area.

Section 2.1.3 Spatial scales of movement

The spatial scale of the Gilbert Bay MPA (60 km²) was determined primarily by the expected spatial scale of the local cod population. MPA boundaries were established based on information from public consultations, local knowledge, and scientific research gathered between 1998 and 2004 prior to designation in 2005 (Green et al. 2004). The MPA also includes a zoning plan with regulations that differ among zones; in contrast to strict protection in some zones, portions of the MPA allow limited fishing should population abundance increase. The inner portions of the MPA that prohibit fishing successfully protect important overwintering and spawning areas. Scientific research and commercial fishing information shows, however, that some Gilbert Bay cod move a short distance beyond the established MPA boundaries in summer. Commercial fishing in those areas in summer has likely caused a decline in the abundance of large-sized Gilbert Bay cod (Morris et al. 2014); the spatial scale of the MPA therefore does not protect all individuals during summer. The persistent low abundance of other northern cod that would otherwise overlap migratory Gilbert Bay cod distribution in summer increases the relative and unintended effect of fishing pressure on Gilbert Bay cod. Increased abundance of northern cod could alleviate pressure on Gilbert Bay cod under current management measures, however, any changes to northern cod management as the stock grows, such as fishing season and quota, could affect the Gilbert Bay cod population.

Section 2.1.4 Threats

Commercial fisheries have targeted Atlantic cod for centuries, and represent the primary current pressure on populations. Several Atlantic cod populations in Canadian waters, most notably northern cod, collapsed during the early 1990s and only recently began to show signs of recovery. Researchers attribute the collapse to overfishing, and to a lesser extent, unfavourable environmental conditions.

The Gilbert Bay population of Atlantic cod sits within the northern cod stock management area. Unlike other northern cod, the Gilbert Bay population likely rebuilt during the northern cod moratorium (Morris et al. 2014). When a northern cod commercial index fishery reopened in 1998 after a six-year fishing moratorium, Gilbert Bay cod were relatively abundant compared to other northern cod. Localized fishing at that time threatened the “golden cod” subpopulation, leading to successful efforts to protect the population from direct overfishing on spawning areas and elsewhere where fish aggregated in early fall.

Importantly, the MPA continues to protect important areas for spawning, overwintering, and aggregation (Janes et al. 2009, Morris & Green 2010). A limited fishery currently targets northern cod, some 20 years after the original northern cod moratorium, however, northern cod abundance remains low. Some of the effort in this fishery targets the boundaries of the Gilbert Bay MPA, and subsequently removes some Gilbert Bay cod that migrate outside the MPA boundaries during summer (Morris & Green 2014). Commercial fishing selects large Gilbert Bay cod, which represents an important component of the population. Large Gilbert Bay cod produce more eggs than smaller fish and contribute valuable reproductive productivity to the population. They also acquire experience on previous spawning, migration, and feeding areas, likely through learned behaviours (Morris et al. 2014). In addition, large Gilbert Bay cod likely play an important community-level role for the population, limiting abundances of other species and perhaps preventing them from colonizing important core habitats within the populations range (Morris and Green, *unpublished data*). Continued fishing threatens abundances of large migratory Gilbert Bay cod that stray outside MPA boundaries, and could affect the population’s ability to rebuild to historical levels.

Section 2.1.5 Conservation status

Atlantic cod abundance in the Western Atlantic remains low, and several populations are considered threatened (COSEWIC 2010); a limited commercial fishery for cod nonetheless continues.

Following decreases in Gilbert Bay cod in recent years, current abundances are much lower than since MPA monitoring began. This population is recognized as genetically and behaviourally distinct from other northern cod populations; the Committee on the Status of Endangered Wildlife in Canada (COSEWIC) has not yet assessed this population, in part because of the existing MPA status with specific management restrictions and detailed monitoring. The Gilbert Bay cod population contributes to overall diversity of Atlantic cod both genetically and behaviourally, and its abundance has declined in recent years and raises concerns regarding its conservation status.

Section 2.1.6 Conservation objective considerations

The conservation objective for the Gilbert Bay MPA focusses expressly on “*the conservation and protection of the Gilbert Bay cod and its habits*”. The 60 km² MPA protects the majority of the life history of the local cod population. Local oceanographic features and the timing of spawning constrain dispersal and increase local retention. The MPA encompasses critical juvenile nursery habitat for the population.

Monitoring data for Gilbert Bay are suitable for evaluating the performance of the MPA in protecting the golden cod population (Morris & Green 2010, Morris & Green 2014). The summer migration of large individuals to areas outside the MPA, where existing commercial fishery regulations do not protect Gilbert Bay cod, limits the full potential of the MPA to ‘conserve and protect’ Gilbert Bay cod. The consequences of this uncontrolled, and largely unknown, mortality on population viability and persistence remain unknown, however, the extirpation of other semi-discrete Atlantic cod spawning components results in little likelihood of rapid re-colonization (Robichaud & Rose 2001, Skjaeraasen et al. 2011, Dean et al. 2014). To fully achieve the conservation objective through MPA management regulations, the Gilbert Bay MPA would need an amendment to its boundaries in order to protect Gilbert Bay cod that mix with other northern cod stocks outside the existing MPA boundary. MPA stakeholders do not currently support changes to the MPA boundary, however, other management measures could enable suitable restrictions outside the MPA to protect the summer migration of Gilbert Bay cod.

Ample evidence shows that ignoring fine-scale population structure risks serial depletion of unique sub-components, thereby lowering stock productivity (Frank & Brickman 2000, Smedbol & Stephenson 2001) and potentially permanent losses of local adaptation that could affect species persistence and resilience or lead to local extinction (Schindler et al. 2010, Dean et al. 2014).

Section 2.2 American lobster (*Homarus americanus*) – Eastport MPA

Section 2.2.1 Biology

American lobster (*Homarus americanus*) is a large decapod crustacean species with a predominantly benthic life history; benthic juvenile and adult stages follow a pelagic larval phase. Lobsters develop and grow through moult cycles, the frequency of which decreases with age (Lawton & Lavalli 1995). Size at sexual maturity and recruitment to the adult phase vary across the species' range (Campbell & Robinson 1983) depending on summer water temperature (Aiken & Waddy 1986). In the Newfoundland region, maturity occurs at approximately 82 mm carapace length (DFO 2013a). Age has been difficult to determine in lobsters until recently (Kilada et al. 2012), however, male and female are presumed to become sexually mature between 8-12 years old in Newfoundland.

Mating in American lobster coincides with moulting during the summer period (June-September). Females extrude eggs, which are fertilized externally and brooded on the ventral surface of the abdomen for 9-16 months (Campbell & Bratley 1986) before spawning. Fisheries management prohibits retaining ovigerous female lobster or previously ovigerous female as indicated by a v-notch (a voluntary v-notching program is active in the Eastport area) (DFO 2013a). These restrictions, while aimed at maximizing egg production, can lead to biased sex ratios and larger female lobsters through time. Decreased probabilities of males surviving and growing to a sufficient size to mate can affect the probability of successful copulation for large females and potentially lead to sperm limitation (Fogarty & Gendron 2004).

Although larval survival has been shown to be temperature and density dependent, survival during the larval phase remains largely undocumented (Ennis 1995). Low temperatures sometimes observed in the Eastport area (~10 °C) have been linked to increased mortality (~ 75%) in late larval phases. Limited data suggest generally low and spatially and temporally dynamic larval survival (Ennis 1995). Available habitat mediates survival during the juvenile and post-larval phases, with significantly higher survival in rocky cobble habitat compared to less complex sandy habitat. Overall survival through this period is expected to range between 0-27% (Lawton & Lavalli 1995). The predominantly rocky bottom assumed for the Eastport area suggests low risk of habitat limitation.

Natural mortality of adult lobsters is low and estimated at 3-8% (Lawton & Lavalli 1995). An extensive fishery across its range, and in the Eastport MPA area, targets adult lobsters. Harvesting of legal lobsters may reach ~90% of legal-sized individuals in some locations (Fogarty 1995). Size-frequency distributions from Newfoundland and the Eastport area (Lobster fishing area -LFA 5) indicate capture of the majority of exploitable biomass within the first year of recruitment to the fishery (~ 82 mm carapace length; DFO 2013a).

Section 2.2.2 Distribution and habitat

American lobster span the eastern coast of North America from Cape Hatteras, North Carolina to the Strait of Belle Isle between Newfoundland and Labrador. Lobsters occur along the entire northern coast of Newfoundland, in designated lobster fishing areas 3-7; the fishery near the Eastport MPA is particularly active (Ennis 2011).

Larval lobsters occur in near-surface waters (upper mixed layer; Ennis 1975), from June – September with emergence/spawning when bottom temperatures reach 12 °C (Aiken & Waddy 1986). Juvenile lobsters (~0-7 years age, >40 mm carapace length) occupy rocky cobble habitats in the nearshore, generally less than 10 m depth. Juveniles are considered shelter limited and strongly prefer complex habitat (Wahle & Incze 1997). Adult lobsters occupy a variety of soft bottom and rocky habitat but prefer complex biogenic habitat associated with algal cover (Factor 1995). Adults generally occupy nearshore environments (<40 m) in the summer months and migrate to deeper waters (<200 m) during

the winter (Campbell 1986). In Newfoundland, the majority of adult lobster occur in water depths less than 30 m depth (Ennis 1984).

Section 2.2.3 Spatial scales of movement

The pelagic larval duration of American lobsters varies with temperature and can span 2-8 weeks (Mackenzie 1988). Passive advection (e.g. Xue et al. 2008) and behaviour (Katz et al. 1994, Stanley et al. in prep-b) mediate dispersal during the larval period. Estimates of larval dispersal distances depend on location of release, but generally scale between 10-100 km (Incze et al. 2010, Stanley et al. in prep-a). After settlement, juveniles become cryptic and generally move less than 1 km until they become large enough to defend against predators and conspecifics (Factor 1995, Wahle & Incze 1997). Adult lobsters can exhibit resident behaviour with little net movement, home ranges less than 10 km, and migratory movements; generally, yearly movement ranges between 10 and 50 km (Campbell 1986, Bowlby et al. 2007).

No operational oceanographic or biophysical model exists to predict the dispersal range of larvae from Eastport; however, expected dispersal could be significant (see section 2.2.6 below) given extended pelagic larval duration in relatively cold temperatures (~10-14 °C). Tagging near the Eastport MPA suggests that the majority (~77%) of adult lobsters moved less than 1 km from their release point in a given year (Rowe 2001).

Section 2.2.4 Threats

Overexploitation poses the greatest current threat to American lobster. Management actions in the region prioritize maintaining egg production through a voluntary v-notch program (~11 % of catch) (DFO 2014a), minimum size requirements, and gear-season restrictions. Though not well documented in the Newfoundland region, potential future threats could include overlap with invasive green crab (Rosson et al. 2006) and development of shell disease (Smolowitz et al. 2005).

Section 2.2.5 Conservation status

American lobster abundance has increased across its distributional range to the highest levels on record (Steneck & Wahle 2013). However, this increase is not uniform and declines continue in some areas. In LFA 5, in which the Eastport MPA resides, recent declines have approached 80 % relative to the early 1990s (DFO 2013a). No fisheries-independent data source for lobster abundance exists for Eastport, however, the ongoing Eastport Lobster Management Area (EPLMA) monitoring program includes adult tagging, research sampling (standard and modified traps) and commercial at-sea sampling (DFO 2014a, b). All of the monitoring data mirror trends from landings data.

SARA: No schedule, No status COSEWIC: Not monitored

Section 2.2.6 Conservation objective considerations

The conservation objective for Eastport is to “*Maintain a viable population of American lobsters through the conservation, protection, and sustainable use of the resources and habitats*”. The Eastport MPA encompasses ~2.1 km², and spans two non-contiguous locations. Monitoring suggests that adults migrate from the MPA to exploitable areas and lobsters within the MPA span a broader size distribution for both males and females (DFO 2014a). A key element of the Eastport conservation objective is potential spillover of individuals from the MPA to the large management Eastport Lobster Management Area (EPLMA). As in Eastport, MPAs in Spain (Columbretes Islands Marine Reserve (CIMR) 45 km²; Goni et al. 2010b) and New Zealand (Leigh Marine Reserve 24 km²; Kelly et al. 2002), support greater abundances of lobsters (*Paninurus elephas* and *Jasus edwardsii*, respectively) than surrounding areas. These areas also produce larger lobsters overall which, through emigration, benefit the adjacent fishery through increased biomass CPUE.

Although benefits of the Eastport MPA parallel those in these other MPAs, the scale of the MPAs merits comment. Both CIMR and Leigh are over 1000% bigger than Eastport, and thus support significantly larger adult biomass. Researchers also noted a highly localized benefit to the fishery in both these systems. These two points suggest a relatively modest expected contribution of the Eastport MPA to the surrounding management area, and a minimal biomass contribution to the surrounding fishery. The probability of even detecting such a small effect is extremely low given the small number

of lobsters migrating and the short migration distances involved (potentially less than 1 km per year, (Rowe 2001). Fishers will presumably capture lobster in the immediate area, as observed in CIMR and Leigh.

Given the low potential for a measurable effect through adult migration, larval spillover offers the most likely mechanism for the MPA to contribute to the EPLMA lobster population. The higher average density and larger size of lobsters within the MPA compared to surrounding areas (DFO 2014a) presumably enhances reproductive potential for the area. Past studies of recruitment relationships in American lobster focused on stock production and settlement indices (Wahle et al. 2004). Dispersal models are typically validated over large scales (i.e. coast of Maine and Bay of Fundy; Incze et al. 2010). The Eastport pelagic environment in which larval lobster are released are among the coldest in the species range (Aiken & Waddy 1986), resulting in an extended pelagic duration compared to more southern populations (Mackenzie 1988). At a surface temperature of 14 °C (a conservative estimate of average pelagic surface temperature), the time from spawning to settlement in Eastport would be ~ 50 days (Mackenzie 1988). Extended pelagic duration likely extends dispersal distances and exposes larvae to pelagic environments over a longer time period and presumably increases mortality. As larvae disperse further and mortality rates become higher and more variable, any signal from surplus larval production from the MPA becomes more diffuse and harder to detect. Dispersal longer than 20 km in most directions, especially from the Duck Islands component of the MPA, would transport larvae well outside the EPLMA. The extended period over which a larva eventually recruits to the fishery, estimated at 8-12 years, further confounds this diffuse signal.

Genetic work across the range of American lobsters suggests some genetic structure. In particular, recent analysis suggests relatively small-scale (between bays) spatial structure in the Newfoundland region (Benestan et al. 2015). Given relatively low presumed movement in adult lobsters, larval connectivity likely drives genetic pattern (Kenchington et al. 2009). Regional differences in swimming capacity of larval lobsters, in response to environment,

could augment dispersal trajectories and potentially contribute to genetic pattern (Stanley et al. in prep-a;b). This developing work on genetic pattern and larval dispersal suggests a larval dispersal range of 10 – 100 km in the region.

Although ongoing monitoring provides information about MPA performance, the spatial scale of American lobster life history stages suggests the scale of the MPA relative to all other sources of variability will limit efforts to monitor and objectively evaluate the current conservation goal. If the Eastport MPA success hinges upon a measurable biological effect at the scale of the EPLMA fishery and landings, Eastport might pose an illusion of benefit where no measurable influence occurs (Agardy et al. 2011).

Section 2.3 Corals (various species) – Laurentian Channel AOI

For the purpose of this report, we focus primarily on a specific group of corals, the sea pens, given that they form the most abundant and diverse coral group within the Laurentian Channel AOI (DFO 2010a, Kenchington et al. 2010, DFO 2012a, Lewis et al. 2014).

Section 2.3.1 Biology

Sea pens are octocorals (soft corals) belonging to the Order Pennatulacea. They consist of multiple polyps living in a whip-like colony up to 2 m long (DFO 2010a) that attaches itself to soft-bottom seafloors using their peduncle. They occur from the intertidal zone to abyssal depths (ICES 2008, Baillon et al. 2014a), and may be sparsely distributed, form fields, and even form large thickets (Baillon et al. 2014b). Sea pens are slow-growing, long-lived organisms that can live up to 40 years and mature at a late age (Baillon et al. 2013).

Most sea pen colonies are gonochoric (separate sexes) broadcast spawners, although hermaphroditic colonies have been observed. The spawning period of two common cosmopolitan sea pen species, *Anthoptilum grandiflorum* and *Halopteris finmarchica*, varies between regions but occurs during late April in southern Newfoundland. Spawning followed the regional spring phytoplankton bloom when bottom water temperature varied between 3.6-4.8 °C. However, other deep-sea pennatulaceans such as *Pennatula aculeata* are reported to have aperiodic reproduction. Female and male

A. grandiflorum produce an average of just 13 oocytes and 48 spermatocysts per polyp, respectively. Mature oocytes measured approximately 1100 µm, suggesting lecithotrophic larval development. Colonies become mature around 24 cm length, which can take several years (Baillon et al. 2013, Baillon et al. 2014a).

Sea pens play multiple roles in marine ecosystem in that they serve as a nursery (Baillon et al. 2012) and refuge (DFO 2010a), host specific fish assemblages (Baker et al. 2012), form biogenic habitat for multiple fish and invertebrate species (Baillon et al. 2014b), and therefore increase local species diversity (Gilkinson & Edinger 2009, DFO 2012a). Because they are habitat-forming species and particularly vulnerable to disturbance, sea pens have been identified as components of vulnerable marine ecosystems (VMEs) (ICES 2008, Kenchington et al. 2010).

Section 2.3.2 Distribution and habitat

A total of 38 coral taxa have been recorded in the NL-Labrador Shelves Biographic Zone (Kenchington et al. 2010), of which 14 different species of corals from three orders (Alcyonaceans, Pennatulaceans and Scleractinians) were identified from DFO research trawl surveys in the Laurentian Channel AOI (Figure 5). In general, corals in the Laurentian Channel AOI associate with the mud and clay substrate that dominate the region. Sea pens represent the most abundant and diverse (6 species) coral taxa within the AOI (DFO 2010a, Lewis et al. 2014). Moreover, Kenchington et al. (2010) identified the west side of the Laurentian Channel as a location of significant concentrations of sea pens in the NL-Labrador Shelves Biogeographic Zone (Figure 6). Cup corals (*Flabellum spp.*) also occur frequently within the AOI (DFO 2010a).

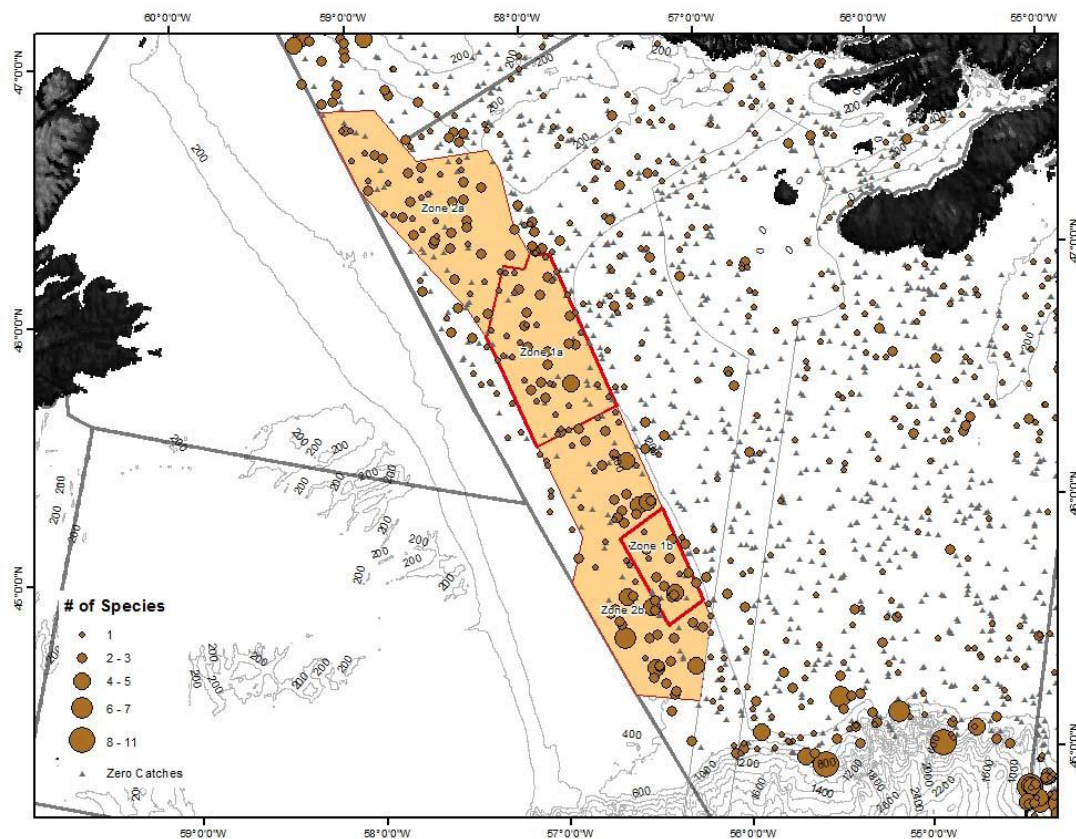


Figure 5: Distribution and diversity of coral species in the southern Newfoundland region. Darker region indicated the Laurentian Channel AOI. Data were collected by DFO multispecies surveys from 2004-2012 and include seapens, gorgonians, soft corals and cup corals species. Red boxes denote one iteration of management sub-zones proposed for the Laurentian Channel AIO. Source: Lewis et al. (2014).

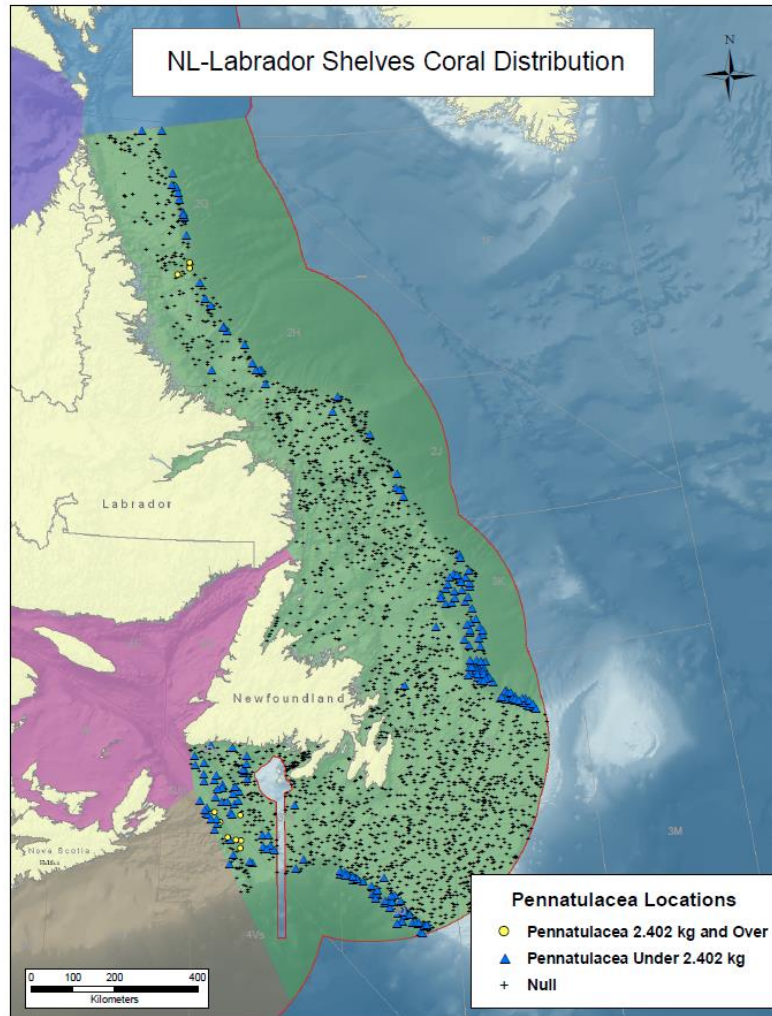


Figure 6: Location of sea pens catches in the NL-Labrador Shelves Biogeographic Zone from research vessel survey bycatch data. Yellow circles indicate significant concentrations of sea pens. Source: Kenchington et al. (2010).

Section 2.3.3 Spatial scales of movement

Because mature sea pen colonies are largely sessile with very limited movement dispersal occurs almost entirely through lecithotrophic larvae which may explain their cosmopolitan distribution (Baillon et al. 2013).

Section 2.3.4 Threats and limiting factors

Given their slow growth rates and longevity, sea pens and other corals are particularly vulnerable to bottom-contact fishing gear, especially bottom trawling (DFO 2012a). Impacts may range from displacement of individuals (DFO 2010a) to mortality. Bottom trawling reduces average coral biomass and size, indicating the removal of older and larger colonies and recolonization by younger and smaller colonies (Gilkinson & Edinger 2009). Habitat destruction and contamination from oil and gas exploration and exploitation also threaten corals (Lewis et al. 2014).

Section 2.3.5 Knowledge gaps

Abundance of corals within the AOI, which is based on kernel density estimates from Kenchington et al. (2010), remains uncertain and point to the need for studies to determine significant concentrations of corals using non-destructive methods such as ROV video transects.

Section 2.3.6 Conservation status

SARA: No Schedule; no status

COSEWIC: Not assessed

Section 2.3.7 Conservation objective considerations

The goal of the Laurentian Channel MPA for this CO is to “*Protect corals, particularly significant concentrations of sea pens, from harm due to human activities in the Laurentian Channel MPA*”. The closure to the Laurentian Channel MPA to all fisheries will likely achieve this goal. Moreover, the termination of bottom trawling, the main threat to corals, will likely result in increased coral density and biomass, therefore creating more nursery, refuge, and biogenic habitats, and enhancement of species diversity. Other known threats that will remain unchanged in zone 2a/b (Figure 4) such as oil and gas exploration and exploitation could potentially mitigate this positive result.

Section 2.4 Leatherback sea turtle (*Dermochelys coriacea*) – Laurentian Channel AOI

Section 2.4.1 Biology

The Leatherback sea turtle is a marine reptile and sole member of the family Dermochelyidae, a family characterized by the lack of a hard bony shell. The largest of the marine turtles, adults usually measure between 130-170 cm (carapace length) and weigh less than 500 kg, but they can reach over 2 m and weigh up to 916 kg. They have the fastest growth rate of living turtles, exceeding 8.5 cm/yr (Zug & Parham 1996). They likely reach sexual maturity at a minimum of 9 years and although their life span remains unknown, they can live at least 23 years. Females nest on exposed and open sandy beaches adjacent to deep waters. Hatchlings usually measure between 5.0-6.5 cm and weigh 40-55 g. (Zug & Parham 1996). Hatchlings immediately move to the marine environment where males remain for their entire life span and females return to land only to nest (COSEWIC 2012a). Nesting typically occurs every 2-3 years (Hughes 1996) and generation time ranges from 30-35 years (COSEWIC 2012a).

Leatherback abundances are difficult to estimate because most of its life cycle is pelagic, and most estimates are derived from nesting females on monitored beaches (Atlantic Leatherback Turtle Recovery Team 2006). Although the Pacific population faces imminent extinction (Spotila et al. 2000), decline in the Atlantic population appears slower, with ~18, 800 nesting females in 1996 (Spotila et al. 1996) declining to ~15 000 nesting females in 2000 (Atlantic Leatherback Turtle Recovery Team 2006). Using aerial and shipboard surveys, Shoop and Kenney (1992) estimated that 100-900 leatherbacks transited through their study area between Nova Scotia and Cape Hatteras in North Carolina, an average of 6.85 turtles per 1000 km of survey track. Canada lacks such estimates and relies on opportunistic reports from fisherman, phone and mail surveys, and the entanglement and stranding networks for cetaceans and sea turtles. Nonetheless this information suggest an even larger density of leatherbacks in Atlantic Canada than along the northeastern United States (Atlantic Leatherback Turtle Recovery Team 2006). Recent estimates place the population of adult leatherbacks in the North Atlantic between 34,000 and 94,000 individuals (COSEWIC 2012a).

Leatherback sea turtles feed primarily on jellyfish (medusae) and soft-bodied invertebrates. In order to maintain their high metabolic rate, they must consume a biomass of prey comparable to their weight, restricting their foraging to highly productive zones such as coastal areas and along oceanic frontal systems (Atlantic Leatherback Turtle Recovery Team 2006).

Section 2.4.2 Distribution and habitat

Leatherback sea turtles occur in the Atlantic, Indian and Pacific Oceans, spanning the largest geographic range of any reptile. They live in temperate and tropical waters from 71°N to 47°S and undertake extensive migrations that may exceed 10 000 km (COSEWIC 2012a). In Atlantic Canada, adult leatherbacks primarily occupy continental shelf and slope waters in summer and fall during their annual migration between their breeding areas at low latitudes (e.g. shorelines of South and Central America, Caribbean and continental U.S.) and temperate foraging habitat, including the south coast of Newfoundland (Figure 7). Although leatherback foraging has been documented near the Laurentian Channel AOI, in waters south and east of the Burin Peninsula and in parts of Placentia Bay (Atlantic Leatherback Turtle Recovery Team 2006, DFO 2012b, 2013b), they likely use the AOI as a migration corridor where they do not actually feed. Moreover, from 1967-2012, only a few sightings of leatherbacks were reported within the Laurentian Channel AOI (Figure 8). Leatherbacks occur in Atlantic Canada mainly from July to mid-October and less frequently from April through December (Atlantic Leatherback Turtle Recovery Team 2006, James et al. 2006, COSEWIC 2012a). They usually exhibit high fidelity to foraging sites in either the eastern or western Atlantic but may change their migratory routes from year to year (James et al. 2005a, COSEWIC 2012a).

Section 2.4.3 Spatial scales of movement

Mature leatherback sea turtles undertake extensive annual migrations from their breeding grounds in the tropical waters to their feeding grounds in cooler temperate waters of the North Atlantic. Therefore, they cover the largest spatial scale of all CO focal taxa in the Laurentian Channel MPA.

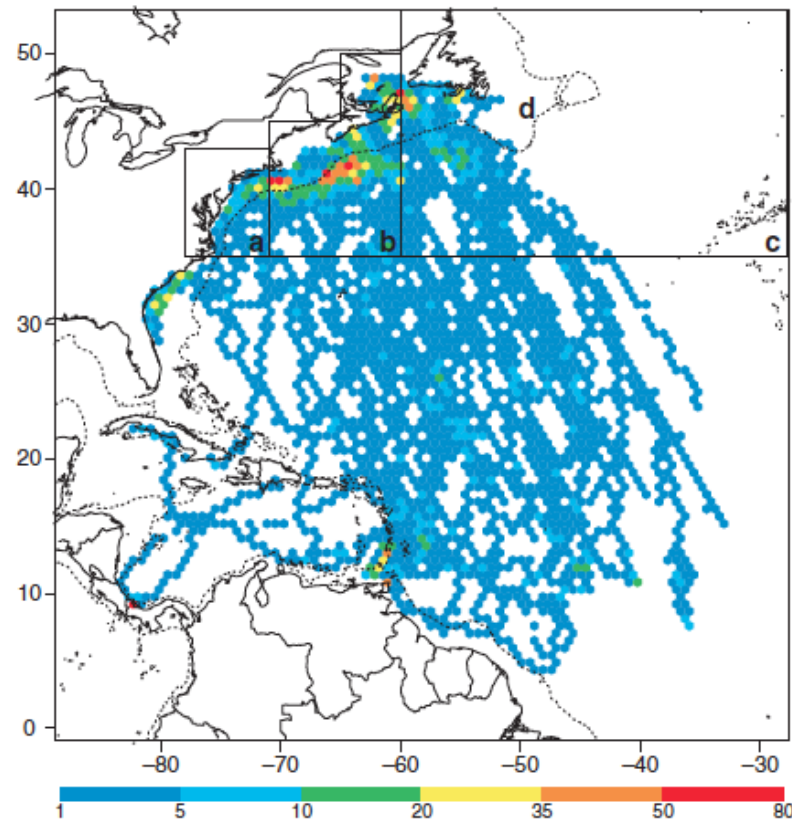


Figure 7: Geographic distribution of 38 Leatherback sea turtles equipped with Argo satellite tags off Nova Scotia during the summers from 1999-2003. Colour represents the number of days turtle(s) spent in each hexagon. Source: James et al. (2005b).

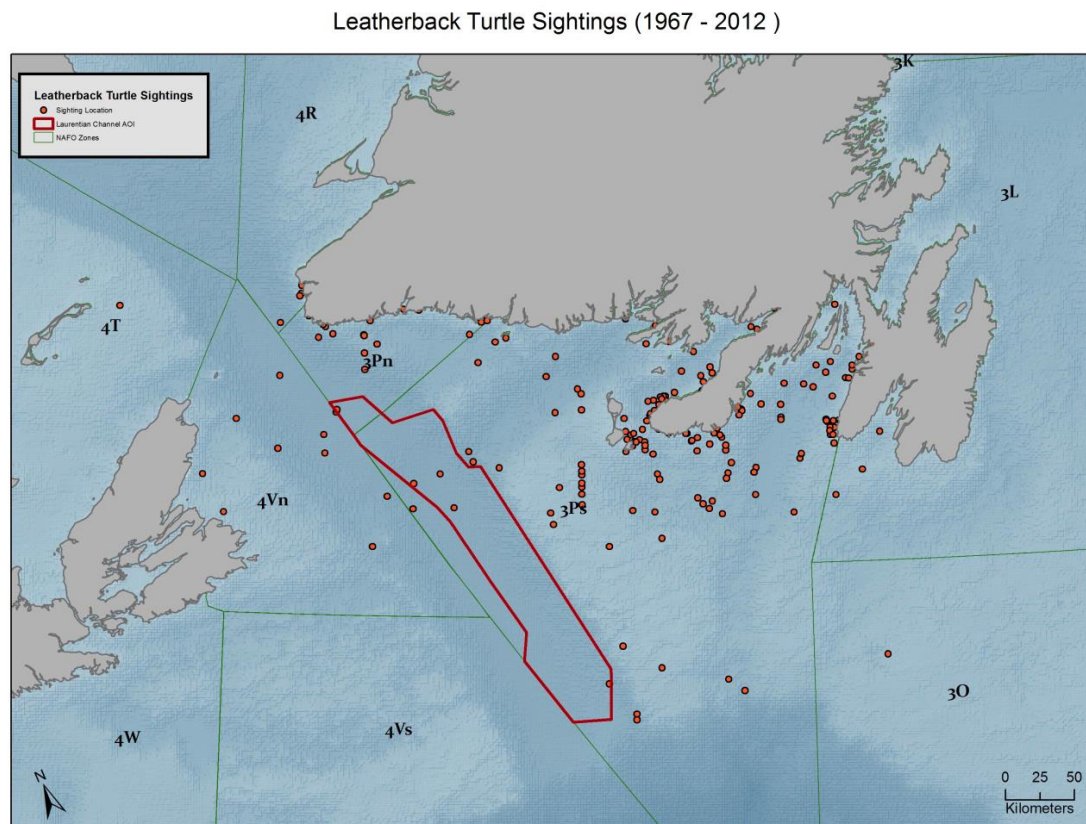


Figure 8: Sightings of leatherback sea turtles over the 1967-2012 period. Red zone indicates location of the Laurentian Channel AOI. Source: Fisheries and Oceans Canada.

Section 2.4.4 Threats and limiting factors

Nesting females have declined over 70% since the 1970s. Intense egg harvesting on nesting beaches, coastal development, and high adult mortality resulting from entanglement with fishing gear all contribute to this decline. In Atlantic Canadian waters entanglement with fixed fishing gear such as longlines, lines of pot gear, gillnets and buoys, ropes and cables represents the primary threat. Although not well quantified, vessel strikes, marine pollution (e.g. plastic debris, balloons), and anthropogenic noise add further threats throughout their distributional range. Oil and gas exploration and development, with associated seismic airguns, are known to induce important behavioural changes. Global warming could also impact leatherbacks by influencing temperature-dependent sex determination, increasing

wind and wave erosion on nesting beaches, and altering present oceanic current patterns (Atlantic Leatherback Turtle Recovery Team 2006).

Section 2.4.5 Knowledge gaps

Many knowledge gaps remain regarding leatherbacks in Atlantic Canadian waters, including aspects of their biology, distribution, habitat requirements, and threats (Atlantic Leatherback Turtle Recovery Team 2006). In the context of the AOI, additional research is needed to assess their seasonal abundance and distribution, and to identify their main foraging habitats.

Section 2.4.6 Conservation status

SARA: Schedule 1, Endangered COSEWIC: Endangered

Section 2.4.7 Conservation objective considerations

Despite limited fishing activity in the Laurentian Channel AOI for the past ~15 years, and especially for fixed fishing gear such as pots and longlines, the complete closure of the AOI to all fisheries should reduce lethal encounters with leatherbacks. Therefore, this conservation objective for the Laurentian Channel MPA, which is to “*Promote the survival and recovery of leatherback sea turtle by minimizing risk of harm from human activities in the Laurentian Channel MPA*”, is likely to be achieved. This reduction will also help to achieve Objective 4 “*Minimize risk of harm to leatherback turtles from anthropogenic activities under Canadian jurisdiction*” of the Recovery Strategy for Leatherback Turtles in Atlantic Canadian Waters (Atlantic Leatherback Turtle Recovery Team 2006). However, because they use the entire continental shelf and productive shelf break during their migration (Witzell 1999), the impact of the Laurentian Channel MPA on the overall Atlantic population will likely be limited (Figure 7). Moreover, injuries resulting from vessel strikes and oil and gas exploration and development will likely remain unchanged within the AOI.

Section 2.5 Northern wolffish (*Anarhichas denticulatus*) – Laurentian Channel AOI

Section 2.5.1 Biology

The northern wolffish is a benthopelagic fish of the family Anarhichadidae that feeds primarily on benthic invertebrates such as crustaceans, echinoderms, and molluscs but also on pelagic fish and jellyfish. Individuals over 125 cm are rare but some reach up to 180 cm in length (COSEWIC 2012b).

Northern wolffish biology remains poorly known, but females reach 50% sexual maturity at ~5.5 years and measure ~75 cm. Their generation time is approximately 10.5 years. Spawning occurs in late fall or early winter with internal fertilization and subsequent deposition of egg masses on the seafloor in rocks and crevices. Adults mainly live on the seafloor but, unlike the two other local wolffish species, they also feed in the water column (Templeman 1984, Kulka et al. 2007, DFO 2010a, COSEWIC 2012b).

The main natural predators of juvenile northern wolffish are Atlantic cod, Atlantic halibut, haddock, and seals (COSEWIC 2012b).

Section 2.5.2 Distribution and habitat

Northern wolffish inhabit cold waters in the Arctic and on both sides of the Atlantic. In Canada, they occur from the Gulf of Maine to the north of the Bay of Fundy, the Grand Banks, Gulf of St. Lawrence, northeastern Newfoundland, Labrador Sea up to Baffin Island and west of Greenland, with a few additional records in the western Arctic (Figure 9). Over 2.5 million northern wolffish are estimated to live in Canadian waters, including over 1 million adults (COSEWIC 2012b).

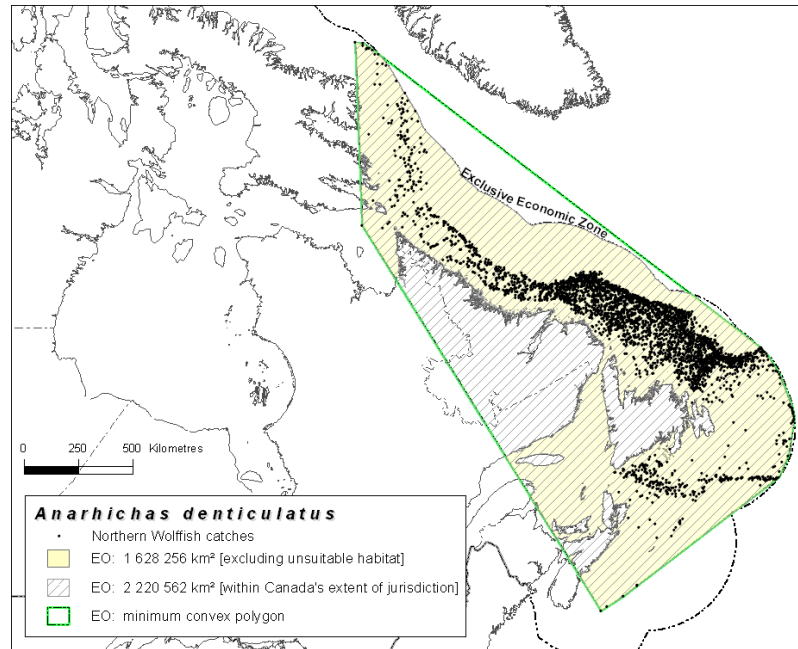


Figure 9: Distribution map of northern wolffish in Canadian waters from DFO research vessels surveys. Source: COSEWIC (2012b).

Northern wolffish most commonly inhabit cold (between 2-5 °C) continental shelf and slope waters from 500-1000 m but also occur between 38-1504 m. Historically they occurred on all bottom types, but since their decline in the 1980s most individuals have been caught on sandy and shell hash seafloor. Water temperature apparently influences their distribution and habitat use (Kulka et al. 2007, COSEWIC 2012b).

Section 2.5.3 Spatial scales of movement

Eggs are thought to be deposited on the seafloor and hatchlings spend time around the “nest”. Later in their development, larvae and juveniles occupy the upper layer of the water column before settling on the seafloor. A tagging study of mature northern wolffish reported limited movement and migration (Templeman 1984, COSEWIC 2012b).

Section 2.5.4 Threats and limiting factors

Research surveys that began in the 1970s demonstrated a major decline in northern wolffish during the 1980s, both in abundance (~96% decrease) and in geographic range. Small increases have been reported in abundance and range since 2002 (COSEWIC 2012b).

Although no commercial fishery targets northern wolffish, bycatch in bottom trawling fisheries remains their main threat. Individuals taken as bycatch were released as standard practice even before their listing as Threatened under the Species at Risk Act in 2003 made release mandatory. Disturbance of their seafloor habitat by bottom trawling and climate change are also considered important threats (COSEWIC 2012b). In fact, ocean warming resulting from climate change may result in a northerly shift of the species as reported in other northeastern Atlantic fish species (Perry et al. 2005).

Section 2.5.5 Knowledge gaps

Little is known about the biology of northern wolffish, such as their reproduction and predators. Also, no information is available on bycatch mortality and the effect of climate warming on their distribution remain unknown (COSEWIC 2012b).

Section 2.5.6 Conservation status

SARA: Schedule 1, Threatened COSEWIC: Threatened

Section 2.5.7 Conservation objective considerations

Northern wolffish could benefit from a closure of the fishery in the Laurentian Channel MPA given their vulnerability as bycatch. Therefore, this Laurentian Channel MPA conservation objective, which is to “*Promote the survival and recovery of Northern Wolffish by minimizing risk of harm from human activities in the Laurentian Channel MPA*”, is likely to be achieved. The cessation of bottom trawling activities could also indirectly benefit northern wolffish by increasing sea pens and other corals densities, since wolffishes occur in higher numbers when corals are

present (Gilkinson & Edinger 2009). However, given that water temperature strongly influences their distribution and that the Laurentian Channel MPA represents the southernmost limit of their distribution range, predicted increases in water temperature may push their distributional range northward in the long-term and mitigate benefits the Laurentian Channel MPA might provide to this species.

Section 2.6 Smooth skate (*Malacoraja senta*) – Laurentian Channel AOI

Section 2.6.1 Biology

The smooth skate (*Malacoraja senta*) is a cartilaginous fish of the family Rajiidae. It is one of the smallest skate species in the northwest Atlantic Ocean, rarely measuring over 67 cm total length, or weighing more than 1.2 kg. They also have the longest tails relative to their length of all skates (COSEWIC 2012c).

Length at 50% maturity is estimated at 50 cm and 47 cm, and age at 50% maturity at 12 and 10 years for males and females, respectively (McPhie & Campana 2009b, a, Simpson et al. 2012). Females lay between 41-56 egg capsules on the seafloor, from which fully formed juveniles (7-10 cm long) develop. The time between extrusion and hatching is unknown but estimated from other skate species at 1-2 years (McPhie & Campana 2009b, COSEWIC 2012c). The estimated generation time of 16 years probably represents an underestimate (COSEWIC 2012c, Simon et al. 2012). As in other skate species, a slow growth rate, late sexual maturity, low fecundity, long reproductive cycles, and long lifespan characterize smooth skate. These characteristics make them particularly vulnerable to even low fishing mortality (McPhie & Campana 2009b, a).

Smooth skates feed selectively on small crustaceans for most of their life, switching to fishes when they reach their largest sizes (COSEWIC 2012c).

Little information is available on natural predators, and they are conspicuously absent from stomach content analyses of 68 potential predator species. Nonetheless, some marine mammals (e.g. grey seals) prey upon adult smooth

skates and large fish species (e.g. Atlantic halibut, goosefish and Greenland sharks) and gastropods (COSEWIC 2012c) consume their egg capsules.

Section 2.6.2 Distribution and habitat

Smooth skates are endemic to the northwest Atlantic, with a patchy distribution throughout Atlantic Canada. They form four different designatable units (DUs) in the northwest Atlantic, distinctly separated by areas where the species is absent (Figure 10). Smooth skates within the Laurentian Channel AOI (Figure 11) are part of the Laurentian-Scotian DU (COSEWIC 2012c).

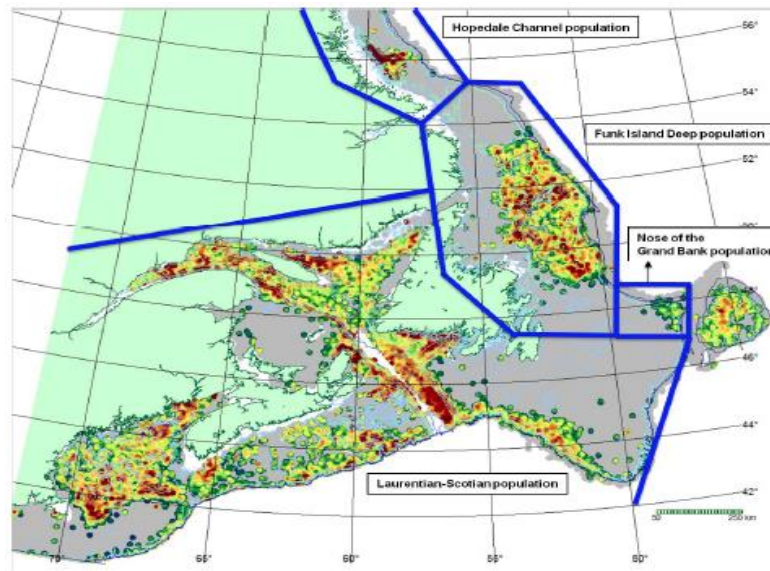


Figure 10: Distribution of smooth skate in Atlantic Canada from 1971-2009. Source: COSEWIC (2012c)

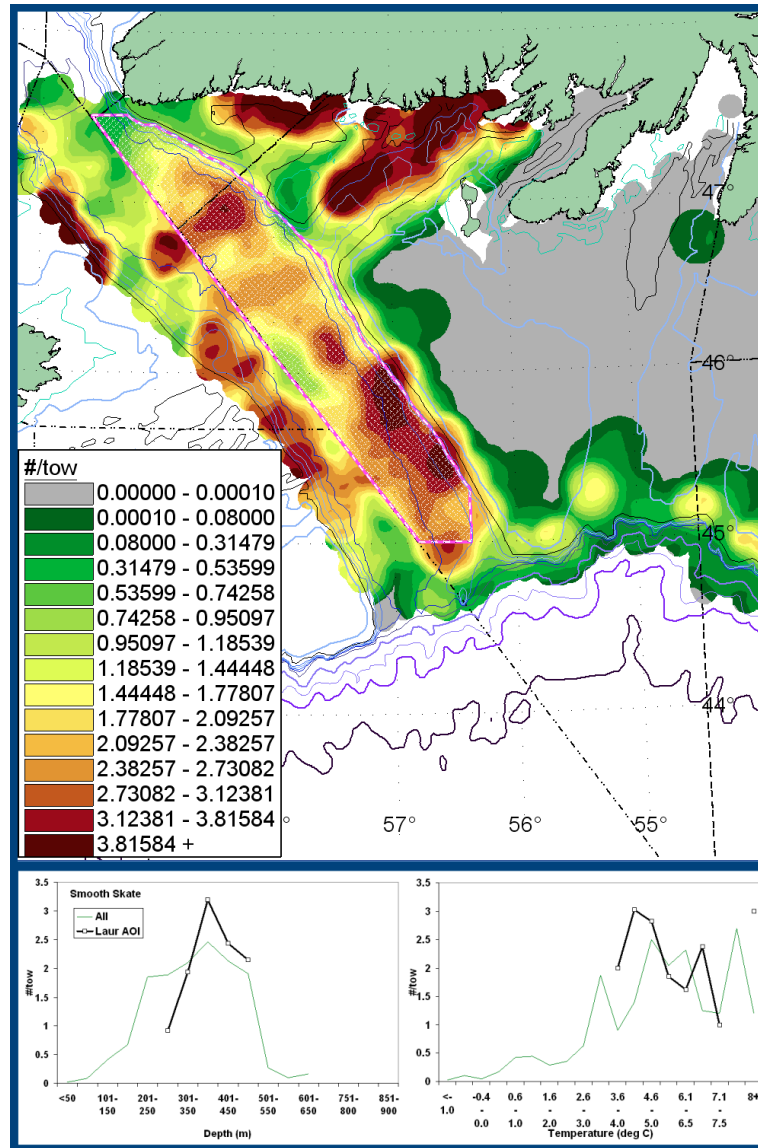


Figure 11: Smooth skate distribution (# of fish/tow) and association with depth and temperature in the Laurentian Channel and on St. Pierre Bank. Dotted pink line represents the old AOI delineation. Note that the actual AOI delineation is smaller in size. Source: Kulka and Templeton (2013).

Canadian waters harbour approximately 80% of the global smooth skate population. Individuals are generally restricted to areas of the continental shelf, with highest densities in the warmer troughs surrounding banks. They generally occur at depths ranging from 70-480 m and at temperatures between 2.7-10°C, apparently seeking out their

preferred temperature and adjusting their depth accordingly. Nonetheless, they have been reported from depths spanning 25-1436 m and temperatures between -1.3-15.7 °C (COSEWIC 2012c, Simpson et al. 2012).

Smooth skate is the second most common skate species in Atlantic Canada following thorny skate. The most recent minimum trawlable abundance estimate for all Atlantic Canadian waters is 44 million individuals. The Laurentian-Scotian DU alone supports an estimated 37.4 million individuals, of which only 5.7 million (14%) are considered mature. This estimate represents 90% of the total estimated abundance of smooth skate in Atlantic Canada. Total estimated abundances have varied markedly over the years, with the steepest declines in the Funk Island Deep DU and the largest increases in the Laurentian-Scotian DU. However, all DUs have remained stable or increased slightly since early 2000 (COSEWIC 2012c).

The Laurentian-Scotian DU covers 48% of the global area of occupancy (AO) of smooth skates and have increased from ~65 000 km² in 1990-1992 to ~120 000 km² in 2008-2010. A closer look at the southwest Grand Banks and southern Laurentian Channel subregion, which include the Laurentian Channel AOI, also shows an increase in the AO in this subregion. In fact, the AO expanded from about 20,000 km² in the mid-1970s to about 30,000 km² after 1995 (Figure 12). In all Canadian waters, the AO fluctuated from approximately 155,000 km² in 1978-1980, to 80,000 km² by mid-1990s and has since increased to about 160,000 km². The marked increase of the Laurentian-Scotian DU largely drove this increase in parallel with the largest decline and fragmentation in the Funk Island Deep DU (COSEWIC 2012c).

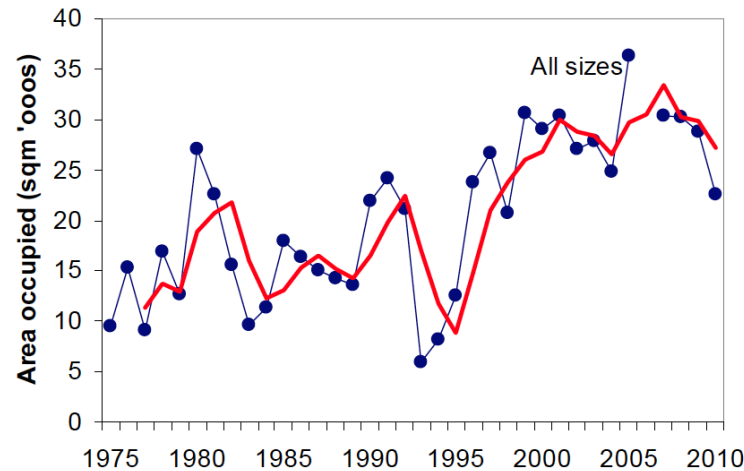


Figure 12: Area occupied by smooth skates in the Laurentian-Scotian DU from 1975-2010. Red line indicates the 3-year running average. Source: COSEWIC (2012c).

Smooth skates generally occur on soft-muddy seafloors but are also reported from broken shell, sand, gravel and pebbles bank habitats (COSEWIC 2012c). Juvenile smooth skates (<48 cm) generally occur within the Laurentian Channel AOI whereas larger adults comprise the majority of the population on the southwest slope and shelf edge of the Grand Bank (Simpson et al. 2012).

Section 2.6.3 Spatial scales of movement

As a relatively sedentary species with limited dispersal (100-440 km) and demersal eggs; the low probability of mixing between the most widely separated DUs suggest a high probability of genetic isolation and little likelihood of repopulation from one DU to another (COSEWIC 2012c).

Section 2.6.4 Threats and limiting factors

Although no commercial fishery targets smooth skate, the overall population decreased by over 90% since 1970 (McPhie & Campana 2009b), largely from bycatch in fisheries targeting others species (Simpson et al. 2012). In fact, abundance declined to their lowest values from the 1980's to mid-1990's, but increased to greater levels since

(COSEWIC 2012c, Simpson et al. 2012). The combination of slow growth, late sexual maturity, low fecundity, and long reproductive cycles make them particularly vulnerable to even modest levels of fishing mortality (McPhie & Campana 2009b, a). The highest reported smooth skate bycatch occurs in the Laurentian-Scotian DU, where they are usually caught in skate longline, crab pot, cod and redfish otter trawl, and scallop dredge fisheries. Post-discard mortality is unknown but could exceed 40% based on data from other skates (COSEWIC 2012c, Simpson et al. 2012). Water temperatures also affect smooth skate distribution; below-average water temperature may have contributed to decreased northern Hopedale and Funk Island Deep DU populations in the 1980s and 1990s (COSEWIC 2012c).

Section 2.6.5 Knowledge gaps

The reproductive cycle of the smooth skate has not been well described and some life history parameters remain largely unknown. Smooth skates are also historically absent from large and, at first glance, possibly suitable habitats. Other factors that may influence smooth skate distributions, such as prey availability and bottom types, require further investigation (COSEWIC 2012c).

Section 2.6.6 Conservation Status

SARA: No schedule, No status COSEWIC: Special Concern

Section 2.6.7 Conservation objective considerations

As a bottom dweller, smooth skate may avoid bottom trawling fishing gears by flattening against the seafloor (Simpson et al. 2012); bottom trawls represent the main fishing gear used in the Laurentian Channel AOI since 1999 (Supplemental Figure 3). This avoidance behaviour, considered in concert with low fishing activities in the AOI since 1999 (Supplemental Figures 2-6), helped the smooth skate population in this area increase to its highest recorded area of occupancy (Figure 12) and abundance. The complete closure of the Laurentian Channel AOI to all fisheries should therefore lead to further increase in smooth skate population size and AO within the AOI. Therefore, this conservation objective of the Laurentian Channel MPA, which is to “*Protect Smooth Skate from human induced mortality in the*

Laurentian Channel MPA”, is likely to be achieved. Furthermore, increased population size within the AOI could produce a spillover effect on nearby troughs and banks within the Scotian-Laurentian DU. However, the sedentary nature of this species limits the likelihood of a spillover effect enhancing population sizes and AOs in other isolated DUs where large population declines occurred, such as the Funk Island Deep DU (COSEWIC 2012c).

Section 2.7 Black dogfish (*Centroscyllium fabricii*) – Laurentian Channel AOI

Section 2.7.1 Biology

The Black dogfish (*Centroscyllium fabricii*) is a small schooling Chondrichthyes (cartilaginous fish) of the family Etmopteridae. Individuals can reach up to 107 cm TL but mature females usually measure 58-70 cm TL. Their growth rate is unknown (Campagno 1984, Scott & Scott 1988). They usually become mature at 55 and 65 cm TL in males and females, respectively (Yano 1995) and are ovoviviparous; producing eggs that develop within the body of the female without placental attachment (Campagno 1984, Scott & Scott 1988). Near-term embryos may be as large as 19 cm before birth, and litter size ranges between 4-40 pups (Yano 1995). Black dogfish primarily feed on cephalopods, pelagic crustaceans, jellyfish, and small redfish (Campagno 1984, Scott & Scott 1988).

Section 2.7.2 Distribution and habitat

Black dogfish span Atlantic Ocean basins from Iceland to the southwest coast of Africa in eastern Atlantic and they also occur on the Mid Atlantic Ridge. In the northwest Atlantic, they occur from south of Greenland and Baffin Island, down to Cape Hatteras and possibly off Florida and into the Gulf of Mexico. In Atlantic Canadian waters, they occur along the continental shelf and slope off Labrador, around Newfoundland (Figure 13), in the Laurentian Channel (Figure 14), on the Scotian Shelf and on Georges Bank (Campagno 1984, Scott & Scott 1988, Kulka 2006, Kulka & Templeman 2013).

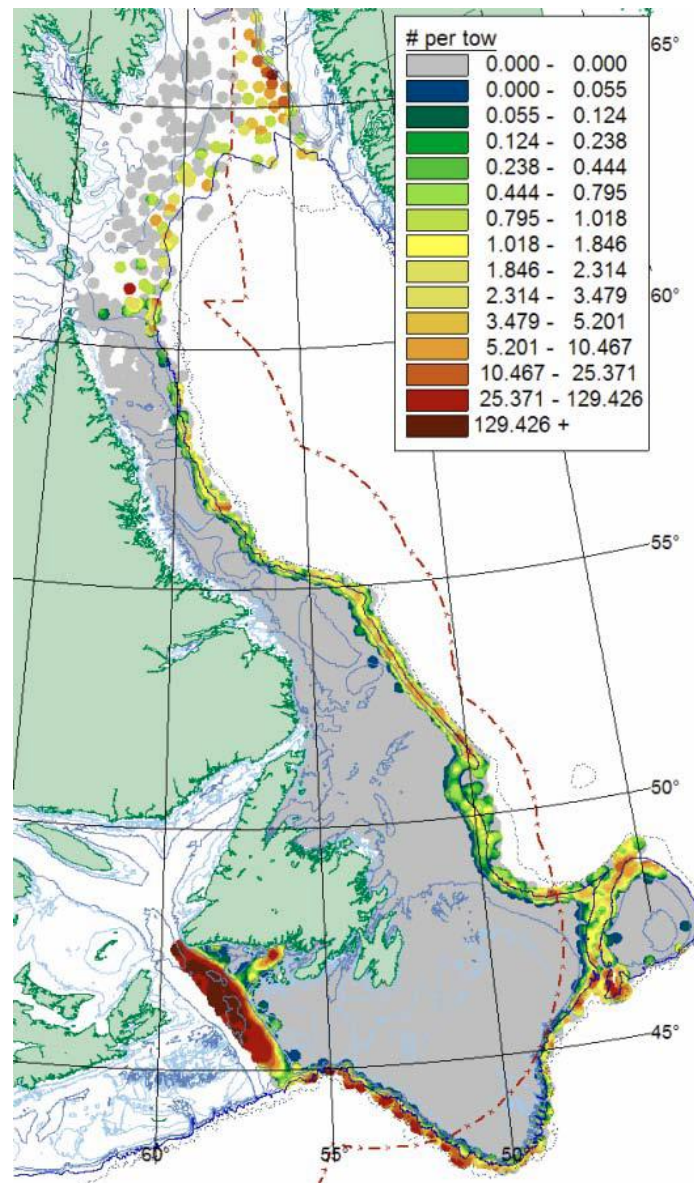


Figure 13: Black dogfish distribution on continental shelf and slope waters around Newfoundland and Labrador. Data based on NL trawl surveys from 1971-2005. Source: Kulka (2006).

Black dogfish are bathydemersal, occupying seafloor and water column habitats in deep waters ranging from 180-1600 m but usually from 460 m and deeper in Canadian waters. They occasionally occur near the surface during winter or in cold subarctic waters (Campagno 1984, Scott & Scott 1988). Female numbers also tend to exceed numbers of males below 1000 m depth (Yano 1995). Black dogfish is the second most abundant species in Laurentian Channel

after redfish. In the Laurentian Channel AOI, they mostly occupy depths ranging from 350-500 m and waters above 3.8 °C, ranging from 4.6-6.5 °C (Kulka 2006, Kulka & Templeman 2013).

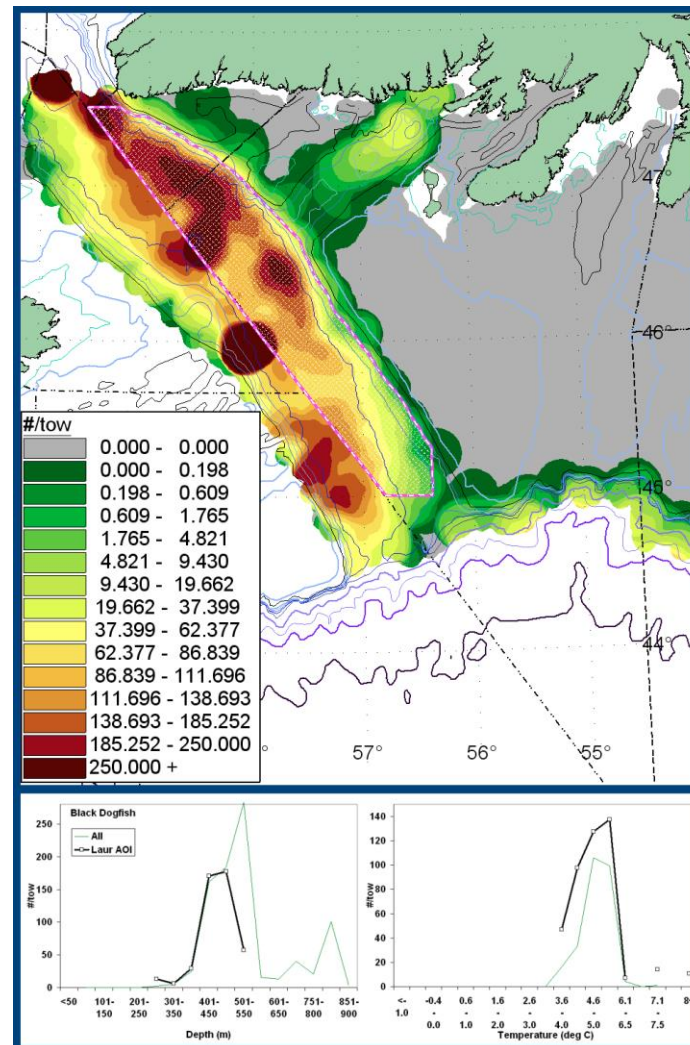


Figure 14: Black dogfish distribution (# of fish/tow) and association with depth and temperature in the Laurentian Channel and St. Pierre Bank. Dotted pink line represents the old AOI delineation. Note that the actual AOI delineation is smaller in size. Source: Kulka and Templeton (2013).

Black dogfish occur throughout Atlantic Canadian slope waters with high concentrations in the Laurentian Channel AOI, approximately 10 times more than in other areas. Their abundance has fluctuated in time; following

declines since the mid-1990s abundances have remained stable in recent years (Figure 15). However, the juvenile population in the Laurentian Channel is believed to be in decline since 1995 (Kulka 2006).

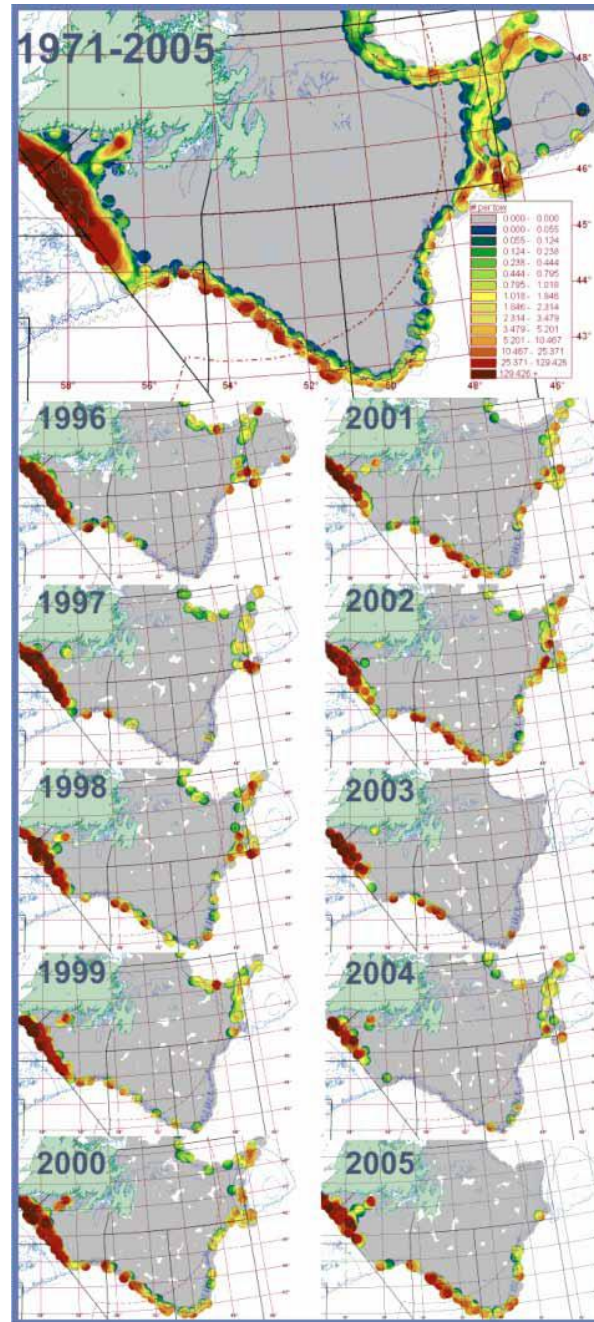


Figure 15: Distribution of black dogfish in the Grand Banks region based on NL trawl survey data from 1971-2005.
Source: Kulka (2006).

Section 2.7.3 Spatial scales of movement

The Laurentian Channel is a key habitat for this species because it is the only known pupping ground in the northwest Atlantic. In fact, pregnant females usually migrate to the shallower (< 400 m) region of the Laurentian Channel where pupping occurs. Juveniles later migrate to deeper waters of the Channel and then move further to deeper slope waters as they mature, resulting in a marked increase in length with depth. Their movement appears to be based on reproductive activity and maturity rather than seasons (Kulka et al. 2006, Kulka & Templeman 2013).

Section 2.7.4 Threats and limiting factors

Although no commercial fishery targets black dogfish, they are regularly caught as bycatch by bottom trawls and bottom line gears, mostly in the Greenland halibut fishery. Catches are therefore usually thrown back at sea (Campagno 1984, Yano 1995). Bycatch in Canadian waters was estimated at 68 tons annually on average for the 1996-2005 period (Kulka 2006).

Section 2.7.5 Knowledge gaps

Black dogfish are poorly studied, and little is known about its biology, especially the timing of pupping. Studies on the reproduction of this species and the importance of the Laurentian Channel for this key juvenile life history parameter are presently underway (Kulka, pers. comm.).

Section 2.7.6 Conservation Status

SARA: Not listed COSEWIC: Not assessed

Section 2.7.7 Conservation objective considerations

The Laurentian Channel is the only known pupping ground for this species in Atlantic Canadian waters. Black dogfish densities are 10 times higher in this region than on adjacent slope waters and the vast majority of this population is composed of young of the year and juveniles, which have declined since 1995 (Kulka et al. 2006). The closure of the Laurentian Channel MPA to all fisheries is expected to protect this key life stage by decreasing juvenile mortality in bycatch, thus leading to an increase of the juvenile population. Therefore, this conservation objective of the Laurentian

Channel MPA, which is to “*Protect Black Dogfish from human induced mortality in the Laurentian Channel MPA*”, is likely to be achieved. In the long term, this increase could further result in more individuals migrating to the deeper slope waters and increasing the size of the mature population inhabiting the nearby slope waters.

Section 2.8 Porbeagle shark (*Lamna nasus*) – Laurentian Channel AOI

Section 2.8.1 Biology

The porbeagle shark (*Lamna nasus*) is a large pelagic cold-temperate species of the family Lamnidae (Campana et al. 2001). In the northwest Atlantic, they measure 58-67 cm fork length (FL) at birth; males and females grow to 253 and 302 cm FL, respectively. Several characteristics limit reproductive output. Males become sexually mature between 6-10 years (50% maturity at 8 years) and females between 12-16 years (50% maturity at 13 years) (Francis et al. 2008). They are ovoviviparous and oophagous, with a gestation period of ~8-9 months; mean litter size averages 4 pups (Francis et al. 2008, Campana et al. 2013). Their estimated life span is between 25 and 46 years, with a generation time of ~18 years (Campana et al. 2013). However, vertebral band counts, on which estimates are based, may underestimate the age of older individuals (>25 years), as reported in the New Zealand population (Francis et al. 2007, Campana et al. 2013).

One of the only two known mating grounds occurs off southern NL and the entrance of the Gulf of St. Lawrence, including the AOI; mating occurs in fall from late September through November (Jensen et al. 2002). The other mating ground in Canadian waters is on Georges Bank (Campana et al. 2010, Campana et al. 2013).

In terms of feeding they are primarily opportunistic piscivores but sometimes feed on cephalopods (Joyce et al. 2002). Humans are their only known predator.

Section 2.8.2 Distribution and habitat

Porbeagle sharks occupy coastal and oceanic waters (Campana et al. 2002). They occur in the northwest and northeast Atlantic, and circumglobally in the Southern hemisphere (Campana et al. 2002, Francis et al. 2008). They occur within the AOI between May and late fall (Campana et al. 2013), though Fisheries Observer Data report relatively few individuals within the AOI (Figure 16).

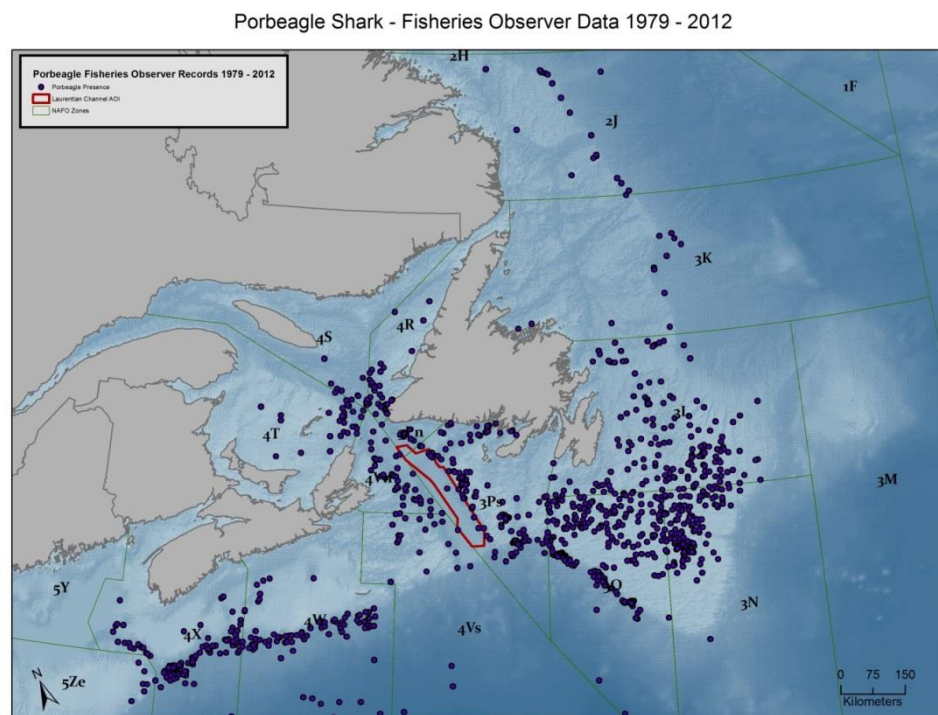


Figure 16: Porbeagle shark occurrence based on Fisheries Observer Data from 1979-2012. Source: Fisheries and Oceans Canada

Since 2000, most porbeagle sharks were caught on shelf edges and deep basins. The closure of the shark fishery in 2000 on the shelf mating grounds of the Newfoundland region, which includes the AOI, likely contributed to this catch bias (Campana et al. 2013).

Temperature largely defines their distribution in the water column and porbeagle sharks are believed to adjust location according to preferred temperature range. They usually occupy waters between 5-10 °C and depths between 35 to 3000 m (Campana & Joyce 2004, Campana et al. 2013).

Section 2.8.3 Spatial scales of movement

Porbeagle sharks from the northwest and northeast Atlantic are believed to form two different stocks with no significant exchange between the two stocks (Campana et al. 2001). In the northwest Atlantic, porbeagle sharks migrate extensively along the east coast of Canada between the Gulf of Maine and Newfoundland (Campana et al. 2002). A satellite pop-up tag study on 21 individuals showed that most males and juveniles remained in Canadian waters all year, although some individuals moved offshore. However, mature females migrated 2356 km during winter at depths ranging to 1360 m beneath the Gulf Stream to a major pupping ground in the Sargasso Sea (Campana et al. 2010, Campana et al. 2013).

Section 2.8.4 Threats and limiting factors

The main threat to porbeagle sharks is bycatch by the longline fishery. Since 2000, most porbeagle sharks were caught on shelf edges and deep basins. The closure of the shark fishery in 2000 on the shelf mating grounds of the Newfoundland region, which includes the AOI, likely contributed to this catch bias (Campana et al. 2013).

The abundance of porbeagle sharks has declined significantly since the beginning of a directed commercial fishery in 1961; the estimated population size of between 197,000 to 207,000 individuals in 2009 represents just 22 to 27% of 1960s value. The estimated 11,000 to 14,000 mature females represents just 12-16% of the pre-commercial fishery level. Population size is thought to have remained stable since the reduction of quotas in 2002. This significant reduction of the total allowable catch (TAC) could allow a slow recovery of the stock if human-induced mortality remains low (Campana et al. 2013).

Section 2.8.5 Gaps and challenges

Much uncertainty surrounds actual population size and the potential for population recovery under current management plans (Campana et al. 2013).

Section 2.8.6 Conservation status

SARA: No schedule, No status COSEWIC: Endangered

Section 2.8.7 Conservation objective considerations

Even though the region covered by the AOI has been closed to the porbeagle shark fishery since 2000 and the species has not been reported as bycatch by the swordfish and tuna longline fisheries (Figure 17), a total closure to all fisheries in the AOI could reduce porbeagle shark bycatch and help in their recovery. Therefore, this Laurentian Channel MPA conservation objective, which is to “*Protect porbeagle shark from human induced mortality in the Laurentian Channel MPA*”, is likely to be achieved. However, with little to no bycatch from the AOI and the vast majority from the basins and shelf edges adjacent to the AOI (Campana et al. 2011), population benefits may be modest. In conclusion, the AOI will likely have a very limited, though positive effect on porbeagle shark recovery.

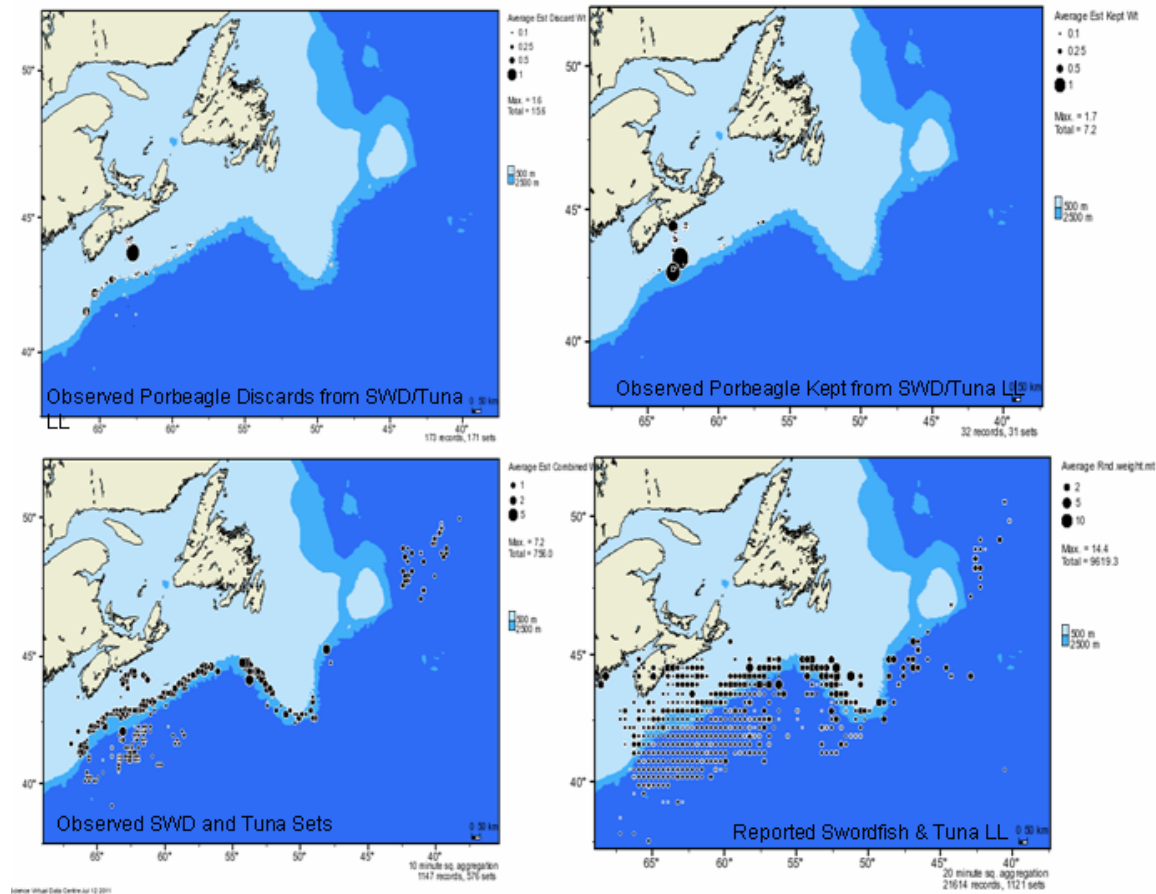


Figure 17: Porbeagle catch during swordfish and tuna fisheries in the 3rd quarter from 2003-2010 in Atlantic Canada. Source: Campana et al. (2011)

Section 3: Monitoring and Scale: Global approaches to MPA monitoring (*Objectives 2 and 3*)

Managers and researchers globally employ many approaches to monitor marine protected areas. The wide range of MPA sizes (Figures 1- 2), protection levels (Supplemental Table 1), and objectives, precludes any panacea for effective MPA evaluation. Most monitoring efforts measure response by surveying a biological attribute using a hierarchical design that attempts to replicate samples within and outside the intervention (protected and/or non-protected areas) (IUCN 2004, Pomeroy et al. 2005, Fox et al. 2014). This approach provides a method to evaluate differences through time, space, or between management actions, while accounting for the inherent variability in the system.

Biological attributes measured for a given species or life-history often vary spatially and temporally (reviewed in Sections 1.1.1 and 1.1.2 as well as for each species in Section 2), and capturing this inherent variability in the system with a hierarchical design therefore provides the most pragmatic approach to evaluating how a system will respond to protection (Willis 2013).

Designing and updating monitoring approaches for MPAs depends on the scale (Sections 1.1.1 and 1.1.2) and the objectives of the MPA (Section 1.1.4). Despite the inherent value of creating monitoring protocols for MPAs, the majority of MPAs fall short of implementing monitoring protocols because managers often assume protection will produce a positive biological response and because monitoring can be costly. The majority of available empirical data and evaluations of MPA efficacy represent single studies utilizing the management action as a treatment. Effective monitoring must consider the intersection of the conservation objectives set out for the MPA and the feasibility of actually detecting a biological response. Using the global protected areas database (IUCN 2015) and relevant peer-reviewed literature, we review and compare monitoring approaches proposed for the NL MPAs (established and proposed) to global MPAs with similar objectives and spatial scales, and highlight potential research opportunities presented by the Laurentian Channel AOI.

Section 3.1 MPA design and monitoring

MPA design approaches vary nearly as much as the number of MPAs, reflecting variability in marine systems, political motivations, and conservation needs in specific applications. Size offers a suitable proxy for MPA objectives, political motivations, and monitoring strategies (reviewed in Section 1.1.1 and Figure 1). Using the protected areas database as a start point, we divided global marine protected areas into three coarse groupings based on their objectives: Fisheries production, targeted species and conservation based MPAs. From these coarse groupings, we selected MPAs for comparison with NL MPAs and the Laurentian Channel AOI based on similarities in the objectives, scale, and focal organisms. The general paucity of available literature and documentation of monitoring programs forced us to build this comparison from both government documentation and peer-reviewed literature.

Section 3.1.1 Small Fisheries production MPAs (Eastport)

Here we present three case studies of protected areas with similar objectives to Eastport but with different approaches to monitoring and gauging success. We chose these examples based on similarities in size ($<5 \text{ km}^2$), objective (fisheries production), and environment (temperate marine).

Case Study 1: Scallop closure (Isle of Man)

Description: The Isle of Man Fisheries Closure encompasses an area of $\sim 2 \text{ km}^2$ off the southwest coast of the Isle of Man and bans all commercial scallop-dredging activity. The fisheries closure was established to address the impacts of fishing activity on scallop (*Pecten maximus*) population parameters (see Section 3.3 for similar analyses) as well as a potential tool for stock enhancement for the adjacent Bradda inshore fishing ground (Beukers-Stewart et al. 2005). The relatively small ($<5 \text{ km}^2$), temperate environment close to shore mirrors Eastport in many respects.

Research and monitoring: The closure was initially intended to serve as a cultivation and stock enhancement tool. Monitoring studies within the conservation area have focused on evaluating the impact of the MPA on the scallop populations (control-impact design; Section 3.2.1), as well as changes in benthic biogenic habitat, community structure, and larval dispersal. Within 5 years of closure, habitat conditions had clearly improved within the closure, with significant increases in freestanding bryozoans and hydroids and declines in encrusting algae (Bradshaw et al. 2001). Correspondingly, scallop clearly increased in abundance and density within the MPA (Beukers-Stewart et al. 2005) over nearly two decades (Brown 2013). Although MPA species composition (Simpsons Index and Shannon-Weiner; Table 6) varied slightly over time, the differences were statistically non-significant and largely driven by relative changes in scallop abundance. Since establishment of the MPA, the community has shifted towards a scallop-dominated benthic community (Brown 2013) not observed outside the closure.

Measuring and monitoring spillover from this closure has proved difficult. Initial indications suggested that small increases in abundance within the closure produced surplus production, which contributed relatively little to the

surrounding area (Wilson 1994). However, surveys of fishermen and landings data indicated increasing catch rates, with generally greater catches immediately adjacent to the closed area (Beukers-Stewart et al. 2005). Particle tracking simulations predicted dispersal scales far beyond that of the MPA and broad connectivity around the Isle of Man. Simulations also suggested high spatial-temporal variability in dispersal, ranging 10-100 km within and among simulated spawning years. While these simulations suggested some self-recruitment, hind-casting models indicated that a significant portion of recruitment to the potential monitoring area came from locations outside the local fishing grounds and beyond the jurisdiction of the local government (Neill & Kaiser 2008). Although higher settlement densities in the MPA were linked definitively to improved habitat conditions (Bradshaw et al. 2003), larval recruits within the closure are likely originated outside the closed area. Attempts to transplant genetically distinct sub-populations of *P. maximus* into the MPA in order to evaluate potential dispersal scales of larvae originating within the MPA produced mixed results in preliminary work, suggesting highly localized recruitment (< 500m; Sweeting & Polunin 2005). However, pre-existing genetic variation in the population might cloud this interpretation (Hold et al. 2010). The variable conclusions drawn from different methods to elucidate the potential dispersal scale of protected scallops speaks to the difficulty of objectively measuring larval export in temperate marine systems (Laurel & Bradbury 2006).

Despite the dramatic increase in density of spawning scallops within the fisheries closure (Beukers-Stewart et al. 2005), there is no data available to evaluate the relative contribution of this increased spawning stock to overall larval production. While this information is unnecessary to evaluate the success of the MPA in providing increased larval production, which it clearly does, it could provide a template to formalize the importance of the MPA to scallop stocks. Total egg production offers an unambiguous metric that provides essential context to the MPA, despite lack of knowledge on source-sink dynamics of the system.

Case Study 2: Skagerrak Experimental Enclosures (Norway)

Description: Fisheries returns for European lobster (*Homarus gammarus*) have declined over the past two decades. Despite active management and monitoring of commercial fishing, the contribution of recreational fishing to observed declines was unknown. This knowledge gap was particularly serious because stock assessments and management decisions were based historically on commercial landings only. Reviews of commercial landings, recreational fishing log-books, and fishermen surveys revealed chronically under-reported fishery-induced mortality and that declining coastal stocks would require additional management intervention (Kleiven 2010).

The Skagerrack Experimental Fisheries Closures were designed to test how protected area management could fit within an integrated coastal management framework on the Norwegian coast. In particular the closures were established to provide context on how lobster would develop in the absence of fishing mortality (Moland et al. 2013a). The fisheries closures are located on the Skagerrack coast and divided into three separate management areas with a summed coverage of 2.2 km² (Kleiven 2010). Like Eastport, these MPAs are located close to shore (> 1 km) in a temperate environment and divided into multiple units within a complex coastal coastline.

Research and monitoring: The Skagerrack Experimental Closures were designed from the onset as an experimental tool to evaluate the effects of fishing on population parameters (e.g. abundance; also see Section 3.3 for similar indices). Studies that seek to determine the influence of fishing on marine biota generally utilize a similar ‘experimental design’, treating MPAs as a manipulation of the natural, fished environment. Ideally fishing closure studies evaluate both the change in the state of the protected relative to unprotected populations, and the magnitude of surplus spillover from these changes. In addition, the behavioural biology of the focal species can provide a context to assess how movement fits the scale of protection offered by the MPA, and thus how much, if any, uncontrolled mortality is expected (see Section 1.1.1 for review of scaling).

Behavioural analysis of the lobsters showed that, despite the relatively small size of the MPAs (reviewed in Figure 2), their size was appropriate given the movement range of acoustically tagged lobster. Over the course of a year the majority of lobsters remained within the reserve > 95% of the time, with a median home range less than 1 km² (Moland et al. 2011a). The movement of lobsters also varied seasonally in that lobsters that left the MPA generally did so in the winter when migrating to deeper waters. Limited fishing activity during the seasonal migration suggested limited mortality resulting from straying from the reserve (Moland et al. 2011b). Not surprisingly, mark-recapture data from the Kávra MPA (part of Skagerrak Experimental Closures) indicated that lobsters released within the reserve were generally recaptured within the reserve. Multiple-recaptures also indicated decreased mortality with time because lobsters were protected from fishing mortality and grew to larger sizes (Moland et al. 2013b). Linear increases in density within the MPA over time showed no signs of slowing down during the first two decades of protection (Moland et al. 2013b). Furthermore, before-after-controls-impact (BACI) analyses (reviewed in Section 3.2.1) of catch data revealed significant increases in average size (+ 35%) and CPUE (+ 245%) compared to similar unprotected areas (Moland et al. 2013a). These large increases occurred relative to controls areas sited within 1 km of the MPA boundaries.

Like Eastport, managers anticipate measurable benefits from small spatial closures for surrounding fishery. The BACI design demonstrated that increases within the reserve were greater than those outside over a similar period, but could not attribute observed increases in CPUE at the (unprotected) reference sites (+ 87%) to spillover from the reserve. A second set of control sites further away from the MPA would enable an expanded BACI model that could determine whether control sites within closer proximity to the reserve exhibit larger increases than those farther afield (see Section 3.2.1, Code 5, and Case Study 3 for examples). The study also provided evidence of decreasing lobster abundance with decreasing proximity to the MPA, generally declining up to 7 km from the reserve. However, they provided no evidence whether habitat or actual spillover from the reserve mediated this increase, as other studies suggest (e.g. Kay et al. 2012).

Changes in management in unprotected areas can also limit the potential to detect an effect through monitoring. For example, the weakest ‘reserve effect’ of the Skagerrak Experimental Closures occurred at the Bolærne MPA, which researchers partly attributed to local changes in fisheries management. In this area, an increase in minimum legal size during the monitoring period increased the density of lobsters remaining the year following in the control area, thus diminishing the relative difference between protected and unprotected areas (Moland et al. 2013a). Examination of any long-term data should consider any changes in management outside the MPA when evaluating MPA performance. When management seeks policy approaches to increase egg production (e.g. Ennis et al. 2003) they should consider changes in minimum legal size or effort when comparing long-term continuous data (see Section 3.2.6 for monitoring design considerations).

Although managers often assume adult spillover benefits from protected areas at some level, the real power of small MPAs such as the Skagerrak Experimental Closure and Eastport, is their ability to increase egg production by protecting species with small movement scales. Behavioural analysis of lobsters in the Skagerrak Experimental Closures, for example, suggested that lobsters moved little and lived the majority of their lives within the MPA (Moland et al. 2011a, Moland et al. 2011b, Moland et al. 2013b). However, surplus production in Skagerrak has not been linked to increased recruitment, even though that pathway offers the most likely link between the positive benefits for MPA residents and any net export to the fishery. Microsatellite analysis of Norway and Sweden lobster demonstrated little evidence of isolation by distance, suggesting high gene flow and dispersal well beyond the Skagerrak area (< 21 km) (Huserbraten et al. 2013). High gene flow likely reflects dispersal distances greater than the local assessment areas, which would diffuse any signal relative to natural variability.

Case Study 3: Lundy Marine Reserve (United Kingdom)

Description: The Nature Conservancy Council (UK) established the Lundy Marine Reserve (LMR) to provide information on marine habitats, changes in community structure in the absence of fishing, and monitor ecological parameters for individual species of high conservation interest (Fowler & Pilley 1992). Levels of access vary within the

LMR, with recreational zones, general use zones, and a no-take area. Although the LMR provided some additional protection through monitoring and spatial management (Fowler & Pilley 1992), fishing activity continued to affect important habitats and species. Continued pressure from extractive fisheries motivated the zoning of a strictly no-take zone (~3.3 km²), to evaluate potential impacts of permitted activities on the Lundy ecosystem (Hoskin et al. 2009). This no-take zone (NTZ) represented the first of its kind in the United Kingdom. Like Eastport, the Lundy NTZ is a temperate nearshore environment that lacks data prior to establishment, and is monitored for lobster production.

Research and monitoring: LMR monitoring utilizes dive surveys to quantify broad temporal changes in abundance and density of associated biota. Monitoring focuses particularly on coral habitats and a population of red band fish (*Cepola rebeszens*), deemed of high scientific value. LMR monitoring within the NTZ focusses specifically on exploited decapod crustacean species, including European lobster (*H. gammarus*), velvet crab (*Necora puber*), brown crab (*Cancer pagurus*), and spider crab (*Maja squinado*).

Changes in size and abundance of the four focal crustacean species were monitored and compared using an asymmetric post-impact field evaluation. Reference sites were chosen based on similar benthic habitat, depth, and environmental characteristics. Sites established near (1-5 km) and far (20-100 km) away from the reserve provide a context to evaluate potential spillover (see Section 3.2.1, Figure 19 and Supplemental R-Code 5 for a similar design). Replicate samples randomly placed within each control location allow comparison of abundances and size distribution within and outside the MPA through time to assess the effects of the NTZ. Moreover, adult spillover would presumably affect sites closest to the MPA (near controls) over time more rapidly than far-field sites (Hoskin et al. 2009, Hoskin et al. 2011).

Of the four commercial species sampled only *H. gammarus* and *N. puber* changed consistently over the monitoring period. Over the observational period, abundances of *H. gammarus* increased 127% whereas reference sites remained unchanged. Five years after MPA designation, lobsters within the NTZ were five times more abundant

compared to outside (Hoskin et al. 2011). Although lobster abundance in the near- or far-control sites showed no significant change, size-disaggregated analysis demonstrated significantly more sub-legal lobsters in the near-control sites over time compared to the far-control sites where no trend was evident. Increased abundance of sub-legals in the near-control sites suggests potential spillover of new recruits to fished areas. Abundances of velvet crab decreased over time by approximately 65% within the NTZ compared to no change in the control locations. The isolated decreases in velvet crab within the NTZ, concomitant with increases in lobsters, suggest a potential shift in predation and trophic structure as an indirect effect of reducing fishing mortality on lobster.

Although higher densities of larger and more fecund lobsters in the NTZ presumably increase reproductive potential, the measured spillover cannot be attributed to increased recruitment. First, the dispersal potential of larval lobsters will likely exceed the 1-5 km distance between the reserve and control sites, especially in exposed areas such as Lundy Island. If larval dispersal distances exceed the area where increased recruitment occurs, then the two metrics cannot be related. Second, even the most optimistic estimates of development time suggest a 7-10 year period to reach recruitment size; it is therefore unlikely that newly-settled NTZ progeny produce the higher abundances of sub-legal lobsters, monitored for just 5 years (2003:2007). Changes in community and food-web dynamics offer a more likely cause of sub-legal lobsters spilling over. All crab species declined in the MPA, and velvet crab especially. Given these crab species likely compete with smaller sub-legal lobsters for resources, the MPA could support higher survival of new recruits that eventually spillover into the surrounding areas (Hoskin et al. 2011).

Application for the Eastport MPA

Monitoring the potential impact of the spawning aggregation to surrounding fished areas requires an operational physical or biophysical model. Like Eastport, surplus production and recruitment likely represent the most important contribution of the fisheries closures in the Isle of Man, Skagerrak and Lundy. Although it is possible to estimate surplus production (e.g. Kelly et al. 2002, Ennis et al. 2003), the full evaluation of the MPA relative to its objectives requires some measure of connectivity. An operational biophysical model can estimate connectivity and dispersal trajectories.

For all case studies presented in this section (3.1.1) the lack of any definitive dispersal model adds significant ambiguity as to where recruitment increases and whether it could be detected. Biophysical models can provide this information and, if required, enable establishment of an effective monitoring framework (i.e. reference sites set at the appropriate scale and location).

In addition to knowing the scale at which an effect might be detected, the level of surplus production created must be considered relative to all other sources of reproduction. Preliminary measurements in Eastport suggest that the MPA, which encompasses only 1.8% of available habitat, produces 10% of the production in the Eastport Lobster Management Area (EPLMA) (Ennis et al. 2003). Annual updates will provide a relative index of Eastport MPA performance in increasing egg production relative to unprotected areas. Control-impact analyses could determine whether egg production in the MPA is significantly higher than in fished control sites, and whether or not the difference changes through time. This information provides a useful yardstick to evaluate reproductive performance, but without understanding recruitment kernels (probability of recruitment vs. distance) and dispersal trajectories, there is no way to assign the effect of this surplus production.

Hindcasting biophysical models can also predict likely sources of recruits to an MPA. This knowledge is particularly important in evaluating the sustainability of the MPA. Given the relatively small size of the Eastport MPA and other case studies in this section, it is highly unlikely that the MPAs are self-recruiting. Exploitation outside the protection that leads to recruitment limitation will reduce the probability of sustaining positive trajectories within the MPAs. As shown for the plaice box below (case study 10; Pastoors et al. 2000), recruitment limitation can mitigate any positive benefit afforded by the MPA.

To date, Eastport monitoring has focussed primarily on American lobster. However, understanding changes in the trophic structure within the MPA could provide new indicators of how the MPA influences the protected population. As shown in the Lundy reserve (case study 3), changes in food-web dynamics likely reduced pre-recruitment mortality

in European lobster. Reduced pre-recruitment mortality in Lundy resulted in measurable spillover of sub-legal lobsters from the MPA. Just as increased abundance of adults can indicate MPA success, so could relative abundance and spillover of juvenile lobster.

Section 3.1.2 Targeted MPAs (Gilbert Bay)

Here we present three case studies of protected areas with conservation-based foci similar to Gilbert Bay, but with different approaches to monitoring and gauging success. We chose these examples primarily based on the targeted objective to protect and provide recovery to populations of threatened or valuable species.

Case Study 4: Poor Knights Island Marine Reserve (New Zealand)

Description: The Poor Knights Island Marine Reserve (PKIMR) on the northeast coast of New Zealand covers an area of approximately 24 km². The marine reserve had a mixed history of exploitation prior to fully no-take designation in 1998 (Sim-Smith & Kelly 2009). The PKIMR supports a diverse and unique assemblage of marine flora and fauna driven by a confluence of factors including tropical circulation-connectivity, steep rocky topography of the islands, saline water conditions and extremely high water clarity. The PKIMR is part of the New Zealand Government's directive to protect biodiversity and unique marine ecosystems (De Buisson 2009). The conservation objectives of the PKIMR are to protect biodiversity and also potentially supplement fished areas by increasing regional production.

Like Gilbert Bay, the intermediate-sized (10-100 km) marine reserve encompasses nearshore coastal environment. In contrast to the single species primary focus of Gilbert Bay, the PKIMR protects several threatened species, but nonetheless provides a useful comparison. In particular, much sampling effort has focused on Australasian snapper (*Pagrus auratus*) which was heavily exploited in the area prior to cessation of all fishing activity in 1998 (De Buisson 2009). Like the Gilbert Bay golden cod, seasonal movement of snapper mediates the protection offered by the reserve and influences how the population should be monitored.

Research and Monitoring: The primary objective of PKIMR is to protect the unique assemblage and high biodiversity of marine biota. Surveys demonstrate consistently higher coral and sponge diversity and density in PKIMR relative to reference areas. Higher coral and sponge coverage presumably support higher fish diversity. Yearly monitoring of reef fish assemblages through baited camera surveys and dive transects provides an index of the biological status of the reserve through time, and show consistently higher fish diversity and abundance within the reserve than within reference areas (De Buisson 2009). Complementary sampling of zooplankton and sedimentary invertebrate communities shows significantly higher sedimentary invertebrate diversity in the reserve than outside, but no difference in zooplankton diversity. Zooplankton abundance, nonetheless, varied spatially with significantly higher zooplankton abundances 1 km from the reserve than within reserve boundaries, suggesting increased abundances of planktivorous fish, and thus an indirect indicator of reserve performance in protecting reef fish species (Sim-Smith & Kelly 2009).

Much of the research published for the PKIMR focuses on the recovery of snapper (Sim-Smith & Kelly 2009). Comparisons of snapper abundance, pre- and post-designation, revealed rapid and sustained recolonization within the PKIMR, that external factors could not explain (see BACI design; Section 3.2.1). After three years of no-take designation, density within the reserve exceeded that in control locations by 8.3 times on average (Denny et al. 2004). As in Gilbert Bay (e.g. Morris et al. 2014) home range analysis determined that despite some migrants, the PKIMR is sufficiently large to protect snapper for the majority of their juvenile and early adult phases (Parsons et al. 2003). Although a robust BACI design aided monitoring the influence of protection on snapper population (reviewed in Section 3.2.1), tagging revealed a potential seasonal bias in sampling. Sampling during the summer migration period, for example, would likely inflate the degree of protection afforded by the reserve, because the PKIMR population would include both resident and migrating-transient fish. Surveys in the spring, prior to the beginning of migration, would likely optimize sampling of the reserve population, and thus provide the most suitable data for a monitoring program (Willis 2013).

The unique environmental conditions and biodiversity in the area also support the only known large spawning aggregation of short-tailed stingray, *Dasyatis brevicaudata* (Le Port et al. 2012). The PKIMR protects only the mating and early juvenile periods, however, reduced mortality during this critical and vulnerable period will benefit the population overall. Stingrays are now advocated as a conservation objective and monitoring focus for the PKIMR (Le Port et al. 2012), illustrating how adaptive management strategies can increase MPA efficacy.

Case Study 5: Exuma Cays Land and Sea Park (Bahamas)

Description: The Exuma Cays Land and Sea Park (ECLSP) surrounds the Exuma Cays island chain in the central Bahamas. The total 442 km² park, including land (Stoner & Ray 1996, Chiappone & Sealey 2000), was established as part of the government of the Bahamas initiative to preserve the natural heritage of the Bahamas. A monitoring program implemented to collect long-term data on the status of biota within the ECLSP provides managers an opportunity to assess the current status of focal species and the ecosystem as a whole. The monitoring program utilizes key biological indicators including focal species abundance, focal species size structure, habitat distribution and complexity, and community composition (Dahlgren 2009).

Monitoring at the ECLSP focuses on several key focal species that were heavily exploited in the area in the past, including the Nassau grouper (*Epinephelus straitus*), spiny lobster (*Panulirus argus*) and the queen conch (*Strombus gigas*) (Chiappone & Sealey 2000). Monitoring has focused on queen conch in particular following dramatic declines in abundance from overfishing both locally in the ECLSP and worldwide (Stoner & Ray 1996). Despite declining abundance, queen conch remains an important and lucrative fishery in the Bahamas. Like Gilbert Bay, the ECLSP monitoring program focusses on biological indicators for key species within the reserve and thus provides a suitable comparison. The reserves offer significant protection to both queen conch and golden cod, respectively, and maintain demographic trajectories that differ from fished areas.

Research and monitoring: ECLSP monitoring divides equally between monitoring the areal coverage and health of different habitat types (soft bottom, coral, and eel grass) and the status of the focal species and fish assemblages. Because ECLSP managers lack pre-establishment data, they estimate current status and trajectories relative to historical baselines. For example, comparisons between surveys in 2003 and 2007 revealed that despite relatively stable fish assemblages in the ECLSP, habitat quality decreased overall, particularly in coral reef coverage, consistent with regional observations (Dahlgren 2009). Although this monitoring program lacks the ability to objectively distinguish the effect of protection relative to prior condition (e.g. BACI design, reviewed in Section 3.2.1), it nonetheless documented how the ecological health and status of the reserve changed through time relative to data collected using a common survey type over a set time-lag.

The monitoring program emphasizes maintenance of the ecological integrity of the ECLSP. In this sense, significant changes in the fish community would be viewed as a negative indicator. Despite no documented significant changes in fish communities from 2003 to 2007 (Dahlgren 2009), several observed changes in the ECLSP ecosystem could portend significant future change. Habitat monitoring suggests an apparent shift in reefs from *Montastraea* spp. to *Porites* spp dominated communities which, if established over the whole ECLSP, would reduce reef formation and structure. Changes in habitat provided by coral species could lead to significant changes in the fish community. In addition, the expansion of invasive Indo-Pacific lionfish (*Pterois volitans*) threatens the integrity of ECLSP fish assemblages through food competition and direct predation. The success of the ECLSP requires continued monitoring of invasive species and habitat type to both detect and predict change within the reserve. The conservation objectives for focal species relate directly to changes in the fish community, and any such changes therefore provide context for expected biological responses in focal species (e.g. change in grouper abundance) to protection.

Monitoring of queen conch within the reserve focusses on both measuring change over time within the reserve and among protected and unprotected areas. Overall monitoring suggests that the reserve provides significant protection

to conch, leading to increased density and greater reproductive success than unprotected areas (Stoner & Ray 1996, Stoner et al. 2012a). Despite consistently higher abundance within the reserve, the queen conch reserve population has declined since monitoring began. Larval connectivity studies suggest higher densities of larval conch in the reserve, however, advection likely transports most larvae to fished areas (Stoner et al. 2012b). The negative relationship between fishing effort and conch production points to a risk of recruitment limitation in the reserve. Indeed, large declines of abundance in recent years inside the reserve suggest that while the ECLSP protects adults from fishing mortality (Stoner et al. 2012b), concomitant declines in production in fished areas influence recruitment and stability of the protected populations (Stoner et al. 2012a). Interpretation of monitoring indices (e.g. abundance) and development of adaptive management to optimize protection will require continued monitoring of the production and recruitment dynamics of this focal species (Chiappone & Sealey 2000, Stoner et al. 2012a).

Case Study 6: The Marine Biological Reserve of Arvoredo (Brazil)

Description: The Marine Biological Reserve of Arvoredo (MBRA) was established in 1990 to protect significant marine biodiversity and coastal ecosystems in southern Brazil. The conservation objectives for the ~170 km² no-take MBRA focus primarily on protecting a representative area of high biological diversity. Additionally the reserve has strong potential to help rebuild exploited stocks of mollusks, crustaceans, and fish (Bouzon & Freire 2007, Martins et al. 2014).

Although the MPA has been established for 15 years, monitoring data and availability of comparative studies are limited. In the past decade much work has focussed on acquiring baseline data of the local biota in the reserve, including crustaceans communities (Bouzon & Freire 2007) and coral-algal habitats (Gherardi 2004). Like Gilbert Bay, the MBRA offers potential protection to several species of large-bodied marine predators. Recent monitoring has focussed on evaluating the efficacy of the MPA in protecting these predatory species; monitoring protocols for the MBRA therefore offer a useful comparison for Gilbert Bay golden cod.

Research and monitoring: MBRA ichthyofauna encompass three main categories according to specific habitat preferences, including rocky reef, flat bottom and pelagic-dwelling fish (Godoy et al. 2006). Initial monitoring focussed on optimizing sampling to census these categories of fish. Comparisons of sampling methods showed that underwater visual surveys (UVS) were more effective and pragmatic than trawling to sample species richness and abundance across all habitat types. In particular, UVS methods enumerated significantly more fish over rocky reef habitat than the same unit effort using trawl gear (Godoy et al. 2006).

The recovery of predatory fish species is of specific interest given recent declines in southern Brazil. Increased predatory fish indicate more complete and robust trophic structure, and thus a healthier ecosystem (Anderson et al. 2014). UVS data were collected for focal species of grouper across all habitat sites both within and outside the MPA. Control-impact analyses revealed significantly higher biomass and average size of several species of grouper in the MBRA than in control sites to the north and south along the Brazilian coastline. Although monitoring focusses primarily on processes and biological attributes within the MBRA, the concomitant analysis of reference sites provided a context to evaluate change associated with natural variability external to protection. The biological response in threatened species of grouper (e.g. *Epinephelus marginatus*) to protection was particularly strong. However, when gauging the overall efficiency of the reserve, despite biological responses several orders of magnitude higher within the reserve compared to reference sites, the observed biomass and size distributions were still well below historical baselines. This result was surprising given the length of time (~15 years) since no-take status was designated and the life cycle and growth rate of the species in question. Researchers attributed the slower than expected response to uncontrolled mortality through illegal fishing activity. To achieve recovery to historical population levels, Anderson et al. (2014) suggested managers focus on reducing all illegal fishing activity and thus limiting mortality. Like Gilbert Bay, uncontrolled mortality, albeit small relative to the mortality offset by protection, hindered the capacity of the protected area to meet its conservation objectives.

Application to the Gilbert Bay MPA

Focussed monitoring on a single species directly informs managers on whether protection is achieving the management objective. In Gilbert Bay and the case studies outlined in this section, monitoring focussed on a single species provided pragmatic indicators regarding conservation objectives for that species. However, a narrow monitoring focus may limit the capacity of a monitoring program to identify the cause of poorer than expected results. All of the sample case studies (3-6) dedicated a portion of monitoring effort to evaluating habitat quality. Changes in habitat can relate directly to declines in the focal species or to changes in the food web. Changes in trophic structure indicators illustrate how protection can influence an ecosystem and predict how focal species respond. Information about changes in focal species and associated habitat and community can help in developing targeted interventions. For example, if reductions in eel grass limit post-settlement survival then active intervention in the form of eel grass restoration could help meet the conservation objective. Similarly, researchers suggested that actively repairing habitat damaged by trawling (trawl scars) could help rebuild damaged systems at the Darwin mounds (case study 7; Van Dover et al. 2014). Broadening the focus of a monitoring program can provide important contextual information about changes within the reserve, and enable a more comprehensive evaluation of why management action is failing or succeeding in to achieving its conservation objectives.

Dynamic environmental conditions can increase natural variability in a study system, and thus introduce bias in monitoring data that is external to the management action. A variety of approaches can partition reserve effects from natural variability (reviewed in section 3.2.1), but all require sampling both inside and outside the MPA. For example, natural variability can be partitioned by comparing the relative temporal change in protected versus replicate control sites. In Gilbert Bay, extending monitoring to reference sites outside the MPA would aid in evaluating changes resulting from uncontrolled mortality (Morris et al. 2014) as well as changes associated with larger-scale environmental processes. Determining how much of the observed pattern (e.g. change in abundance) can be attributed to a management action is difficult without some sampling external to the MPA.

Managers of Gilbert Bay and all case studies in this section use changes in the abundance of a focal species as an index of MPA performance. Whether change occurs through time within the reserve or relative to exterior reference sites, the level of positive change generally gauges MPA success. However, the benchmark by which managers assess the positive change can change interpretation of monitoring data. For example, increases in predatory fish within the MBRA suggested success in meeting conservation objectives, but this change in abundance still fell well below the capacity of the system, given previous reported landings (Anderson et al. 2014). Similarly, a past review of monitoring protocols for Gilbert Bay noted the lack of reference levels as a limiting factor in gauging success from the monitoring program (DFO 2010c). Development of a historical baseline or potential carrying capacity for Gilbert Bay would provide a yardstick to predict population potential, and determine the point at which any positive trend in abundance be expected to peak.

Much of the baseline research on Gilbert Bay golden cod provides essential information about the potential effectiveness of the MPA. Indeed, few examples of MPAs exist that have developed such a complete research portfolio on the spatial scale of movement for larval juvenile and adult life history stages. In particular, the recent finding that migrants move beyond the boundaries of the MPA provides essential context to understand recent declines in abundance resulting from mortality outside the MPA (Morris et al. 2014). Monitoring programs must evaluate the timing and spatial extent of movement of focal species when developing monitoring programs. The validity of long-term datasets also assumes stable catchability (Katsanevakis et al. 2012). Variation in the timing of surveys when the catchability of the target varies seasonally (i.e. snapper in the PKIMR; Willis 2013) can create issues when extrapolating data across multiple years. For Gilbert Bay for example, the suitability of long-term trends drawn from research CPUE (DFO 2010c) requires sampling during time periods of consistent probability of catching a size or age-class of cod. Sampling conducted during times when cod might migrate out of the research area (Morris et al. 2014) could underestimate the influence of protection and thus bias MPA evaluation. While this concern has not been a problem in Gilbert Bay

monitoring, it nonetheless illustrates the importance of maintaining consistency when sampling in light of known movement patterns of the focal species.

For MPAs with single species objectives, it is possible that monitoring programs can become more narrowly focused, especially with limited resources. However, under such specific monitoring conditions, it is important to maintain a research component in addition to standard monitoring. Research conducted in parallel to monitoring programs can evaluate potential change in habitat or community structure that would otherwise go unnoticed. A comprehensive research program can utilize the protected area and associated monitoring to facilitate investigation of species attributes applicable beyond the evaluation of the MPA (see Section 3.3). Synergy among monitoring and research activities can lead to a better interpretation and explanation of changes observed based on monitoring.

Section 3.1.3 Conservation based MPAs (Laurentian Channel AOI)

Here we present four case studies of protected areas with similar designs and objectives to the proposed Laurentian Channel AOI. Given the scale of the proposed Laurentian Channel AOI, it is unlikely that any one program would provide a suitable template for monitoring. We instead chose examples to reflect a diversity of approaches that have been used to protect and subsequently monitor biodiversity over large scales.

Case Study 7: Darwin mounds (United Kingdom)

Description: The 1380 km² Darwin Mounds MPA off the west coast of Scotland prohibits all bottom trawling. A series of seamounts characterize the area and create a unique “tail-like” formation that supports a high diversity of suspension feeders and infauna (De Santo & Jones 2007, Serpetti et al. 2013). Some infaunal species, such as the large and fragile xenophyophore (*Syringammina fragilissima*) are quite rare. In addition, the area surrounding the mounds supports a unique formation of cold-water corals *Lophelia pertusa* growing on sand, as opposed to hard-substratum where they usually occur (Serpetti et al. 2013).

A system of vessel monitoring and global positioning systems monitor fishing activity around the MPA (De Santo & Jones 2007), where bottom depths up to 1000 m force bottom trawlers to utilize relatively long head-lines (~ 2 km). Head-line length presents a risk to the MPA in that a ship could be outside the trawl exclusion zone but the trawl itself could drift towards the protected habitat and thus cause damage. To mitigate the risk of incidental trawl drift, the MPA was designed with a buffer zone of 2.2 km around all seamounts (Sweeting & Polunin 2005). Boundaries closer to the edge of the protected habitat would likely lead to MPA failure.

Like the Laurentian Channel AOI, the Darwin Mounds MPA was designed to protect biodiversity in the area, encompass a range of depths, and protect a relatively large ($> 1000 \text{ km}^2$) area compared to most MPAs (Figures 1-2). Similarity in design and objectives make the Darwin Mounds a particularly suitable comparison with the Laurentian Channel AOI.

Research and monitoring: The objective of the Darwin Mounds MPA is to protect an area of unique habitat and relatively high diversity from negative effects associated with bottom trawling (Huvenne 2011). Benthic surveys 11 years after MPA establishment showed that while the MPA protects coral, areas remain where coral demonstrated little signs of recovery (Huvenne 2011). Benthic macrofaunal abundance varied among species and size classes, with no clear increase or decrease in abundance inside relative to outside the MPA. Somewhat surprisingly, biodiversity values (Shannon's diversity and Simpson's dominance) estimated from sediment samples were higher outside than inside the MPA. While this result contradicts previous work in the MPA, differences in mesh sizes when sieving sediment cores likely contributed to the finding (Serpetti et al. 2013). The smaller mesh size (250 μm) employed by Serpetti et al. (2013) captured more of the small meiofaunal species (i.e., nematodes) and thus the functional composition of the sample. Trawling activity typically damages fragile organisms which, in turn, attracts a suite of opportunistic polychaete and nematode detritivores that elevate total biodiversity (Serpetti et al. 2013). Ongoing recovery of biogenic habitat

unique to the Darwin Mounds (Huvenne 2011), challenges a complete evaluation of the efficacy of protection in meeting conservation objectives, based on sedimentary biodiversity.

The complex biogenic habitat provided by *L. pertusa* also facilitates higher abundances and diversity of demersal fish species on the reefs. Over 90% of species present and 80% of the total abundance of fish in the survey area associated directly with the reef habitat compared to sandy bottom (ICES 2004). The conservation objectives of the Darwin mounds trawling closure are to *conserve* and *provide for recovery* of the unique coralline habitat associated with the area. Evidence from video transects suggest that management actions for the MPA (e.g. VTS and buffer zone) are generally effective in preventing damage from fishing activity (Huvenne 2011). However, simply removing the threat might not be enough to ensure the MPA meets the objective of recovery within a reasonable time frame, given that generation times of *L. pertusa* approach 100 years (Van Dover et al. 2014). Indeed active intervention might be the only recourse to recover this unique area. Geoengineering or rebuilding raised seabed habitat damaged through trawling (e.g. trawl scars) and transplanting laboratory grown fragments of *L. pertusa* could provide an active recovery strategy over a reasonable (~ 10 yr) timeframe. Van Dover et al. (2014) costed such a restoration plan at ~4.5 million US dollars, and argued that the tremendous ecological and heuristic value of these habitats would offset these costs, and provide benefits into the future. However, the probability of successfully applying active remediation in to deep-sea ecosystems remains largely untested.

Case Study 8: Gray's Reef National Marine Sanctuary (United States)

Description: The ~ 58 km² Gray's Reef National Marine Sanctuary (GRNMS) is located 28 km offshore of Sapelo Island, Georgia. The GRNMS protects a high diversity of coral and rocky habitat which, in turn, supports a diverse assemblage of fish and invertebrate species, as well as migratory sea turtles and whales. This healthy live bottom habitat in the GRNMS supports productive communities at all trophic levels. The GRNMS is one of the best examples of southeastern living bottom habitat.

The objectives of the GRNMS split into three themes (NOAA 2006):

- 1) Improve the conservation, understanding, management, and wise sustainable use of marine resources.
- 2) Enhance public awareness, understanding, and appreciation of marine environment.
- 3) Maintain for future generations the habitat and ecological services, of the natural assemblage of living resources that inhabit these areas.

The multi-use GRNMS designation allows various levels of human activity and fishing. However, the sanctuary prohibits any activity causing habitat damage (i.e. trawling, benthic sampling devices, oil extraction, or discharging deleterious material). Despite various levels of protection zones within the sanctuary, regulations collectively afford a much higher level of protection than in areas surrounding the GRNMS (NOAA 2006).

Like the Laurentian Channel AOI the conservation objectives of the GRNMS focus on maintaining marine habitats and their associated biota. Although considerably smaller than the Laurentian AOI, the GRNMS provides a useful comparison for monitoring programs given similarities in conservation priorities and prior history of exploitation. In particular, the GRNMS environmental monitoring programs could prove useful for monitoring the status of the Laurentian AOI and temporal changes within.

Research and monitoring: Since its inception, the GRNMS has fostered a diversity of studies evaluating the biotic and abiotic conditions of a protected reef habitat. Compared to other MPAs, considerable peer-reviewed and government documents detail the processes and status of the reef (NOAA 2015b).

The GRNMS monitoring program utilizes four main research monitoring strategies that feed into the overarching conservation objectives (NOAA 2006). Here we review these strategies for monitoring and also draw on published results as needed.

Strategy 1 “Investigate ecosystem processes”

To successfully manage the GRNMS for future generations, the management plan emphasizes understanding and evaluating ecosystem function. In particular, MPA monitoring and research seeks to identify the “*processes, species or relationships that are most critical for sustaining ecosystem function*” (NOAA 2006). To accomplish this goal, NOAA has amassed a long-term dataset to provide baseline information on key ecosystem processes. Three primary research activities and associated indicators populate this dataset: 1) Develop a trophic model of the sanctuary (e.g. Ecospace; Walters et al. 1999); 2) Evaluate ecosystem components (e.g. species abundance) that influence trophic dynamics; 3) Evaluate invertebrate recruitment dynamics (e.g. coral). This information provides managers a framework to assess changes within the system compared to those outside, and thus evaluate the status of the GRNMS and management efficacy. For example, researchers cleared plots inside and outside the protected area to evaluate how long it takes to colonize and establish living substrate, as well as the successional changes associated with reef development in protected and unprotected systems. Understanding the overall recruitment-succession process provides key information about the functional role each species plays in reef recovery, and thereby provides information about which species could be transplanted to maximize recovery in damaged systems (NOAA 2006).

Strategy 2 “Marine Research Area”

As a multi-use protected area with various levels of permitted activity, the GRNMS supports exploration and the objective to “*enhance public awareness, understanding and appreciation of marine environment*”. While the GRNMS provides greater overall protection than outside areas, some human-induced influences (e.g. diving and artisanal fishing) could, nonetheless, mitigate MPA objectives. Managers designated a ‘research area’ as a control site to evaluate the potential impacts of human disturbance within the GRNMS, similar to the strictly no-take area within the Lundy Marine Reserve (Section 3.1.1 Case Study 3).

Survey mapping of fish (e.g. Munoz et al. 2010) and coral assemblages (e.g. Freeman et al. 2007) within the GRNMS identified representative areas (see Section 1.1.3 for similar approaches) *a priori* to establishment of the

research area. In 2011, managers set aside the southern third (~ 20 km²) of the GRNMS as no-take and used the area to evaluate the influence of protection on ecosystem variables such as habitat, community composition and fish abundance (Kendall et al. 2008). Results from this designated research area will influence future changes in sanctuary use.

Strategy 3 “Assess and characterize sanctuary resources”

To fully assess ecosystem changes (Strategy 1) and monitor the impact of permissible activities (Strategy 2) requires a thorough characterization of habitat and associated biota. Spatial surveys provide the baseline against which to assess change and evaluate scientific understanding of physical and ecological interactions/dependencies within the GRNMS (NOAA 2006). Detailed side-scan imaging and multibeam bathymetry of the sanctuary provides a physical layer for a scientific GIS database. All georeferenced and descriptive data collected in the sanctuary, including those from non-monitoring studies (e.g. Freeman et al. 2007), will be added to this database (NOAA 2006), for example, yearly surveys of fish abundance and habitat coverage updates. Detailed identification guides ensure that smaller, often conspicuous, but nonetheless important species receive equal monitoring attention. Habitat surveys cover all habitat types including dominant coral as well as soft-bottom areas. Analysis of sedimentary communities, such as those in the Darwin mounds (Case study 7), and associated abiotic variables (sediment size, dissolved O₂, and total organic carbon) are conducted in addition to coral diver surveys (Balthis et al. 2007). Collectively, this monitoring approach and integrative database, provides managers with a multifaceted and synoptic spatial description of the GRNMS.

Strategy 4 “Monitor the status of fish health”

“Fish health” in this context represents a catch all term for the fish community, determined by monitoring changes through time. As discussed in Strategy 1 and 3, the GRNMS prioritizes understanding the trophic structure of fish communities and monitoring how they change in space and time. In addition to “within-reserve” studies (e.g. Strategy 2) they also identify a need to evaluate changes in the reserve relative to unprotected areas. Control-impact

based comparisons of temporal changes in community composition and species abundance provide an indicator to gauge changes inside versus outside the reserve and account for natural variation unrelated to the management intervention.

Strategy 5 “Monitor water and sedimentary quality”

The primary objective of “*maintenance for future generations*” requires identifying threats related to human activity outside the reserve. In particular, water quality within the system and its interaction with coastal development adds concern. Monitoring of water quality (i.e. salinity, pH, and organic load) provides an indication of potential threats to the sanctuary, whereas sediment monitoring provides an indicator of effects from coastal development. For example, recent surveys reported contaminant loads within the sediment equivalent to areas outside the sanctuary (Balthis et al. 2007). The risks posed by increased coastal development, in tandem with transport of potential contaminants in prevailing currents, points to the need for continued monitoring of water and sediment quality.

Strategy 6 “Monitor large-scale oceanographic processes”

Regional scale oceanographic and climatic processes could influence biological responses within the sanctuary. Extrapolation of physical parameters in the reserve (e.g. temperature, conductivity, fluorescence, radiation and circulation) to national monitoring systems will help ensure the most accurate assessment of how larger-scale processes contribute to changes within the GRNMS. For example, an accurate understanding of transport patterns and processes important to biological recruitment events will require precision of model estimates for the GRNMS.

Strategy 7 “Monitor socioeconomic impact”

Socioeconomic trends such as coastal population growth and development can influence sanctuary resource. Monitoring reserve use through telephone and logbook surveys provides a mechanism to evaluate potential risks posed by human activity. Logbook programs for fishers within the GRNMS, for example, could provide managers an index of how mortality associated with human activity changes year to year. These types of assessments, in conjunction with

Strategy 3, aid in the prediction, planning, and adaptation of management strategies for the GRNMS. This adaptive management approach can help managers assign risk when balancing the objectives in order to enhance appreciation and use of the marine environment, while maintaining the GRNMS for future generations.

Case Study 9: Papahānaumokuākea Marine National Monument (United States)

Description: The ~362,000 km² Papahānaumokuākea Marine National Monument (PMNM) encompasses the northwestern Hawaiian Islands archipelago. The archipelago, which is among the most remote in the world, received special designation under maritime law conveying the fragility and value of its unique ecosystems. The habitat provided by the PMNM supports high biodiversity; indeed 25% of species are considered endemic to the Hawaiian Islands. The trophic structure of the PMNM is also uniquely ‘top heavy’ with apex predators such as sharks and jacks dominating nearly 50% of the biomass (NOAA 2015a).

The overall mission of the PMNM is to “*Carry out coordinated and integrated management to achieve the primary purpose of strong and long-term protection of the marine ecosystems in their natural character, as well as the perpetuation of Native Hawaiian cultural practices and the conservation of heritage resources of the Northwestern Hawaiian Islands*” (NOAA 2004). Like the Laurentian Channel AOI, the PMNM focuses on maintaining ecosystem processes and biodiversity by restricting human use over a large area. Though considerably larger than the Laurentian Channel AOI, the PMNM also monitors a relatively large area (>15,000 km²; Figures 1-2) and shares similar objectives that make for useful comparison.

Research and monitoring: The PMNM management objectives span 7 categories that cover the biological, cultural and socioeconomic value of the area (Table 4). Each management objective has a suite of actions and potential indicators that all feed back to the overarching goal of long-term protection of the marine ecosystem.

Table 4. Management objectives of the Papahānaumokuākea Marine National Monument (NOAA 2004).

ID	Management objectives
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- 1 *Protect, preserve, maintain, and where appropriate restore, the natural biological communities, including habitats, populations, native species, and ecological processes of the Sanctuary as a public trust for current and future generations.*
 - 2 *Provide for comprehensive and coordinated conservation and management that recognizes and complements existing jurisdictional boundaries and management regimes and involves stakeholder communities.*
 - 3 *Manage, minimize, or prevent negative human impacts by allowing access only for those activities that do not threaten the natural character or biological integrity of any ecosystem of the region.*
 - 4 *Enhance public awareness, understanding, and appreciation of the marine environment and cultural and maritime heritage resources.*
 - 5 *Support Native Hawaiian cultural, religious, and subsistence practices that are consistent with the long-term conservation and protection of the region.*
 - 6 *Support and coordinate research and long-term monitoring that improves management decision making and is consistent with the conservation and protection of the region.*
 - 7 *Maintain ecosystem integrity by limiting and controlling fishing activities using an ecosystem-based management approach. Maximize ecosystem protection while minimizing adverse socioeconomic impacts. Limit fishing activities to areas that minimize or prevent interactions with corals, seabirds, endangered Hawaiian monk seals, and other protected wildlife, or that do not threaten the natural character or biological integrity of any ecosystem of the region.*
-

Much of the monitoring of the PMNM comes from enforcement of usage restrictions and larger-scale proxies of habitat integrity and associated biota. In particular, the PMNM has implemented a seascape management approach to monitor and manage usage of biota in the Hawaiian archipelago. Seascape management builds on the premise that effective management of human impacts on species and assemblages of concern requires spatially explicit information that quantifies the supporting seascape. For example, benthic habitat can be a major determinant of fish assemblages and has therefore been used as a proxy for evaluating natural resources over large areas (Friedlander et al. 2010). Corroborating this assertion, recent modelling of habitat and fish assemblage data for the Great Barrier Reef, Australia,

showed that abiotic habitat representativity alone could be used to meet biodiversity conservation targets > 90% of the time (Sutcliffe et al. 2015). Changes in seascape characteristics in this way can proxy larger-scale biological change. For example, reductions in coral and increases in macroalgal cover in protected relative to unprotected areas provided an indirect indicator of the PMNM performance in protecting benthic assemblages (Friedlander et al. 2010). This approach parallels the spatially aggregated database described earlier for the Gray’s Reef National Marine Sanctuary (Case Study 8).

The monitoring approach for the PMNM encompasses three main research themes that collectively inform management objectives (Table 4). These themes focus on measuring the biotic and abiotic resources of the reserve, creating predictive models to evaluate future trends in the marine ecosystem (e.g. in response to climate change), and partitioning the human and natural influences on trends observed in the PMNM (Table 5). Monitoring of the PMNM focusses primarily on model-based approaches to evaluate current and potential future status of resources within the PMNM. By applying a seascape-based approach, managers can direct focussed monitoring (e.g. biodiversity hotspots) and develop synoptic assessments for a very large management area.

Table 5. Monitoring themes of the Papahānaumokuākea Marine National Monument (NOAA 2011).

Theme	Actions	Examples
Map natural resources	Benthic habitat mapping, Bathymetric LiDar, Basic research	<ul style="list-style-type: none"> • Create a GIS based inventory of benthic habitat. • Survey replicate sites on different habitat types to establish a definitive connection between biodiversity or focal species abundance and habitat (e.g. Friedlander et al. 2010). • Identify hotspots for biodiversity or areas essential to focal species (e.g. Hawiian monk seal habitat; Friedlander et al. 2005). • Rugosity surveys of hard bottom habitat (Friedlander et al. 2010).
Model ecosystems	Linkages, impacts, management, ecosystem functioning, predictive models	<ul style="list-style-type: none"> • Ecosystem models that predict how a censused ecosystem will likely respond to change (e.g. Trophic structure or dispersal; Walters et al. 1999, Laurel & Bradbury 2006 respectively).

Monitor natural resources	Human vs. natural change Resource status and trends	<ul style="list-style-type: none">• Apex predator increased in protected areas. Fish biomass increased over time under protection and decreased without protection (Friedlander et al. 2010).• Species richness and diversity remained stable over time within Hawaiian protected areas (Friedlander et al. 2010),
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The monitoring plan of the PMNM includes specific detail for research on threatened species. In particular, targeted monitoring has focussed on the threatened Hawaiian monk seal (*Monachus schauinslandi*). Population models for the monk seal predict a population decline below 1000 individuals, despite directed management to mitigate further declines. Home range analysis suggests that the PMNM protects monk seals for the majority of their life-history, so changes in PMNM coverage are not required (NOAA 2011). Genetic analysis reveals some weak population structure within the PMNM that divides monk seals into six potential sub-populations. Population estimates place the majority of remaining monk seals in one sub-population unit on the French Frigate Shoals, although age structure skewed to older individuals portend future declines (Littnan et al. 2009). Active mediation has been suggested in the form of translocation of individuals between sub-populations to offset declining abundance and increase breeding success, as well as a potential cull of Galapagos shark (*Carcharhinus galapagensis*) to reduce juvenile mortality (NOAA 2011).

Given the large size of the PMNM and the goal of permitting multiple sustainable activities, managers propose Ecosystem Based Management (EBM) as the most pragmatic solution to managing a complex system (NOAA 2011). EBM, defined broadly as an integrated approach that expressly considers the entire ecosystem, includes the underling linkages and cumulative impacts of human activity within and as part of the ecosystem (Toonen et al. 2011). The successful application of EBM to the large-scale area encompassed by the PMNM requires delineating biological units of the area, much like Department of Fisheries and Oceans has defined ‘ecoregions’ for Canadian marine systems (DFO 2010b). Connectivity defines the level of connection between two areas and can be used as a proxy to delineate potential

management units. Given the high biodiversity associated with the PMNM, delineations would best be made by considering the collective connectivity and dispersal potential of multiple species. Genetic surveys of approximately 60 species of marine biota, including fish, corals, and invertebrates identified 4 concordant barriers to dispersal in the Hawaiian archipelago (Toonen et al. 2011). This result suggested spatial disaggregation of monitoring and management with respect to the barriers identified by Toonen et al. (2011). In addition, the multi-species approach to delineate connectivity within the Hawaiian archipelago helps define the probability of spillover to unprotected areas near the main islands.

Case Study 10: The Plaice Box (Europe)

Description: The 38,000 km² plaice box, a large fisheries management MPA, spans waters along the Dutch, German and Danish coasts of the North Sea. The MPA was established in 1989 to offset population declines associated with bycatch of undersized plaice (*Pleuronectes platessa*) in their main nursery area. The objective of the MPA is to decrease juvenile mortality of plaice and thereby enhance fisheries recruitment (Pastoors et al. 2000). The Plaice Box is comparable in size to the Laurentian Channel AOI and occupies a similar temperate marine environment. Although the objective and associated indicators focus on production, the observed patterns and potential failure of the Plaice Box MPA are directly relevant to the development of monitoring in Laurentian Channel.

Research and Monitoring: The Plaice Box specific objective is to reduce fishing mortality (bycatch) of juvenile plaice. Unfortunately, juvenile surveys and recruitment indices showed decreased recruitment as a function of time post-protection. This result was surprising given the removal of the principle vector of non-natural mortality but a major shift (cooling) in environmental conditions coincided with the establishment of the Plaice Box. The observed declines in recruitment were therefore partially attributed to environmental change in which decreased growth rate and increased natural post-settlement mortality offset the benefits provided by reduced fishing mortality. Concomitant changes in abundance of demersal fish and epibenthos in the southeastern North Sea also likely contributed to increased natural mortality (Pastoors et al. 2000). Though the causal links between increased natural mortality and recruitment limitation

seem intuitive, the lack of suitable reference sites precluded any objective evaluation of these potential factors (Pastoors et al. 2000). Monitoring focused exclusively within the reserve and suggested that the MPA failed to meet its objective of decreasing post-settlement mortality, however, monitoring could not definitively attribute this failure to the management action.

Decreased spawning stock biomass was also posed as a potential link to decreased recruitment after protection (Pastoors et al. 2000). For species that span the boundaries of a reserve for a portion of their life history, as expected for plaice in the fisheries closure, uncontrolled mortality outside the reserve can offset any potential benefit associated with the management action. This is particularly true where large-scale closures redistribute and sometimes concentrate fishing effort. Multiple studies show that concentrations of fishing effort at the periphery of marine reserves, known as ‘fishing the line’ (Kellner et al. 2007), strongly mitigate against marine reserve success. For example, density gradients and modelling exercises showed that concentrated fishing activity at the borders of the Dana Point State Marine Conservation Area, California (USA) reduced stocks within the reserve (Kellner et al. 2007). For the Plaice Box, a lack of spatially aggregated catch data from active fishers precluded a similar analysis to explain reductions in spawning stock biomass.

Although monitoring in the plaice box suggested that the MPA failed to meet its objectives, the protection of spawning and nursery areas nonetheless provided positive benefits for mobile species (Beck et al. 2001, Dahlgren et al. 2006). Modelling work in the Mediterranean concluded that protection of the spawning and juvenile nursery areas of Mediterranean hake (*Merluccius merluccius*) could offer detectable benefits. Limiting fishing activity in the spawning-nursery reduced post-settlement mortality and increased the probability of successful spawning (Horwood et al. 1998, Apostolaki et al. 2002, Gruss et al. 2011). Similarly, demographic models on blacktip shark (*Carcharhinus limbatus*) spawning grounds in Terra Ceia Bay (Mexico) predicted significant positive benefits of protecting key nursery habitat (Heupel & Simpfendorfer 2005). These predictions, however, assume that changes in natural conditions will not offset

the benefit offered by increasing production and decreasing post-settlement mortality, as occurred in the Plaice Box (Pastoors et al. 2000).

Application to the Laurentian Channel AOI

Monitoring protected areas that seek to protect biodiversity requires a multi-faceted approach, particularly for large ($>10,000 \text{ km}^2$) protected areas characterized by a diversity of habitat types and biological communities. Successful monitoring at large spatial scales will require comprehensive understanding of the spatial characteristics and dynamics of the protected area. Habitat evaluation is commonly featured in biodiversity-based monitoring programs. Whether monitoring entails direct measurements of specific habitat features (i.e. Darwin Mounds and Gray's Reef; case studies 7 and 8) or broad-scale habitat characterization via remote sensing (Papahānaumokuākea Marine National Monument; case study 9), a comprehensive evaluation of habitat characteristics underscores successful monitoring. Given the relationship between species and their habitat, the ability to successfully describe and predict species distributions, as well as detect change, is predicated on a foundation of the biotic and abiotic supporting factors. Targeted monitoring of specific biological attributes (i.e. abundance or species richness) can be added to this baseline, which can also be used in developing predictive models. For example, the Gray's Reef Monitoring plan dedicates a significant portion of effort to developing a geo-referenced database that overlays biological features collected overtime on a map detailing habitat distributions (case study 8; NOAA 2006). Integrated spatial management or seascape management provides a pragmatic approach to evaluate and partition monitoring over large-scale and/or dynamic conditions. Variation in habitat condition affects species composition and expected response to protection, and therefore monitoring should be equally as responsive. For example, different species assemblages characterize different depths within the Laurentian Channel AOI (Supplemental Figure 7) and therefore require specific monitoring consideration. Multivariate cluster analysis based on habitat type, species assemblages or genetic similarities (e.g. PMNM; Toonen et al. 2011) can help to identify discrete assemblages requiring specific monitoring approaches.

Large spatial coverage increases the potential influence of dynamic environmental conditions. As seen in the Plaice Box (case study 10), environmental variability can alias results of the monitoring program and confound data interpretation relative to conservation objectives. Objective evaluation requires partitioning the reserve effect from natural variation. Statistical control can account for natural variation either using reference sites (Section 3.2.1) or by direct measurements of environmental conditions. Where possible, monitoring should complement existing environmental monitoring and/or circulation models developed for the region. For example, the addition of a data buoy at Gray's Reef (case study 8) ensures the highest possible precision in surface circulation and temperature predictions for the regional ocean monitoring program (NOAA 2006). Collection of ADCP (acoustic-Doppler current profiler) data at multiple locations in the Laurentian AOI could increase spatial resolution or validate circulation models developed for the region, such as the NEMO-OPA 3D hydrodynamic model of the Gulf of St. Lawrence (see Chasse & Miller 2010 for model details). Increasing environmental monitoring precision will better enable managers to account for natural variability when assessing the benefits of protected areas (NOAA 2006).

The majority of MPAs reviewed here include highly migratory species (e.g. sharks, whales and turtles). However, none of these studies included conservation objectives specifically focused on species that spend the majority of their life history outside the reserve. Uncontrolled mortality and natural variability, which cannot be partitioned from the impact of the reserve, likely explain this omission. For example, although monitoring measures abundance of spawning aggregations of the short-tailed stingray in the Poor Knights Marine Reserve, this abundance is not used as a performance indicator for the reserve. More often, conservation objectives and associated monitoring focus on maintaining the conditions or habitats needed to maintain these species while in the reserve (e.g. Cetaceans in Gray's Reef National Monument). If employed, direct monitoring of migratory species typically measures abundance or density of a specific life stage, or, incorporates their presence into analyses of species and functional diversity. Monitoring programs that encompass only a portion of the life history, or time during migration, of a highly migratory species will find objective evaluation of specific conservation objectives difficult, if not impossible.

Lewis et al. (2014) provide a detailed list of potential indicators for the Laurentian AOI (Table 5 *in* Lewis et al. 2014) in addition to those detailed in Sections 3.2 and 3.3. In comparison to the case studies presented this list of indicators and research objectives leaves few gaps. Given the size of the Laurentian Channel AOI, successful evaluation will require a synoptic monitoring program. Where possible, developing monitoring plans should incorporate habitat and environmental conditions into assessments.

Section 3.2 Novel approaches to monitoring

Stakeholders expect marine protected areas to provide positive (e.g. increased biomass, reproduction, density, etc.) or mitigating (e.g. cessation of habitat destruction) effects in order to prove an effective management tool. An effective conservation portfolio therefore requires assessment of the actual impacts of MPAs as management interventions. The challenge for monitoring is to determine objectively whether the system within protection differs, or is better, than the system would have been had the MPA never been established. While we can directly measure the former, the latter must be evaluated through indirect means (Osenberg et al. 2010).

Conservation objectives influence how MPA design will be evaluated relative to other management options. Whether or not an MPA meets its conservation objectives may depend on whether detecting a biological response is even possible in light of natural variability and other drivers of change, and whether any observed response can ultimately be attributed to the management action. Although opinions on the absolute efficacy of MPAs as a management tool can be mixed (e.g. Hilborn et al. 2004, Hilborn 2007), the MPA literature overall suggests that MPAs promote diverse, and in some cases hierarchical, biological responses.

The diversity of methods to monitor change mirrors the diverse biological responses expected of MPAs (e.g. NRC 2001, Roberts et al. 2003b, IUCN 2004, Pomeroy et al. 2005, Osenberg et al. 2010, Stelzenmuller & Pinnegar 2010, Fox et al. 2014). In general, biomass, density, and abundance of protected organisms are expected to increase through time and space, (reviewed in Sections 1.1.2 and 1.1.1 respectively). Although many stakeholders expect

differences between protected and unprotected areas for a given species (though see Section 1.1.3 for considerations of location), it is rarely possible to conclude that a protected ‘ecosystem’ differs from unprotected ecosystems. This is especially true for systems like the Laurentian Channel which were in relatively pristine condition and sparsely exploited (see Supplemental Figures 2-6) prior to protection. Given the potential diversity of changes in response to protection (Figure 3), it is important for monitoring programs to consider a portfolio approach when monitoring; focusing beyond conventional changes in species abundance and density.

Here we review approaches to monitor MPAs beyond simplified changes in abundance. We discuss current approaches to monitoring in existing and proposed NL MPAs, and highlight areas of opportunity for monitoring the efficacy of the MPA in fulfilling, directly and indirectly, its conservation objectives.

Section 3.2.1 Before after controls impact analyses (BACI) and monitoring design

Environmental monitoring of any type is a form of scientific research that must conform to the basic tenets of experimental design (Underwood 1992, Underwood 1994). Control-impact monitoring designs provide the most widely used tool to determine whether observed differences inside versus outside an MPA are attributable to the management action. Control-impact designs compare the biological systems under protection (a ‘*treatment*’) to those in suitable reference sites outside. Although control-impact analyses can provide a relative index of biological status, they generally fall short of providing a mechanism to evaluate objectively differences relative to external forcing (e.g. environment; Section 3.2.4) and prior condition (see Sections 1.1.3 and 1.1.4) (Osenberg et al. 2010). Control-impact analyses lack the ability to differentiate between management impacts and natural variability, thus limiting their ability to successfully evaluate MPAs. For example, the rapid effects suggested by Halpern and Warner (2002) may simply reflect an artifact of the monitoring programs used in the meta-analysis. In reviewing the literature utilized in the Halpern and Warner (2002) meta-analysis, it was found that most studies were control-impact based, and therefore lacked the ability to assign differentiate management effects from natural variation given the lack of prior knowledge of underlying

variability (Osenberg et al. 2010). Control-impact designs lacking prior condition data risk misattributing ‘site effects’ to ‘reserve effects’ (Osenberg et al. 2010). Indeed no two sites are completely alike if sampled sufficiently, and in this sense spatial variation always complicate control-impact designs. Not surprisingly, subsequent studies refute Halpern and Warner (2002)’s early meta-analysis , and show variable reserve effects that depend on species, locations, and timing (e.g. Willis et al. 2003a, Russ et al. 2005, Guidetti & Sala 2007, Goni et al. 2010b, Russ & Alcala 2010, Willis 2013).

Considering the limitations of control-impact monitoring to unambiguously assign the effect of a management action, protected area researchers generally recommend before-after-controls-impact (BACI) based designs for monitoring (e.g. Russ 2002, Benedetti-Cecchi & Osio 2007, Tetreault & Ambrose 2007, Osenberg et al. 2010). BACI designs anticipate that the management action will result in a different pattern of change, pre and post protection, compared to a control location, if the management action indeed leads to a detectable effect (Underwood 1992). By sampling before and after the intervention, at a series of locations, the relative changes among replicate sites accounts for natural variation between locations, such as habitat differences (e.g. Francini-Filho & Moura 2008, Kay et al. 2012). This approach also accounts for prior trajectories of protected populations that control-impact approaches cannot (Rowley 1994, Gruss et al. 2011). Replicated sampling in the time before implementation can account for temporal and spatial variability, thus differentiating the effect of the management intervention from that of natural variation through time. This spatially and temporally replicated design, referred to as Beyond-BACI (Underwood 1992, Underwood 1994), is generally considered the optimal approach to evaluate the impacts of protected areas in marine systems (Osenberg et al. 2010, Willis 2013).

BACI designs assume any detectable MPA effect should produce a statistically significant interaction in the measured biological variable (i.e. abundance) between the timing of management (before-after) and the location of the sampling site (MPA-control) (Underwood 1992). In this way the control site, not influenced by protection but similar in

relevant ecological characteristics (i.e., habitat), acts as a covariate for the impact site and accounts for variables external to management that influence the response (e.g., fluctuations in climate influencing abundance). Ideally, sampling would be replicated randomly within a control and impact treatments as well as through time. This random-replicated design can account for spatial and environmental variation statistically (asymmetrical design 'Beyond-BACI'; Underwood 1992, Underwood 1994), which is described in greater detail below.

Asymmetrical BACI (aBACI) monitoring design offers the most pragmatic tool for evaluating the impact of management for a single MPA or cluster of MPAs within a common geographical area. This design compares replicate sites sampled randomly within a focal MPA to sites nested within suitable reference areas (see Section 1.1.3 for considerations on replicate site selection). For each sampling event, the average biological response (i.e., abundance or biomass) within a location and among randomly selected sites, is compared between protected and non-protected areas (location) before and after (time) establishment. A significant interaction between time and location indicates a reserve effect. For example, Mateos-Molina et al. (2014) employed an asymmetrical BACI design to evaluate the benefits of the Mona Island MPA, Puerto Rico, to exploited reef fish species. Their monitoring program assessed abundances of 10 species of reef fish exploited by local fisheries at the beginning of protection and then again 4 years after, based on underwater visual transect surveys at random locations within two no-take MPA sites and at a fished site outside the MPA. Statistical analyses used differences in average abundance of each focal species among treatments as a function of time (before-after), location (treatment and two controls) the interaction between time and location. The statistically significant interaction between location and time in this model confirmed the effect of protection on the fish assemblage (Mateos-Molina et al. 2014). Differences in responses across time and between protected and non-protected areas were interpreted by pair-wise comparisons conducted on significant interaction terms. Fish abundance overall increased during this period at all locations but the magnitude of difference was greater within the MPA (Mateos-Molina et al. 2014). Contrastingly, the Hawke Box closed area (8610 km²) off the coast of Labrador was shown to fail in its objective to protect pre-recruit snow crabs. Comparisons through time inside and outside the MPA found no evidence of increased

pre-recruit survival (Mullowney et al. 2012). By accounting for larger scale changes in the ecosystem with a BACI design, Mullowney et al. (2012) were able to attribute the failure of the Hawke Box to meet its objective on the redistribution of fishing effort (pots) and associated increased juvenile mortality.

International interest in MPAs has focussed attention on developing monitoring programs that are robust to natural variability, operate over constrained time-scales and, importantly, utilize sound statistical design principles. Researchers consistently identify BACI monitoring designs as the best tool to meet these criteria. For example, a BACI asymmetrical design detected increased biomass of greenback parrotfish (*Scarus trispinosus*), black grouper (*Mycteroperca bonaci*) and yellowtail snapper (*Ocyurus chrysurus*) within the Brazilian Marine Extractive Reserve of Corumbrau (MERC) despite significant confounding variation in habitat quality and environmental conditions (Francini-Filho & Moura 2008). Several other studies report subtle increases in abundance of reef fishes (Arrábida Marine Park, Portugal) (Currambene Creek, Australia) and benthic macrofauna in response to protection over a short-times scale (> 4 yrs) utilizing a BACI analysis (Costa et al. 2013, Winberg & Davis 2014). In contrast, the absence of data prior to the implementation of a fisheries closure in the North Sea precluded comprehensive evaluation of changes in plaice (*Pleuronectes platessa*) abundance in response to protection (case study 10; Pastoors et al. 2000). The inability of a control-impact analysis to evaluate change relative to parallel changes in the larger North Sea ecosystem over a short period of monitoring (>10 yrs) contributed to the inability to detect an effect. Collectively these examples highlight the importance of BACI designs for robust monitoring of protected areas in dynamic systems and potentially over short timescales.

Application of traditional BACI monitoring analyses to the Eastport and Gilbert Bay MPAs is problematic because of the paucity of appropriate monitoring data prior to MPA establishment. Both areas were fished prior to establishment, however, the lack of spatially referenced cod and lobster landings data precludes spatially reference control impact analysis. In particular, without landings data specifically within the boundaries of the MPA(s), it is not

possible to compare MPA effects relative to outside areas. However, many MPAs lack data prior to establishment (e.g. Halpern et al. 2004), and this gap should not hinder MPA monitoring.

Gilbert Bay monitoring does not necessarily require an external reference location given that the MPA seeks to protect a specific stock of cod encompassed almost completely within its boundaries (Morris et al. 2014). While a BACI analysis would provide powerful insight into the efficacy of the MPA in light of external conditions, suitable reference locations would be difficult given the absence of similar unique subpopulations outside the MPA.

Similarly for the Eastport MPA, lack of spatially referenced landings data eliminates the potential for a BACI analysis. Given recent declines in lobster landings on the northeast coast of NL, and particularly in LFA 5 (DFO 2013a), a BACI design would provide a mechanism to account for any changes in ecosystem processes or population trends over the broader area and beyond MPA boundaries. However, in the absence of pre-MPA data, higher abundances, densities, and median size within the MPA compared to adjacent areas through time nonetheless provide an index of MPA performance (also see Section 2.2.6 for considerations of monitoring the Eastport MPA). Consistently higher indices inside relative to outside the MPA suggest that management intervention and not random spatial variation resulted in the observed trends in the Eastport Lobster Management Area. Similar patterns in abundance and size were reported for spiny lobster (*Panulirus cygnus*) in the similarly sized Kingston Reef Marine Sanctuary (KRMS), Australia (Babcock et al. 2007). The lack of data detailing the status of protected and non-protected areas prior to the establishment of the KRMS was, in part, mitigated by replication of control sites (n=7) and treatment sites (n=8), which demonstrated coherent trends through time suggesting that management interventions resulted in positive indicators (Babcock et al. 2007). Although the small size of the Eastport MPA precludes increased monitoring replicates within the MPA, increasing the number of monitoring sites outside the MPA could strengthen conclusions drawn from the control-impact monitoring design.

Applications of BACI analysis in temperate protected area systems are rare (Willis 2013). In a review of temperate MPAs, Lester et al. (2009) reported BACI analysis in only 1 of 31 MPAs reviewed. However, several recent studies in temperate systems demonstrate how BACI can be applied. In particular, BACI designs were successful in evaluating the positive effects of protection for European lobster and Atlantic cod (Moland et al. 2013a). The researchers in both examples noted that the BACI design was imperative for evaluating the influence of protection on previously exploited populations within dynamic environmental conditions.

Annual DFO stock assessment monitoring from the NL and Maritimes region has accumulated a long history of trawl data for the Laurentian Channel AOI (Figure 4). Fisheries independent data for trawl sets within the AOI boundaries since 1995 have used a common random stratified sampling method. The Maritimes Region monitors the southern side of the AOI (strata 558:559: Figure 18) employing different trawl gear and two months later in the year than the NL survey. The ability to merge Maritime and NL Region trawl surveys hinges upon similarity in species specific catchability between nets (e.g. Warren et al. 1997) and the variability imparted by differences in sampling season. Nonetheless, monitoring data from the NL region (Figure 4) offer a template to assess status of the area pre- and post- MPA designation. Given the relative paucity of examples of temperate MPA evaluated with BACI design (sensu Moland et al. 2013a), the limited availability of pre-designation data, and the large spatial coverage, the Laurentian Channel AOI offers excellent potential and incentive to collect baseline data (e.g. species biology, movement ecology and habitat availability) and employ a robust BACI monitoring design. The conservation objectives for the MPA primarily focus on recovery and maintenance of focal species populations through the cessation of fishing mortality (see Sections 2.2-2.8). Assuming the MPA contributes to reduced mortality in resident biota, fisheries induced and otherwise, a BACI framework would provide an appropriate mechanism to test the hypothesis of positive effects inside relative to outside the protected area.

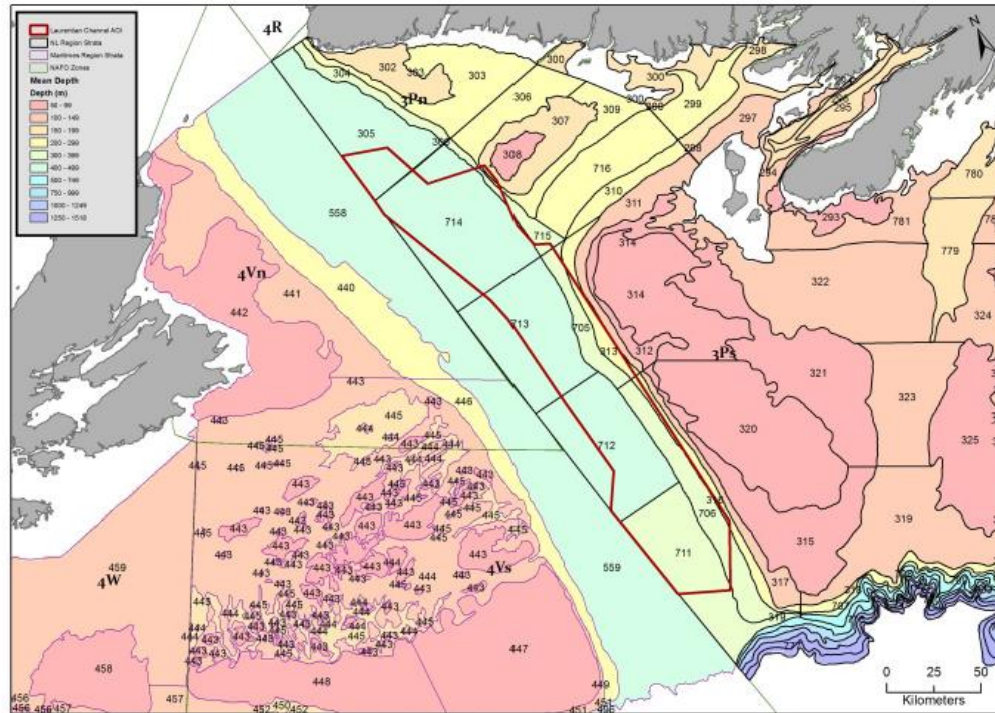


Figure 18: Multispecies trawl survey strata in the Laurentian Channel AOI region. Figure from (Lewis et al. 2014).

Asymmetric BACI designs require replicated random sampling within and outside the protected area through time (pre- and post-designation). Sampling should target areas with similar habitat (see Section 1.1.3) and, preferably, similar historical levels of exploitation. For example, the candidate sampling locations for assessment of the Laurentian Channel AOI could be selected based on available data and habitat similarity. Sampling locations should be similar in size (i.e. 20 km² grids) and sites with locations should be selected at random from year to year (Figure 19). This sampling design can incorporate fine- (>20 km) and large-scale (>500 km) variation into the model. Samples are assumed to be independent and therefore incorporate rates of turnover and life cycles of the sampled biota (reviewed in Section 1.1.2) (Underwood 1992). Accounting for turnover and life cycles can be challenging, especially for monitoring any long-lived species in the MPA (Section 2), however, randomization will reduce the chance of sampling a released individual (e.g. sea turtle, marine mammal) more than once, and thus limit violation of this assumption.

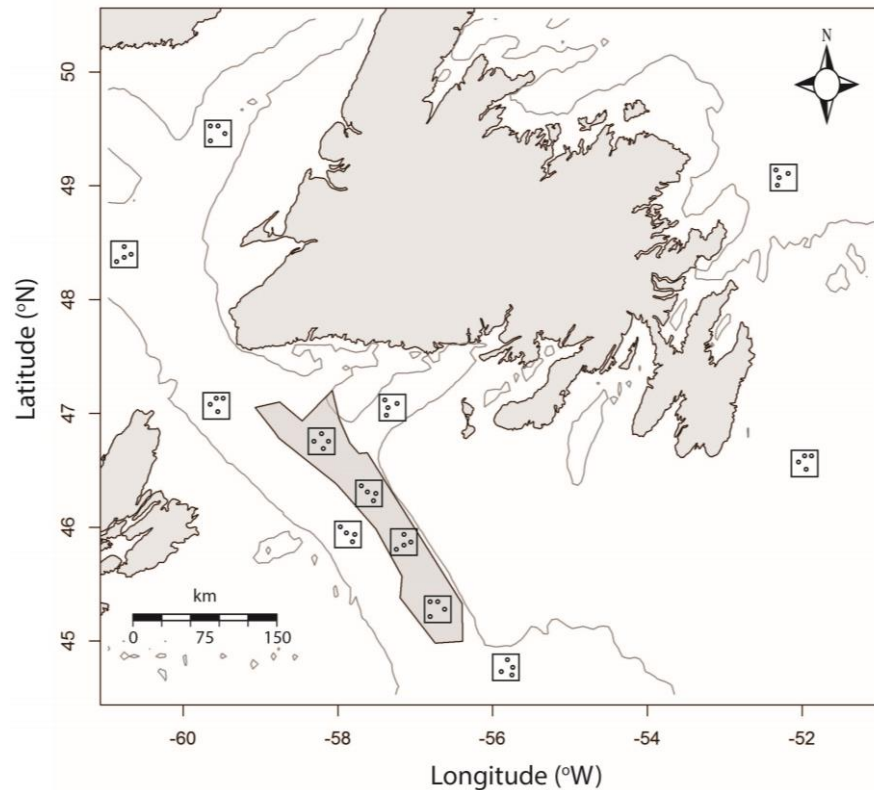


Figure 19: Hypothetical BACI sampling design. Each square represents a 20 km² sampling ‘location’ pre-selected based on common attributes (e.g. habitat). Note the locations were chosen haphazardly on a chart and are not based on similar habitat characteristics. Circles within the squares represent trawl sets or ‘sites’, the position of which is randomized year to year within a given location. Area of the Laurentian Channel AOI is shaded grey.

To test for a reserve effect, the BACI design models the measured biological trait (e.g. abundance, CPUE, density) of a focal species as a function of *designation* (fixed factor; pre or post), *location* (fixed factor; protected or unprotected), and *site* (random factor nested within location). The hypothetical design illustrated in Figure 19 encompasses 4 locations within the MPA and 8 locations (4 near and 4 far) outside the MPA, each with 4 nested trawl sets, randomized in each year of sampling. This design can be modelled using a general linear model (GLM) for a single species or a permutational multivariate analysis of variance (PERMANOVA; Anderson 2001) to test for community level effects. For example, Moland et al. (2013a) used a GLM with a Poisson-distributed error structure to test for significant interaction between designation and location on the CPUE of European lobsters in the Skagerrak MPAs,

Norway. In an analysis of community response to protection, Mateos-Molina et al. (2014) tested for average interaction between designation and location for the dominant large-bodied fish species of the Mona Island MPA with PERMANOVA.

Statistical analyses can be conducted using a variety of statistical packages which permit the use of random and fixed factor analysis. For this review we provide examples using the open source statistical platform *R* (R Development Core Team 2014), utilizing the linear mixed-effects model (*lmer*) function in the package '*lme4*'. Examples of some of the packages and applications to the questions outlined above are summarized in the Supplemental R-code section.

Some MPAs are designed to promote spillover (Section 3.1.1), and although spillover was not among the proposed conservation objectives of the Laurentian Channel AOI, its large size nonetheless creates strong potential for spillover for some species found within (see Figures 1-2 and Section 1.1.1). The hypothetical sampling design (Figure 19) selects control sites both near (<100 km) and far (>500 km) from the MPA boundaries. Any adult spillover will likely occur closer to the protected area boundaries, as reported for a variety of MPA systems (see Section 1.1.1). Given this assumption, spillover could be detected by adding a third term, *distance*, (proximity to the MPA, Fixed factor; near or far) to the models where a three way interaction between distance (near-far), location (control-impact) and designation (before-after) would imply a potential spillover effect (see example in Supplemental R-code 1:4). Utilizing a similar design, Hoskin et al. (2011) found evidence of a reserve effect and localized spillover of European lobster, brown crab (*Cancer pagurus*) and velvet crab (*Necora puber*). In their analysis, distant control sites helped to confirm a reserve effect as well as highlight a proximity effect (spillover) in change of invertebrate abundances pre- and post-designation of a no-take area within the Lundy Marine Conservation Zone.

A sampling program such as that outlined in Figure 19 requires the location of suitable reference sites and the availability of comparable data. As reviewed in Section 1.1.4, the reserve encompasses two distinctly different coarse

habitat types⁴: deep, relatively flat seafloor (300-450 m) and the steeper sides of the channel that rise up to <100 m (see Supplemental Figure 7 and discussion in Section 1.1.3). Given differences in biota within these habitat types (see Section 2), effective monitoring would require separate sampling sites within each habitat (i.e., doubling the sampling locations outlined in Figure 19). Previous level of fishing activity also merits consideration, given the limited fishing activity within the majority of the AOI in the decade prior to establishment (Supplemental Figures 3-6).

The multi-species trawl survey conducted annually by DFO (reviewed by Lewis et al. 2014) represents a valuable data addition to BACI analyses. The application of surveys during multiple seasons (spring, summer and fall) and with different gear types will require statistical control or some assumptions. Data can be pooled in this analysis, assuming survey season and gear type have little impact on abundances of a given species, though temporal autocorrelation may add complications. Alternatively, a statistical offset () could be included in the model as a term to be added to the linear predictors which would partially account for variation as a product of sampling season or gear type. The offset might simply be a multiplier of how many more times a gear type is likely to capture relative to a common reference point. For example, an offset could account for differences in capture efficiency of Western IIA trawl and Campelen trawl gear types. Similarly, an offset can account for seasonal variability in catch rate, assuming fish are present in all seasons but vary in catchability with season (Rakitin & Kramer 1996) (Supplemental R-code 5). Alternatively assuming that a given species occurs year round and can be captured by all gear types, but with varying efficiency, then data could be scaled to a common range, (i.e. mean of 0 and standard deviation of 1) among seasons. Because the BACI analysis compares abundances among pre- and post-designation and protection status, scaling would be sufficient to account for variation in capture efficiency. Any statistical summary will critically depend upon accounting for differences in catchability among gear types, species, and seasons (Katsanevakis et al. 2012).

⁴ Note that these are a very simplified overview of potential habitat categories based on limited information. The resolution of partitioning habitat categories will increase with more baseline data.

This section presents analyses which evaluate differences in biological response to protection using a BACI design. However, change relative to a prior condition might not always be the desired metric to evaluate an MPA relative to a conservation or research objective. For example, a simple comparison of abundance in protected relative to unprotected areas can provide a metric of success that can be compared to a vast literature based primarily on control-impact based analysis (e.g. reviews by Halpern & Warner 2002, Lester et al. 2009, Stewart et al. 2009). Fortunately, the statistical and design principles for a BACI analysis parallel those for a control-impact analysis. BACI and control-impact differ principally in that the assignment of a significant ‘reserve effect’ is not predicated on the interaction between designation and location, but instead focusses on the treatment (i.e., whether the protected and unprotected systems differ). For example, with abundance of Eastport lobster or Laurentian Channel porbeagle shark as the response variable, significant differences could be tested using a GLM (Supplemental R-code 6). The GLM approach permits flexibility in the model to account for different error structures for non-normally distributed residuals often associated with count data. Post-hoc comparison among locations would allow testing of the reserve effect. The addition of a time (year) variable could evaluate how change might accumulate or plateau over time (Supplemental R-code 7).

Whether the analysis utilizes a BACI or control-impact design, monitoring must follow the basic tenants of statistical design to reach a definitive conclusion on the efficacy of protection. The success of monitoring will invariably hinge on statistically detecting an effect based on available data.

Section 3.2.2 Biodiversity monitoring

The overall objectives of many conservation areas, including the Laurentian Channel, revolve around maintenance of species richness and restoration of ecosystems (Lester et al. 2009). Species richness often correlates positively with a variety of ecosystem processes such as productivity, resilience, stability and resistance to invasion (Bellwood et al. 2006, Whitfield et al. 2007, Claudet et al. 2010). Species richness and biodiversity can also provide important details on demographic-trophic processes regulating the system, which can alter the expected magnitude and

time-scale of the response (see Sections 1.1.4 and 1.1.2 respectively). Although single species approaches provide information for the conservation and research objectives directly (i.e., does the provision of protection produce a net positive benefit to the species of interest), they ignore parallel changes that may reflect dynamic and non-linear responses. For example, abundances of rock lobster (*Jasus edwardsii*) and urchin (*Heliocidarus erythrogrmma*) increased in response to protection from a series of MPAs in Tasmania. Positive responses in both species generally continued until abundance and average size of lobster peaked 7 years after protection, resulting in a concomitant decline in their urchin prey species (Barrett et al. 2009). Similarly, fish species richness and invertebrate density declined after protection, contrary to a-priori predictions (e.g. Cote et al. 2001, Halpern & Warner 2002, Halpern 2003) in response to a higher than expected increase in large predatory fish previously targeted by fisheries in Southern Australia (Edgar & Stuart-Smith 2009). Species that declined were prey of species that increased in response to reduced fishing mortality (Edgar & Stuart-Smith 2009). In Mombassa Marine Park, Kenya, reduced fishing mortality of algal-consuming parrot fish led to increased coral cover, through mediation of coral-algal competition (McClanahan 1997). These examples highlight the dynamic responses of ecosystem components that would remain undocumented if monitoring focused on a single species.

The terrestrial and marine literature both review biodiversity extensively. Biodiversity may be parameterized in numerous ways for comparative analysis, including summary metrics such as species richness, Simpson's index, and the Shannon-Wiener index (Table 6). These indices may be treated as response variables to protection within the survey design and analysis templates outlined above (Section 3.2.1.). Addressing trends in these indices through time provides a metric of how the MPA protects associated biota. For example, in the Apo Island marine reserve (< 1 km²) species richness increases linearly with time and nearly tripled after 17 years (Russ & Alcala 2011). This increased species richness occurred inside and outside the reserve and could not be attributed to changes in habitat, therefore suggesting successful protection and associated spillover.

Table 6. Common indices of biodiversity. N = total number of species in a sample, n is the total number of a given species in a sample, N_o = the total number of organisms in a sample, and K is the total number of samples (Magurran 2004).

Biodiversity metric	Description	Calculation
Species richness	Total number of species present in a given sample or strata	N
Rarefaction	Calculation of species richness as a function of the number of samples taken	$f_r = K - \left(\frac{N}{n}\right)^{-1} \sum_{i=1}^K \frac{N - N_i}{n}$
Simpson index	Probability that two individuals from the same sample belong to the same species (or grouping factor). Accounts for species richness and the proportion of a species in a sample.	$\sum (n/N_o)^2$
Shannon-Wiener index	A metric which evaluates the uncertainty of predicting which species an organism will be from a sample given the total number of species in the sample. This parameter accounts for the richness and evenness of the species in a sample.	$-\sum_{i=1}^N p_i \ln(p_i)$

Though the value of measuring biodiversity is well reviewed (e.g. Airame et al. 2003, Baskett et al. 2007) and suitable for some MPA objectives (i.e. the Great Barrier Reef), the majority of monitoring focuses on changes of single focal species in response to protection. Often, those studies that evaluate biodiversity quantitatively focus on broader scale gap-analyses, which evaluate how a current management intervention explicitly protects the biodiversity in a given area. For example, existing MPAs along British Columbia’s coast generally did poorly at protecting the associated biodiversity when any fishing activity was permitted (IUCN categories IV – VI, Supplemental Table 1) (Ban et al. 2014). Similarly, gap-analysis suggested that MPA extent and placement in the Philippines was inadequate for protecting marine biodiversity, even though such protection was the primary objective for developing the MPA system

(Weeks et al. 2010). Gap-analyses provide a useful tool for evaluating whether an MPA or MPA system is representative, but do not provide a useful monitoring tool for a single MPA at an established location.

For the Laurentian Channel AOI, biodiversity monitoring within and outside the MPA over time can provide a relative index of performance relative to the global objective to “*Conserve biodiversity in the Laurentian Channel MPA through protection of key species and habitats, ecosystem structure and function and through scientific research*”. A large-scale review and control-impact analysis of MPAs in the northern coast of the Mediterranean Sea, found significantly higher species richness in protected compared to unprotected areas. Corresponding trophic structure within the MPAs was more ‘top heavy’ with increased abundance in large predatory fishes, presumably partly because of increased fishing mortality outside the MPA boundaries (Guidetti et al. 2014). In contrast, protection increased the density of individual species within the Rottneest Island Marine Reserve, but resulted in no significant difference in species richness (Vanderklift et al. 2013). These contrasting patterns indicate no significant change in trophic structure in the MPA with protection and hence no change in biodiversity overall. Only a sample weighted species richness analyses (e.g. Table 6) could detect a small effect of protected status. Benthic sediment samples from the Darwin Mounds MPA, United Kingdom, found that in general indexes of biodiversity were higher in trawled areas than sites within the MPA. This result was partially attributed to the colonization of opportunistic detritivore species utilizing disturbed habitat (Serpetti et al. 2013). These examples highlight the potential use of biodiversity as a metric of MPA performance and the importance of considering biodiversity indices along with the relative abundance of constituent species.

Biodiversity measurements can also provide a tool to assess the suitability of reference sites. Monitoring MPA performance relative to reference sites is generally predicated on similarities in habitat type (see Section 1.1.3) and the presence of the focal species. However, given the dynamic interactions between multiple species, the community composition of reference sites should also be considered to identify potential variability in non-focal species. For example, researchers identified suitable reference sites for the Maria Island Marine Reserve explicitly through multivariate analyses (Edgar & Barrett 1997). In light of the available survey data, the Laurentian Channel management

plan could use a similar biodiversity-community based approach to select reference sites. If community data were available, the applicability of external reference sites for the Eastport MPA could be evaluated in the same way. Those sites found to be not comparable (e.g. different community assemblage) could be removed from further monitoring and their incorporation into long-term trends adjusted.

Section 3.2.3 Functional diversity monitoring

Increased pressure from human activity and concomitant alteration of marine ecosystems have been attributed to increasing disruption of ecological processes (Green et al. 2009b, Claudet et al. 2010). Ecological processes support the services provided by ecosystems (e.g. fisheries). As management moves to manage how humans interact with entire ecosystems (e.g. integrate coastal zone management), maintaining ecosystem services (and thus monitoring) becomes critical to gauging success (Fraschetti et al. 2010). Loss of biodiversity or components of the ecosystem often leads to disruption in ecosystem services and overall reduced resiliency of the system to stress (Bellwood et al. 2006, Green et al. 2009b).

Biodiversity represents a quantifiable metric to help inform conservation objectives (Section 3.2.2) both directly (conserve biodiversity) and indirectly (e.g. potential positive relationships between biodiversity and focal species abundance). However, species richness alone cannot explain variability in ecosystem processes, because relative abundance and biological attributes of constituent species also contribute to those processes and may not co-vary with species richness. For example, comparisons of benthic sedimentary communities revealed that higher biodiversity outside the Darwin Mounds MPA in areas which were subject to trawling activity (Serpetti et al. 2013). Increased diversity was primarily driven by higher numbers of detritivore species in disturbed areas. Though the biodiversity values were higher, the increased species richness in this case was likely comprised of functionally redundant species and therefore the assignment of MPA performance based on biodiversity comparisons might be misleading. Increasingly, researchers view the maintenance of functional structure as more important to ecosystem services than the maintenance of taxonomic diversity (e.g. Snelgrove 1999, Danovaro et al. 2008, Snelgrove et al. 2014), and therefore

might be a more suitable indicator of conservation objectives that emphasize maintenance of ecosystem services (Claudet et al. 2010).

Functional diversity describes the unique behaviours or ecological roles that species play in communities, and links strongly with ecosystem services (Danovaro et al. 2008). Protecting the most functionally distinctive species will maintain unique biological features such as morphological characteristics and associated behaviours. For example, Bellwood et al. (2006) found that a single species of batfish (*Platax pinnatus*) was the primary driver of phase shifts from macro-algal to dominated coral dominated system, because of its ability to consume large amounts of macro-algae. Mouillot et al. (2008), formalized this concept with the Conservation of Biological Originality index (CBO), a method that incorporates morphological, taxonomic, and distributional data to quantify the effect of protecting critical traits as opposed to those that are functionally redundant. CBO analysis revealed that the Bonaifacio Strait Natural Reserve, France, maintained higher levels of functional diversity and better protected ecosystem services than fished areas (Mouillot et al. 2008). This finding suggests that by protecting multiple species and increasing resiliency MPAs buffer against erosion of ecosystem services. However, the ability of an MPA to conserve functional originality varies among systems and with MPA design. Indeed, a review of Mediterranean protected areas illustrated that the current system of MPAs was only slightly better (~ 26%) at maintaining or improving functional diversity than a random placement. In particular MPAs frequently ignored the most functional original species, those with rare functional traits, thus limiting their potential to meet targets for maintaining functional diversity and associated ecosystem services (Guilhaumon et al. 2015).

While conservation objectives for the NL MPAs focus primarily on individual species (Section 2), the maintenance of intact functional ecosystems can indirectly indicate MPA performance, which in turn supports the more specific objectives. In this way functional diversity invariably links to biodiversity (Green et al. 2009b) and monitoring functional diversity would therefore inform biodiversity focused conservation objectives. For example, in the deep sea,

ecosystem function has been linked to both biodiversity (species richness) and functional diversity (Danovaro et al. 2008). Monitoring both species diversity (Section 3.2.2) and functional diversity will inform both the broad umbrella objective and, indirectly, the species objectives of the Laurentian Channel AOI.

The calculation of functional diversity requires information on the functional traits of species in the system. Traits may be based on feeding types (Danovaro et al. 2008), morphology (Mouillot et al. 2008), trophic guild, semelparity, reproduction, or behaviour (Guilhaumon et al. 2015) (reviewed in Table 7). For example, a new analysis of groundfish survey data, (Pedersen et al. in prep) documented relatively stable functional diversity leading up to the collapse of the groundfish fishery compared to significant changes in abundance and biodiversity over the same period. They attributed this result in part to the functional traits that characterize fish exploited by the fishery (Supplemental Table 4), leaving rarer, less impacted species comprising the majority of functional diversity remaining within the system. The maintenance of the functional scope of the system will likely play an integral role in the recovery of groundfish community.

Table 7. Review of traits used to describe functional diversity in marine systems. These examples represent the subset of literature reviewed for this report that provided detailed information about traits used to calculate functional diversity.

Trait	Ecosystem function	Levels	Reference
Feeding types	Representation of ecological role defining the selection of food items	Numerical indices including buccal size, mouth morphology, and feeding appendage morphology	(Danovaro et al. 2008)
Trophic guild	Food web structure	Carnivorous, piscivorous, herbivorous, omnivorous	(Guilhaumon et al. 2015)
Diet	Food web structure, energy, and material flow between species	Benthivorous, Planktivorous, Piscivorous	(Wiedmann et al. 2014)
Trophic level	Food web structure	Numeric index assigning the estimated position in the food chain	(Pedersen et al. in prep)
Morphological features	Speed, manoeuvring ability, endurance, adaptation to lighting	Biomass, caudal ratio, eye diameter, eye position, oral gape height, mouth protrusion	(Danovaro et al. 2008, Mouillot et al. 2008, Claudet et

	conditions, food selection	length, height of gill raker, body depth, ratio length to height/width, caudal peduncle length, position of mouth, orientation of mouth, teeth, gut length	al. 2010, Schleuter et al. 2012, Pedersen et al. in prep)
Body size	Index of placement in food web and ecological role	Maximum body length	(Wiedmann et al. 2014, Guilhaumon et al. 2015, Pedersen et al. in prep)
Body shape	Sediment re-suspension and release of material. Paths of energy and material flow. Energy allocation	Flat, Eel-like, Normal, Elongated, Short/deep	(Wiedmann et al. 2014)
Vertical distribution	Benthic pelagic coupling	Benthic, demersal, pelagic	(Wiedmann et al. 2014, Guilhaumon et al. 2015)
Habitat	Distribution and plasticity to change	Sand, rocky, biogenic	(Wiedmann et al. 2014, Guilhaumon et al. 2015)
Migration	Seasonal movements and habitat utilization. Terrestrial – marine coupling	Anadromous, catadromous	(Guilhaumon et al. 2015)
Tolerance range	Combination of tolerance range and body shape influences the availability of imported energy	Highly, moderately or not: temperature, salinity and depth tolerant	(Wiedmann et al. 2014)
Reproduction	Reproductive capacity	Oviparous, ovoviviparous, viviparous	(Guilhaumon et al. 2015)
Sex-change	Reproductive plasticity	No sex change, simultaneous hermaphrodite, protandrous hermaphrodite, protogynous hermaphrodite	(Guilhaumon et al. 2015)
Population doubling time	Index of the temporal scales over which populations operate	Average reported minimum population doubling time	(Pedersen et al. in prep)
Fecundity and offspring size	Production rate and energy allocation in the	Fecundity Large offspring,	(Wiedmann et al. 2014)

	ecosystem	Medium offspring, Small offspring, No larval phase, Lecithotrophic, Planktotrophic	(Guilhaumon et al. 2015)
Larval behaviour	Connectivity and dispersal		
Aggregation	Index of the sociality or schooling behaviour	Rare, irregular, schooling, shoaling, none	(Pedersen et al. in prep)

Functional traits can be inferred from primary literature, taxonomic keys, and online databases. In particular, FishBase (Froese & Pauly 2015) includes an extensive repository of functional traits for fish species (e.g. Table 5; Supplemental R-code 8:12). Unfortunately, FishBase covers only fish species and lacks information in invertebrates (e.g. American lobster and sedimentary fauna), reptiles (e.g. leatherback sea turtle), or mammals. Other online resources such as the World Register of Marine Species (WoRMS 2015), Encyclopedia of Life, and peer-reviewed literature offer a wide range of species information and links to related resources.

Because the data used to infer functional attributes often mix quantitative and qualitative characteristics (Table 7), functional diversity cannot be calculated in the same way as conventional biodiversity metrics (e.g. Shannon diversity or species richness; Soykan & Lewison 2015). Functional diversity is instead calculated using Gower dissimilarity, which utilizes pairwise functional distances, using both quantitative and qualitative data (Gower & Legendre 1986). With Gower dissimilarity several variants of functional diversity can be calculated in multidimensional trait space (Table 8; Laliberte & Legendre 2010). Like biodiversity, functional diversity can be based on presence-absence of a given trait or weighted by abundance. If a species is equally likely to be captured in by a gear type and different time of year, then the presence-absence weighting could offer a potential solution to different sampling times and gears (Section 3.2.1).

Table 8. Variants of functional diversity (Laliberte & Legendre 2010)

Trait	Description	Citation
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Functional evenness	The evenness of abundance distribution in functional trait space. This metric measures the evenness of the distribution of traits across a trait gradient.	(Mason et al. 2005)
Functional dispersion	Dispersion of traits within a community. The average distance in multi-dimensional trait-space of species within a sample.	(Laliberte & Legendre 2010)
Functional richness	The amount of functional space filled by a community. For one trait this would be interpreted as the difference between minimum and maximum values of a given trait within a community.	(Villegier et al. 2008)
Functional divergence	How abundance of traits is spread along a functional trait access within the range occupied by the community. Low divergence would arise with the most abundance species have traits which are closest to the centre of the trait range	(Villegier et al. 2008)

Functional diversity (Table 8) can then be calculated for each trawl set and used as a variable to compare among years (Section 1.1.2), protection status, or distance from the reserve (Section 1.1.1) (Supplemental R-code sections 13:14). Each variant of functional diversity (Table 8) can be analysed using a BACI model (Supplemental R-code 1:5) or a control-impact design (Supplemental R-code 6:7).

Functional diversity provides an indicator that indirectly supports species specific objectives but also provides a broader perspective on how the system may change through time, both with and without protection.

Section 3.2.4 Genetic diversity monitoring

Many MPAs explicitly set out to maintain species and (less frequently) functional diversity, but the potential for MPAs to conserve genetic diversity receives less attention. Fisheries can select for specific traits (i.e. growth) which, in turn, may alter genetic structure and diversity of a system. Maintenance of ecosystem function depends not only on protecting key species and associated traits, but also the genetic variation within each species (Allendorf et al. 2008). Several researchers hypothesize that MPAs will conserve genetic diversity and prevent the extinction of rare alleles by protecting populations from selective fishing mortality (e.g. Palumbi 2003, Garcia-Charton et al. 2008), however, this

idea lacks thorough empirical testing. Surveys of queen conch (*Strombus gigas*) in the Bahamas suggested a fishing induced trait selection for small thick shelled individuals, relative to those in protected areas (Stoner et al. 2012b), though this selection was inferred by relative abundance of a non-selected trait. Studies that compared genetic diversity among protected and non-protected areas produced varied results. For example, significantly higher allelic frequency of white sea bream (*Diplodus sargus*) was reported in protected areas (Tabarca and Cape Palos-Hormigas Islands protected areas; Perez-Ruzafa et al. 2006), however, comparisons further down the coast found no significant genetic difference (Cerbère-Banyuls marine reserve; Lenfant 2003). Similarly, another study reported no significant difference in genetic diversity between protected (Cabrillo National Monument and Vandenberg Air Force Base) and nearby fished areas, for commercially harvested owl limpet (*Lottia gigantea*) (Fenberg et al. 2010). In all these examples, high gene flow relative to the spatial scale of the MPA characterized areas with no significant MPA benefit.

The potential of a marine protected area to conserve trait diversity depends, in part, on the dispersal range of the species. MPAs are assumed to offer little benefit for panmictic species beyond traditional fisheries management (reviewed in Section 1.1.1; Fenberg et al. 2010). For example, extensive movements of lobster across the boundaries of the Eastport MPA (Rowe 2001), coupled with the relatively small size of the MPA compared to the dispersal potential of the species (Section 2.2.3), limits the potential of the MPA to offer significant protection of genetic traits unless increased in spatial scale or nested within a larger network of MPAs. In contrast, genetic data of reef-dwelling species along the coast of South Africa revealed limited dispersal, irrespective of reproductive mode (e.g. brooders and broadcast spawners), increased the probability of genetic structure at a scale not encapsulated by the current MPA system (Wright et al. 2015).

The relatively large size (95th percentile globally, Figure 2) and limited exploitative history (Supplemental Figures 3-6) of the Laurentian Channel AOI suggest greater genetic differences could develop over time relative to unprotected areas. Higher genetic diversity could represent an indicator for the ‘protection from harm’ (Section 2.3-2.7)

objectives of the target species, except in wide ranging sea turtles and potentially porbeagle sharks. Higher relative genetic diversity in the MPA would indicate that it offers a potential tool to conserve the genetic variation of associated species through the cessation of mortality associated with human activity.

Section 3.2.5 Abundance indicators

For conservation objectives that focus on a single species, comparison of the number and size of individuals both within and outside protection offers the most pragmatic measure of MPA performance. Abundance can be measured explicitly (e.g. Costa et al. 2013), or as a function of space (density; e.g. Babcock et al. 2007) or effort (CPUE; e.g. Moland et al. 2013a). Abundance-based metrics are the most common indicator used to evaluate MPAs by far, and their application has been reviewed extensively (e.g. Lester et al. 2009, Stewart et al. 2009). Optimization of abundance monitoring depends on appropriate survey design, spatial scale(s) of the focal species/life history stage, the timescale of the expected response and selection of suitable reference locations (reviewed in Sections 3.2.1, 1.1.1, 1.1.2 and 1.1.3, respectively). In addition, the conservation objective itself can help formulate the most appropriate indicator.

The Eastport MPA conservation objectives focus on fisheries production. Spillover anticipated from the protected area to the surrounding locations provides the critical link, where reduced fishing mortality within the MPA produces a positive response in the focal species (lobster). Current indicators for the MPA focus on size distribution, movement rates, and abundance analyzed within a control-impact design. Size distribution and abundance both provide information on differences in population demographics under protection, and mirror numerous other studies that monitor MPA spillover. The combination of both indicators as a biomass weighted-CPUE (BWCPUE) can provide a different and potentially more robust metric. BWCPUE is a weighted mean calculation calculated as:

$$\bar{x} = \frac{\sum_{i=1}^n x_i w_i}{\sum_{i=1}^n w_i}$$

Where w refers to biomass, x denotes abundance of organisms within the size class and i refers to specific sample⁵. For example, in the Columbretes Islands Marine Reserve (CIMR), 17 years of MPA protection produced a net benefit of 10% increased landed catch of spiny lobster (*Palinurus elephas*) in weight (Goni et al. 2010b). Because large individuals tended to be preferentially captured after migrating from the park, this use of abundance in isolation masks the true output to the fishery. Importantly, Goni et al. (2010b) evaluate not only the difference between the MPA and surrounding areas but also the surplus production, thus providing real insight into not just MPA performance, but also how much it contributes to the fishery.

Evaluating MPAs based on anticipated larval spillover is a complicated process with few convincing published examples (see Table 1 and Section 1.1.1). Most studies evaluate the reproductive potential of the MPA relative to unprotected areas. In Eastport, skewed size distributions suggest larger lobsters within the MPA. Given known relationships between lobster size and fecundity (e.g. Currie et al. 2008) this size distribution could be combined with lobster abundance to estimate productive capacity. Logbook data and at-sea sampling can provide an index of the relative size and abundance and therefore production outside the MPA. The average relative production per unit area (PPUA) can be calculated using these two data sources and compared between the two protected zones of the MPA and the surrounding Eastport lobster management area. MPA performance could be then evaluated as the average difference in PPUA with a control-impact analysis (Section 3.2.1). Additionally, evaluation of production in a relative sense can help gauge how much the MPA contributes to the local production and how this extra production compares to area lost to fishermen (sensu Goni et al. 2010b). In the CIMR, reproductive output from spiny lobster increased by an average of 41% per year of protection, representing 80% of the reproductive potential arising from just 18% of the region. A similar calculation for the Eastport MPA through time would provide a powerful indicator of how the MPA contributes towards larval spillover, relative to the larger area as a function of changing management plans. To date this calculation

⁵ Note that calculation of BWCPUE as presented assumes that catchability does not trend with organism size.

has been made once, noting the Eastport MPAs represent ~10% of the production of the assessment area, though they cover ~1.8% of available habitat (Ennis et al. 2003). This preliminary calculation deserves further attention as an indicator; in particular it should be expanded to account for increases in egg production attributable to v-notching or minimum legal size to avoid overestimating the relative contribution of the MPA.

The Gilbert Bay MPA focuses internally with a conservation objective and associated indicators to monitor the endemic ‘golden cod’ population. In terms of abundance indicators it is difficult to establish an appropriate reference point by which to gauge these internal measurements. The limited movement of cod beyond the MPA boundaries (Morris et al. 2014) generally precludes a control-impact analysis except when applying a more general indicator such as biodiversity (Section 3.2.2) or functional diversity (Section 3.2.3) that may help support the local cod population. However, limited tagging suggests that adult cod stray from the MPA boundaries (Morris et al. 2014), and a more extensive tagging program could quantify the number and biomass of fish leaving the MPA relative to the estimated population within. This analysis would be analogous to monitoring programs that quantify spillover, except in this case limited spillover (as opposed to increased spillover over time) would produce a positive result. As in the Eastport example, the spillover of biomass from the CIMR (Goni et al. 2010b) provides a useful model. Given the relationship between cod size and fecundity (Kjesbu et al. 1991), calculating the approximate biomass of cod that leave Gilbert Bay and thus may be exposed to fishing mortality, relative to those that remain within the MPA, could provide a powerful covariate to model cod population dynamics within the MPA. The relative reproductive “escapement” could provide an indicator to quantify the real threat posed by fishing outside the MPA to the main MPA objective.

The Laurentian Channel conservation objectives vary more in terms of species-specific biology (Sections 2.3-2.8). Monitoring for the proposed AOI, relative to the conservation objectives, fall between Eastport and Gilbert Bay. Laurentian Channel monitoring is neither designed nor proposed to produce a spillover effect for fishing, but specific objectives pertaining to species that extend well beyond the AOI boundaries point to the need for monitoring at external

reference locations (see Section 3.2.1) is appropriate. The previous sections (3.2.2-3.3.4) outlined several ways to assess the AOI relative to unprotected areas; these metrics, combined with single species abundance, biomass, or density comparisons provide a useful template for potential indicators.

Section 3.2.6 Potential design pitfalls

Monitoring programs seek to evaluate how the management action (presence/absence of protection) influences a biological metric of interest. Although a simple inside-outside or “control-impact” (Osenberg et al. 2010) approach can and has (e.g. McClanahan et al. 1999, Halpern 2003) been used to evaluate MPAs, it ignores the considerable spatial and temporal variability typical for marine systems (Miller & Russ 2014). Environmental factors such as time of day, temperature, season, and habitat can all contribute to bias and variability in a measured biological trait, if not accounted for. Effective monitoring programs deal with this variability by explicitly incorporating such variation into statistical models, or through randomisation to negate external confounding effects (Willis 2013). For example, the Department of Fisheries and Oceans Canada utilizes random stratified trawl surveys to account for potential artifacts of spatial location within depth strata, among other potential sources of bias. Randomization, in this sense, is more cost effective than measuring and accounting for external influences in statistical models, however, it may miss important covariates that mediate biological response.

Accounting for external influences on the biological response is important. For example, fish movement during all life histories typically varies with time of day, season, habitat (e.g. juvenile cod; Shapiera et al. 2014), and in response to environment (e.g. temperate larval fishes; Stanley et al. 2012). Sampling that ignores sources of variability in the movement of individuals risks incorporation of bias over time, both within and among life history stages. For example, the Eastport MPA is monitored and assumed to be a potential source of larval lobster for the broader management area. Lobster larvae vary their vertical position in the water column in response to light and ontogeny (Stanley et al. in prep-b). Any monitoring program that measures concentrations of larval lobster would require

sampling at a consistent time of day to avoid bias. Acoustic monitoring for Greenland cod (*Gadus ogac*) revealed highly mobile juvenile stages where movement ranges varied with time of day (D. Schornagel, pers. comm.), season, and in response to temperature (Shapiera et al. 2014). As with larval lobster, monitoring programs evaluating biomass or abundance of juvenile cod would require sampling consistency for time of day and season. New Zealand monitoring programs designed to evaluate the efficacy of marine reserves for enhancing density and abundance of Australasian snapper (*Pagrus auratus*) demonstrate that because of seasonal migrations, spring and fall sampling would be most efficient (Willis et al. (2003a). During summer Leigh Marine Reserve populations included non-resident fish, and therefore did not represent the true reserve population (also see case study 4; Willis 2013). Similar movement patterns occur in Gilbert Bay Atlantic cod, where adults migrate beyond the boundaries of the reserve during summer (Morris et al. 2014). Given this behaviour, Gilbert Bay managers should follow the example of New Zealand snapper monitoring (Willis et al. 2003a), and should consider temporal variability in adult cod abundance when designing and implementing monitoring.

These examples highlight how considerable effort and resources may be wasted if monitoring programs do not specifically plan for comparability (Katsanevakis et al. 2012). Particularly when developing time-series, a balance must be struck between changes needed for effective monitoring (i.e. gear or effort) and how those changes may affect comparability with previous methods. Lack of consideration for comparability of monitoring methods can jeopardize continuous long-term datasets. Given the long lags (>20 years) observed in response to protection (e.g. Shears & Babcock 2003, Shears et al. 2006, Russ & Alcala 2010; reviewed in Section 1.1.2) long-term datasets can be invaluable. DFO random stratified survey data comprise the historical monitoring data for fish assemblages in the Laurentian Channel AOI and surrounding area. In 1996 sampling gear switched from an Engel 145 otter trawl to a Campelen 1800 shrimp trawl in an attempt to capture an increased percentage of small bodies fishes, including juvenile stages of groundfish and associated prey species, capelin (*Mallotus villosus*) and sandlance (*Ammodytes sp.*). Because catchability of species and size ranges typically vary with gear type, a series of comparative trawls were conducted to estimate

conversion coefficients and ensure the continuation of an inter-comparable long-term dataset (Warren et al. 1997). The large disconnect between 1995 and 1996 converted data raise into question the success of these conversion experiments in maintaining the data-series (Bratney et al. 1999). Similarly in a review of New Zealand MPAs, Willis (2013) noted that inconsistencies in monitoring methodologies, in large part resulting from no clear directive among researchers and managers, led to a broad lack of inter-comparability among data sources. The disparate methodologies and data sources used to monitor and assess MPAs in New Zealand reduced the ability to develop a robust consensus statement for MPA efficacy. These experiences illustrate the potential costs and statistical issues associated with changing monitoring programs.

Few marine protected areas are placed randomly. Habitat quality, focal species (e.g. Gilbert Bay), spawning stock (e.g. Eastport) or areas with the best ratio of cost to benefit (e.g. Marxan; Delavenne et al. 2012), often define the placement of MPAs. The ability to monitor MPA performance relative to non-protected areas is predicated on the availability of suitable reference sites with similar habitat and exploitation histories (see Section 1.1.3) (Miller & Russ 2014). Habitat differences between MPAs and reference sites in particular can confound objective comparisons, particularly for species with strong habitat associations (e.g. healthy coral habitat for reef fishes) (Osenberg et al. 2010) and habitat-mediated movement patterns (Gruss et al. 2011). For example, placement of an MPA *a priori* on habitat superior to available reference sites, would inflate biological response over time in the absence of a well replicated BACI design or other means to consider habitats (Section 3.2.1). The degree of monitoring bias associated with not accounting for habitat differences will vary based on strength of habitat association (Osenberg et al. 2010), however, habitat versus management actions may be difficult to disentangle (Miller & Russ 2014). In the Santa Barbara Channel Islands reserve network protection and habitat both influenced abundances of different life history stages of California spiny lobster (*Panulirus interruptus*) (Kay et al. 2012). Given that specific habitat types partly defined reserve locations (Airame et al. 2003), incorporating habitat explicitly into statistical models helped mitigate any bias that might dilute or exaggerate the influence of protection (Osenberg et al. 2010). Similarly, changes in the placement of reference sites for

the Tuhua Marine Reserve, New Zealand, onto sand habitat complicated MPA evaluation. Because reef fish naturally associate with reef habitat the ‘reserve effect’ measured relative to sub-optimal sandy habitat greatly exaggerates the influence of protection and the value of the monitoring program (Willis 2013). A positive effect of protection on community composition of reef fish species in the Banyuls Cerbère Marine Reserve (BCMR), France, was only statistically discernible when the model accounted for habitat type (depth). A depth stratified analysis of fish community found little variation in community composition but significantly higher abundances in the BCMR (Bell 1983). The incorporation of relative occupancy models for different habitat types can account for habitat variability and thus improve the value of control-impact analyses (Osenberg et al. 2010, Miller & Russ 2014).

Given the scale and diversity of habitats of the Laurentian Channel AOI, the application of habitat in assessment models will likely require a diversity of sampling approaches, therefore increasing the complexity to monitoring. Relative to examples provided, the application of habitat in control-impact assessments or the Laurentian Channel AOI will invariably be coarser, potentially divided into monitoring zones or broad habitat types (e.g. pelagic, benthic and shelf; Section 1.1.3). However using sampling data without some *a priori* understanding of the habitat within the system runs the risk of exaggerating the magnitude, or lack thereof, an effect.

Bioequivalence testing provides a method to assign biological significance in highly variable systems, where environmental “noise” negates the possibility of the null hypothesis of no difference to be true. This method assumes an effect is biologically significant only when it exceeds an *a priori* threshold (i.e. 100%), in light of uncontrolled variability in the system. Based on this method, Willis et al. (2003b) argued that few studies reporting a statistical difference, ignoring any design issues, would detect a difference using a 100% difference criterion. The *a priori* significance threshold will vary depending on the MPA, measured biological response, and what is considered a biologically and/or ecologically significant effect. For example, changes in fish size in the Maria Island Marine Reserve, Australia were detectable with an effect size of only 10%, whereas abundance alone would require a much higher effect

(~100%) to assign a significant difference (Edgar & Barrett 1997). Hilborn et al. (2004) argue that studies reporting MPA benefits are often equivocal because they lack sufficient replication and ignore natural variability when comparing MPAs and non-protected reference sites. When variability cannot be fully controlled by spatial and temporal replication, bioequivalence offers conservative approach to help guard against statistical error.

Power analysis tests provide an opportunity to utilize existing data to determine objectivity the magnitude of biological response or sampling effort required to assign a significant difference between protected and control areas. Power analysis can provide important information on how much ‘signal’ (e.g. change in abundance) or sampling effort would be required to detect a significant change in light of the variability ‘noise’ of sampling. Power analyses can become particularly important when gauging the probability of a monitoring program to be able to detect an effect given a level of sampling effort. Generally the strength of the response, relative to variability, scales inversely to the spatial and temporal timescales required of sampling to measure the response. For example, the length of time and amount of sampling required to detect significant change in abundance of the long-lived migratory leatherback turtle would be significantly greater than that for smooth skate which spend more time, and are more numerate, within the Laurentian Channel AOI. Power analysis based on Laurentian Channel survey data could provide important information about variability and direct the most efficient allocation of resources to sampling. If a system is under sampled or the response is too small then the monitoring program runs the risk of not being able to tell whether any change (positive, negative, or neutral) has occurred.

In short, the success of a monitoring program depends on whether it incorporates or accounts for sources of variability in the system. Ignoring temporal or spatial variability in a system introduces ambiguity in the monitoring framework and detracts from any conclusions drawn from monitoring data.

Section 3.3 Research opportunities in the Laurentian Channel AOI

Beyond their use in management, monitoring programs for protected areas provide a mean to test ecological hypotheses, assess the impact of anthropogenic disturbance, and more fully evaluate the function of ecosystem components underlying ecological processes. Research programs implemented in the Gilbert Bay MPA, for example, have been used to identify factors affecting the trends observed by the monitoring program, and thus have provided key scientific information required for adaptive management. MPAs provide a tremendous opportunity for research and education (Airame et al. 2003). Marine reserves offer numerous potential scientific benefits and MPAs can be as important to marine science as clean apparatus is to chemistry, and for the same reason (Christie & White 2007).

To date, six primary research objectives have been developed for the Laurentian Channel AOI (outlined in Lewis et al. 2014; Supplemental Table 5). The majority of these research objectives focus on understanding the focal species (Section 2.3-2.8) as well as biological attributes of the proposed MPA. These objectives have been well developed with justifications and indicators developed for each. While these research objectives broadly provide critical information about the ecosystem (habitat and plankton dynamics; objectives 1:3; Supplemental Table 5) without the need for invasive sampling methods, they do not provide information on community responses of organisms to the absence of human disturbance, particularly fishing.

Unfished areas represent repositories of complete ecosystems and therefore provide an opportunity to understand how ecosystems change naturally and respond to environmental variables (e.g. temperature change). In particular, large protected areas such as the Laurentian Channel AOI, present an opportunity to evaluate trophic interactions with a less altered natural food web and develop strategies to monitor biological attributes of larger individuals not present in the fished areas (see Section 1.1.1). In this sense MPAs offer an experimental instrument to gain baseline information to understand how marine communities respond and recover from human activity (Preuss et al. 2009). For example, the United States Pacific Fisheries Management Council (PFMC 2015) and the New Zealand Department of Conservation

(Willis 2013) both emphasize the capacity of protected areas to serve as controls for assessing long-term environmental variation and potential habitat alteration associated with fishing, as specific objectives of their respective MPAs.

Here we propose eight additional potential research opportunities (Table 9) for the Laurentian Channel AOI and detail the information they can provide to fisheries and ecosystem management, in addition to increased understanding of marine ecosystems of Atlantic Canada. These potential of each research opportunity will vary among species and need not be confined to species specifically mentioned in the conservation objectives (Section 2). Methodologies and utility of each measure is provided in Sections 3.3.1 - 3.3.6. **Table 9.** List of proposed research objectives, potential monitoring plans, and potential data sources. TS = trawl survey, ROV = ROV surveys*, LL = long-line surveys (sharks), and EM = environmental monitoring. Supplemental Table 4 includes a sample functional database.

ID	Proposed research objective	Monitoring	Data source
1	Community and functional change as a function of time, habitat and environment	Yearly trends in species and functional diversity as a function of depth and habitat strata (e.g. rocky, soft or biogenic bottom)	TS, ROV, EM and functional databases
2	Growth rates of large fish	Yearly changes in the size distribution, age-size relationships from otoliths	TS and otolith sampling
3	Impact on fishing on genetic and functional diversity of fish assemblages	BACI comparisons of functional and genetic diversity	TS and genetic samples
4	Impact of fishing on size of sexual maturity	BACI comparisons of average size of species previously captured directly or indirectly by fisheries	
5	Estimating natural mortality	Yearly changes in size distributions and abundance	TS, LL, and ROV
6	Estimating fishing mortality	Sum of mortality outside the reserve – mortality within	
7	Estimating natural behaviour	Changes in spatial distribution and habitat associations of focal species over time	
8	Density thresholds	Evaluation of the natural density of a given species in a unit area and habitat type	TS, LL, ROV, and EM

* Note that the abbreviation ROV in this table is considered to represent surveys with various instrumentation including fixed station moorings, autonomous underwater vehicles (AUV), or tethered remotely operated vehicles.

Section 3.3.1 Community responses to fishing activity

Many past studies have documented the influence of selective mortality on marine communities in ecological (e.g. Myers & Worm 2003, Worm et al. 2006) and protected area (see Section 3.2.2) literature. For the Newfoundland and Labrador region, the Laurentian Channel AOI could provide a tool to evaluate population processes and associated communities of groundfish species (e.g. *Gadus morhua*) in the absence of human-induced mortality. Comparisons of community structure and function in the absence of fishing could provide managers an important means to understand the influences of fishing beyond the target species, and more importantly understand how fishing could alter ecosystem function. In addition, differences in community structure between the Laurentian Channel and fished areas could help to explain the surprisingly slow recovery of groundfish species after the establishment of the moratorium (e.g. Myers et al. 1997). Community structure may change over time and, in some cases, might not fully stabilize for decades or longer (reviewed in section 1.1.2). However, monitoring sequential changes in diversity (species and functional; Sections 3.2.2, 3.2.3) through a program such as that proposed in Section 3.2.1 (Figure 19) would provide an index of how an ecosystem could recover and how they differ from exploited systems.

A substantial literature details the effects of fishing on seafloor biota. However, few studies evaluate fishing-related effects relative to a control or reference unfished area. Though effects of fishing effort are sometimes unambiguous (e.g. major declines in abundance or target and non-target species), the magnitude of the effect could potentially be underestimated without a suitable reference point. Recent analyses of groundfish survey data in NAFO divisions 2J3KLNO demonstrate significant changes in phylogenetic and functional diversity leading up to the collapse of ground fish stocks (Pedersen et al. in prep). Considering the slow recovery of Atlantic cod, comparing how communities change in protected vs. unprotected areas could provide managers information on how permissible ocean activity (e.g. non-moratorium fishing activity) could affect the natural recovery of communities.

Changes in the relative abundance of community constituents generally alter predatory-prey dynamics (Guidetti 2006). Continued monitoring of marine reserves could therefore be used as form of large-scale manipulation of predatory density, and thus assist in developing predictive predator-prey models. For example, studies of fish communities associated with kelp (*Echlonia radiata*) forest habitat inside and outside the Alphabet Bay and Tabletop Marine Reserves (New Zealand), suggest that fishing targeting larger-bodied predatory species leads to large-scale positive effects on the abundance and diversity of small cryptic reef fish species (Willis & Anderson 2003).

The successful application of this research objective will be predicated on a specific understanding of the relative change invoked by the establishment of the MPA on the marine fish community. The limited fishing activity, relative to unprotected areas, prior to establishment (reviewed in Section 1.1.3 and Supplemental Figures 3-6) potentially indicates areas within the AIO which are not representative of those typically exploited by fisheries. Candidate areas for community comparisons must be selected based on similarities in habitat availability and groundfish communities prior to MPA establishment (e.g. through DFO groundfish surveys).

Section 3.3.2 Growth rates of large-bodied organisms

Fishing activity imparts skewed size distributions that complicate estimation of growth parameters of species in fished areas. Fishing can impact growth indirectly through habitat alteration and directly through the selective removal of faster-growing individuals (Enberg et al. 2009). MPAs sufficiently large to capture the spatial scale of a focal species home range (Section 1.1.1) can potentially protect species throughout their life history, and are therefore more likely to contain a complete size range. For example, cessation of fishing activity within Georges Bank permitted tracking of Atlantic sea scallop (*Placopecten magellanicus*) cohorts over longer time periods than fished areas. Increased temporal resolution in tracking cohorts provides a more robust framework to develop von Bertalanffy growth parameters (Hart & Chute 2009). Similarly, the successful estimation of growth parameters for American lobster was predicated on

modelling the full species size range. In particular, the inclusion of the maximum recorded size of lobster, which are rare in fished areas, provides more realistic growth estimates (Raper & Schneider 2013).

The relatively large size of the Laurentian Channel AOI (Figure 2) creates strong potential to protect a wide range of species through their adult life history stages. The AOI would likely protect adult northern wolffish and black dogfish (Sections 2.5 and 2.7 respectively) for example, because of limited expected movement. Provided that age could be determined from otoliths, a size-at-age model, such as von Bertalanffy growth could utilize a subset of captured individuals from trawl or long-line surveys (e.g. Hart & Chute 2009, Raper & Schneider 2013). Accurate assignment of growth can aid management in estimating exploitation rates and assigning size thresholds for harvesting based on maximum yield or egg-per-recruit models (Raper & Schneider 2013).

Section 3.3.3 Influence of fishing on size of sexual maturity

Fisheries can also indirectly influence the size of sexual maturity by removing individuals that reach sexual maturity later in life or at larger sizes. Decline in the size at sexual maturity has been attributed to this selective process for multiple species (e.g. Rijnsdorp 1993). Estimating and comparing maturity ogives (proportion mature as a function of age or size) between protected and unprotected areas can provide a relative index of how fishing activity influences reproduction.

Changes in size at sexual maturity will be most pronounced for those species influenced by fishing activity, both targets and bycatch. Genetic exchange from unprotected areas (e.g. larval recruitment) and concomitant changes in environmental conditions which influence reproduction, could confound the interpretation of any change in size a sexual maturity relative to the management action. It would be expected that any change in the size at sexual maturity would manifest over an extended period of time (decades), permitting multiple generations under differential selection.

Section 3.3.4 Influence of fishing on genetic diversity

Fisheries that exploit specific size ranges can potentially artificially select certain traits (e.g. growth and size at maturity; Section 3.3.2 and 3.3.3) and therefore reduce the genetic diversity of exploited populations (reviewed by Kuparinen & Merilä 2007). Assuming the MPA eliminates this selective pressure, and environment remains relatively constant, it will presumably maintain comprehensive trait ranges and potentially maintain genetic diversity lost in targeted populations (reviewed in Section 3.2.4). Comparisons of fished and suitable unfished reference areas based on genetic diversity (e.g. allelic frequency) as a response variable, (see Section 3.2.1 for monitoring designs) can aid in understanding how fishing influences trait diversity and predicting how fished populations may respond to stress (i.e. climate change) or potentially recover with reduced fishing effort.

Section 3.3.5 Estimating natural and fishing mortalities

Stock assessments and population models include instantaneous rates of mortality as key parameters. Estimates of natural rates of mortality (NM) vary with the quality of collected data. Ideally the sampled population will contain a complete size distribution of all associated predators and prey of a complete size distribution, and thus represent a “natural” population with minimal human perturbation (Macpherson et al. 2000). MPAs can potentially meet these criteria. A reliable estimate of NM also provides an opportunity to calculate fishing mortality according to:

$$\text{Fishing mortality} = \text{total mortality} - \text{natural mortality}$$

where *total mortality* represents mortality estimated from a fished population and *natural mortality* is the mortality rate assumed of a species or population in the absence of exploitive pressure (Macpherson et al. 2000). For example, natural mortality rates calculated from Spain’s fully protected Medas Islands Marine Reserve provided an opportunity to calculate natural mortality rates for common littoral fishes in the Mediterranean Sea (*Coris julis*, *Diplodus annularis*, *D. sargus*, *Serranus cabrilla*, and *Symphodus roissali*) (Macpherson et al. 2000). Similarly, instantaneous mortality rates for juvenile fishes of the genus (*Diplodus*) were equivalent between protected and unprotected areas, suggesting that

despite higher abundance of predators, marine reserves in the northern Mediterranean were not necessarily sinks for juvenile fishes (Macpherson et al. 1997).

Calculation of mortality depends on available data and the species in question. Mark recapture-methods can provide abundance and mortality indices, but are likely impractical for benthic species captured by trawl or long-line, because of high sampling mortality. However, mortality can also be calculated from abundance data. For example, abundance data from trawl surveys could be used to fit an exponential mortality model assuming availability of a reliable size-at-age model (sensu Macpherson et al. 1997, Macpherson et al. 2000). With estimates of numbers-at-age, mortality can be calculated according to:

$$N_t = N_0 * e^{(-z*t)}$$

where N_t is the number of individuals at age t , N_0 is the number of individuals at age 0 and z is the estimated mortality rate.

The use of trawl surveys to calculate natural mortality would require consideration of the mortality induced by each trawl set. The application of abundance-based estimates of mortality would be appropriate assuming small sampling mortality relative to the underlying variation in the system. Otherwise the model would have to include sampling mortality. For species that range far beyond the boundaries of the MPA (e.g. leatherback sea turtle and porbeagle shark; sections 2.4.3 and 2.8.3) uncontrolled mortality in unprotected areas would confound estimates of natural mortality within the MPA.

Section 3.3.6 Estimating density thresholds

Marine reserves reduce mortality, support more complete trophic structures, provide better habitat and subsequently generally support higher abundances of organisms (e.g. Abesamis et al. 2006, Babcock et al. 2007). In this sense, marine reserves can be treated as a large-scale density manipulation to compare how density impacts population

parameters such as mortality (Section 3.3.5), growth (Section 3.3.2) and behaviour. In addition, habitat-specific density thresholds can be calculated over time to estimate the carrying capacity of a given habitat type (e.g. biogenic coralline habitats). These carrying capacities can be used to evaluate the relative value of habitat within and outside the MPA for associated species or communities of species. Optimizing any future placement of protected areas in the Newfoundland and Labrador region will require understanding the capacity of habitats to support focal species.

Section 4 – Summary and conclusions

Marine protected areas globally have been established to address a wide range of objectives including biodiversity protection, conservation of threatened species, fisheries enhancement, and replenishment of depleted fish populations. Although MPA advocates generally predict net positive effects of MPA closures (e.g. Halpern & Warner 2002, Lester et al. 2009), others suggest that not all biological responses to protection result in systems ‘better’ than they were prior to the management intervention (e.g. Verhulst et al. 2004, Agardy et al. 2011). Concluding whether the state of the marine system has changed, and whether that change is positive, in response to protection requires a foundation of clear and testable objectives and sound monitoring.

Monitoring in the context of protected areas represents a form of observational research that documents variability in natural systems by comparing them with manipulated systems over time (Willis 2013). Sound monitoring must consider movement scales of the organisms (Section 1.1.1), temporal scales over which a change may occur (Section 1.1.2), and consider the condition of the system prior to protection (Sections 1.1.3, 1.1.4, and 3.2.1). Current monitoring programs for the Eastport and Gilbert Bay MPAs, and those proposed for the Laurentian Channel AOI (e.g. Lewis et al. 2014), provide a framework for evaluating MPA performance, however, there are limitations to these monitoring programs (summarized in Table 10). Specifically, design of monitoring programs (reviewed throughout

report and specifically in Section 3.2.6) must integrate considerations for species-specific biology (Section 2) and spatial-temporal scales of the response that are specific and related to the MPA.

The field of MPA science and monitoring globally evolves continually. Protected areas around the world with similar conservation objectives and design offer a useful template to re-evaluate the monitoring programs for NL MPAs (Section 3.1). Based on the detailed synthesis of MPA literature presented above, many monitoring efforts focus on habitats and communities that support target organisms. Similarly, statistical evaluations of biological change in response to protection typically incorporated large-scale environmental monitoring, either directly measured or incorporated into statistical evaluations of biological change in response to protection (e.g. BACI; Section 3.2.1). In many instances, broadening the focus of monitoring to incorporate indicators that identify specific targets would allow managers to partition natural variability in the system more objectively in order to evaluate the success of the MPA in achieving its objectives.

Table 10. List of established and proposed conservation objectives (COs) for protected areas in the Newfoundland region. Each CO links to a list of associated benefits, as well as limitations to its success and/or monitoring. COs are partitioned by protected area.

Conservation Objective	Benefits	Limitations
Laurentian Channel AOI		
<i>Protect corals, particularly significant concentrations of sea pens, from harm due to human activities</i>	CO is broad which focusses attention on multiple species of cold water corals.	Remaining oil and gas development poses an unknown threat to the viability of conservation and recovery benefits, however, reduced activity in the AOI should promote coral survival within the AOI. Wording does not highlight a definitive variable to be measured or level at which it could be assessed. Objective is somewhat ambiguous. What constitutes ‘significant’ and how much harm needs to be

		prevented for success?
<i>Ensure that human activities do not impair the reproduction and survival or disrupt important aggregations of Black Dogfish</i>	<p>Monitoring will be simplified by limiting attention to a specific species.</p> <p>Objective defines desired trajectory in that a recovery is prioritized.</p>	<p>Objective is ambiguous*. however, reduced activity in the AOI should promote survival within the AOI.</p> <p>Very little is known about the species and therefore it is difficult to predict the response to protection.</p>
<i>Protect areas of immature Smooth Skate abundance and ensure that human activities do not impair the reproduction and survival of the stock</i>	<p>Monitoring will be simplified by limiting attention to a specific species.</p> <p>Objective defines desired trajectory in that a recovery is prioritized.</p>	<p>Objective is ambiguous*, however, reduced activity in the AOI should promote survival within the AOI.</p> <p>Wording does not highlight a definitive variable to be measured or level at which it could be assessed.</p> <p>Response to protection could be limited as non-natural mortality was reduced prior to establishment.</p>
<i>Ensure that human activities do not impair the reproduction and survival of Porbeagle</i>	<p>Monitoring will be simplified by limiting attention to a specific species.</p> <p>Objective defines desired trajectory in that a recovery is prioritized.</p>	<p>Objective is ambiguous*, however, reduced activity in the AOI should promote survival within the AOI.</p> <p>Migratory behaviour potentially exposes adults to mortality outside the MPA, which could offset benefits provided by protection.</p> <p>Wording does not highlight a definitive variable to be measured or level at which it could be assessed.</p>
<i>Promote the survival and recovery of northern wolffish by minimizing risk of harm from human activities</i>	<p>Monitoring will be simplified by limiting attention to a specific species.</p> <p>Objective defines desired trajectory in that a recovery is prioritized.</p>	<p>By limiting activity the AOI should promote survival within the AOI, however, the location of the AOI is at the southernmost limit of the wolffish distribution and therefore continued trends of bottom water warming could change the distribution and limit coverage by</p>

		the MPA.
		Wording does not detail how much reduction in human harm is required or what the variable of interest is.
<i>Promote the survival and recovery of leatherback sea turtle by minimizing risk of harm from human activities</i>	Monitoring will be simplified by limiting attention to a specific species. Objective defines desired trajectory in that a recovery is prioritized.	Difficult to partition the success of protection from uncontrolled mortality, however, reduced activities in the area should promote survival within AOI boundaries. Response to protection could be limited as both threats to and presence of leatherbacks prior to establishment was limited. Wording does not detail how much reduction in human harm is required.

Gilbert Bay MPA

<i>Conservation and protection of the Gilbert Bay cod and its habitats</i>	CO is broad which focusses monitoring effort on both the species and associated habitat.	Although the MPA protects cod within its boundaries, the potential influence of uncontrolled mortality on adults that leave the MPA on population viability is unknown. Reduced activity in the AOI should conserve and protect habitats within the AOI. Declining trends in cod in the region could point to larger-scale environmental impacts which could mitigate success of CO. Wording does not provide a reference level or target by which to compare the measured response.
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Eastport MPA

<i>To maintain a viable</i>	CO is broad which	Given the small size of the MPA,
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<i>population of lobster through the conservation and protection and sustainable use of resources and habitats</i>	focusses monitoring effort on both the species and associated habitat.	<p>larval dispersal will likely explain any spillover. Measuring larval dispersal will be difficult.</p> <p>More information on larval dispersal and mortality required before increases in egg production can be used to measure the success of the MPA at reaching the CO.</p> <p>The biology of the species and the wording of the CO suggest that success will depend largely on management outside the MPA, which can limit the success of monitoring.</p>
<i>To ensure the conservation and protection of threatened or endangered species</i>	CO could be applied to multiple species with a threatened or endangered designation.	<p>Though no species is defined this objective likely focus on wolffish.</p> <p>The absence of information regarding the current population results in scale by which to gauge the success of management or to partition management effect from natural variability.</p> <p>Uncontrolled mortality by fishing is probable given the spatial scale of movement.</p>

*Does regulatory action limiting local mortality constitute success, or, does the species also have to maintain a positive demographic trajectory?

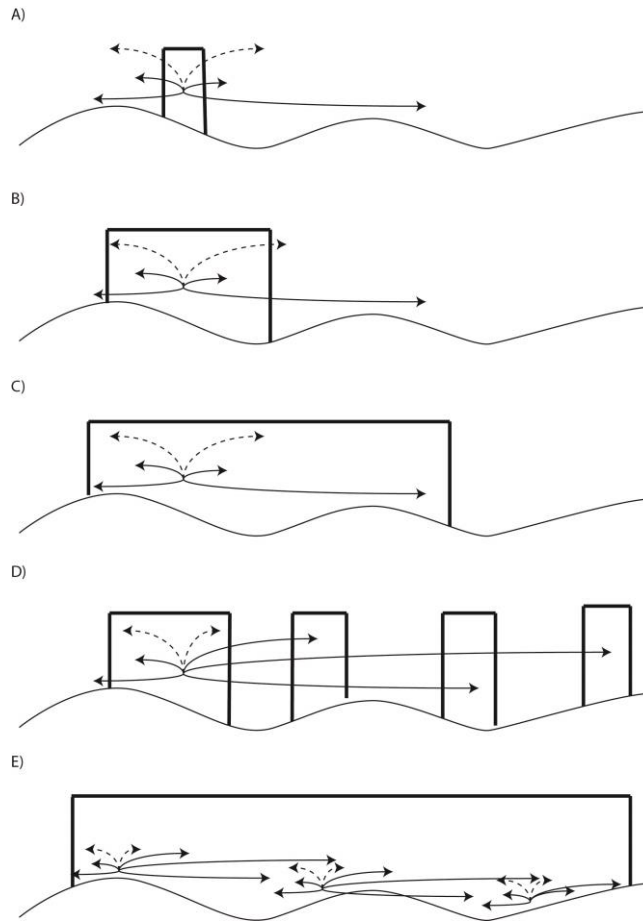
Relative to other MPAs, particularly in temperate waters, the Laurentian Channel AOI could be one of the largest no-take areas established on a continental shelf in the world (Supplemental Figure 2) and is among the largest 5% of MPAs globally⁶. Like other large protected areas (Supplementary Table 2), the Laurentian Channel AOI offers tremendous opportunity to protect marine species, habitats, and associated ecosystem function (reviewed in Section 3.3).

⁶ Based on analysis of listed MPAs within the IUCN Protected Areas Database. Refer to Supplemental R-Code for analysis details.

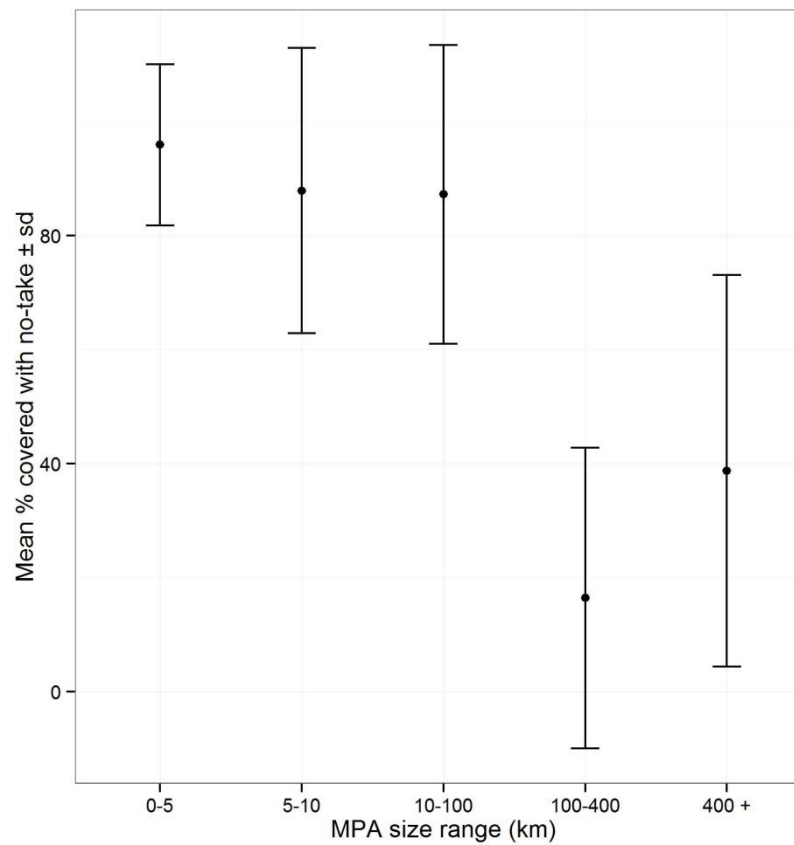
Intact ‘natural’ ecosystems offer a tremendous, and unfortunately rare, opportunity for research. In particular, the Laurentian Channel AOI creates the opportunity to estimate natural rates of mortality, population growth and community dynamics of coastal species, which is not possible in disturbed (fished) areas. Although Laurentian Channel AOI objectives do not focus on fisheries production, stock assessments and fisheries management will benefit immensely from the ability to evaluate population traits and rates of change in natural settings.

We present this review as a guide to contemporary marine protected area science and a template for the review of existing and proposed monitoring programs for MPAs in the NL region. More thorough and integrated assessments provide an essential step toward successful evaluation of marine protected areas, and ultimately will offer greater return on financial and human resource investments. Although this examination focusses on conservation objectives and designs similar to protected areas in the NL region, the background and insight for optimizing monitoring programs have broad applicability to current and future MPAs in Canada.

Section 5 – Supplemental material

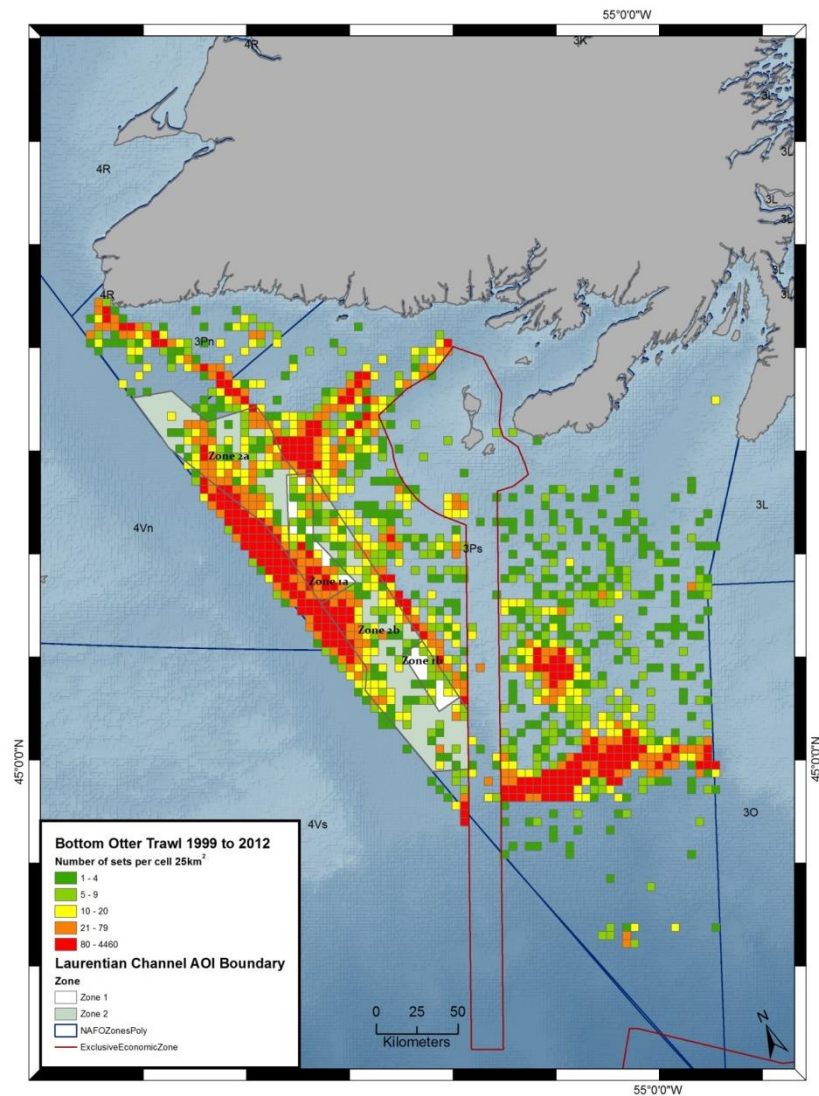


Supplemental Figure 1: Hypothetical models of MPA design (outlined by thick black bars) with scaling considerations for adult (dashed) and larval (solid) movement. A) Design where significant spillover of adult and larval stages leads to little self-replenishment and small local returns. B) Design of sufficient retention for persistence and moderate spillover, leading to local fishing benefits through adult and larval subsidies. C) Design with strong persistence and no loss to fishing, however, size does not permit spillover resulting in no benefit to surrounding areas. D) Design of multiple reserves at varying spacing protecting multiple dispersal distances, sufficient adult protection for persistence and provision of recruitment subsidy to fished areas E) Large scale MPA design to encompass multiple habitat, home-ranges and ecosystem components. Diagram modified from Halpern and Warner (2003) and movement principles from Palumbi (2004).

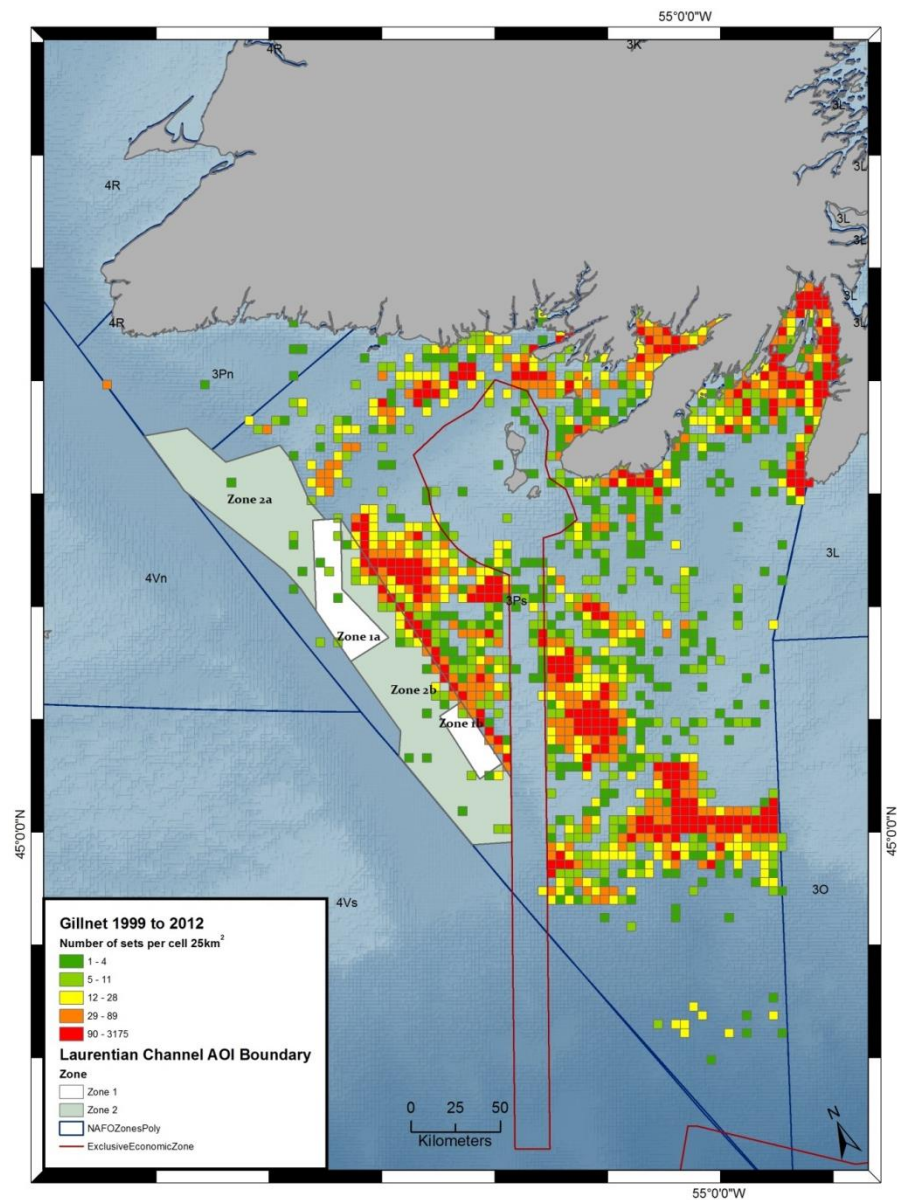


Supplemental Figure 2: Plot describing the percent spatial coverage of no-take areas relative to the size of the MPA. Data was obtained from the World Protected Area Database⁷ and subsetting to those MPAs with multi-use designs (no-take and take). Sample size varied among categories: 420 (0-5 km), 30 (5-10 km), 116 (10-100 km), 47 (100-400 km) and 213 (400 + km).

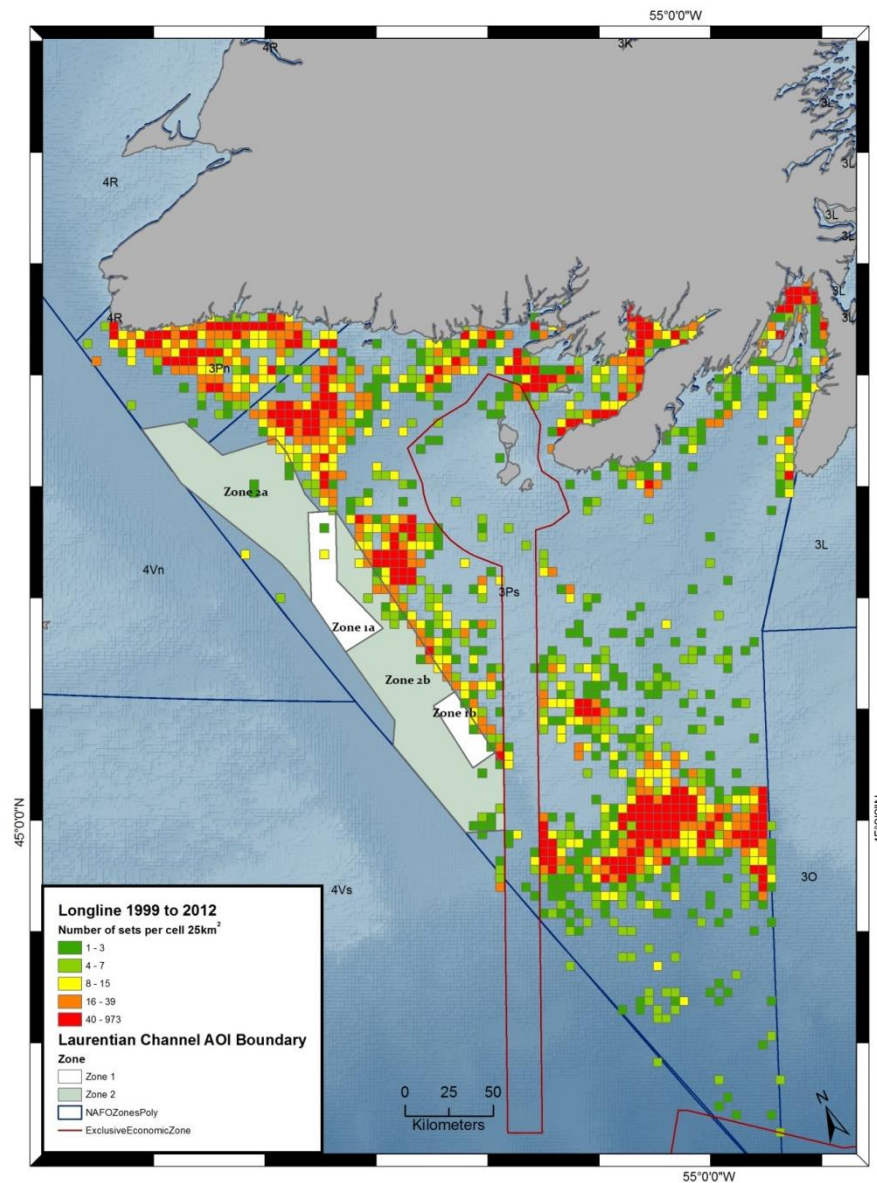
⁷ Data subsetting to management areas in the ocean classified as marine reserves, monuments, sanctuaries, parks or protected areas.



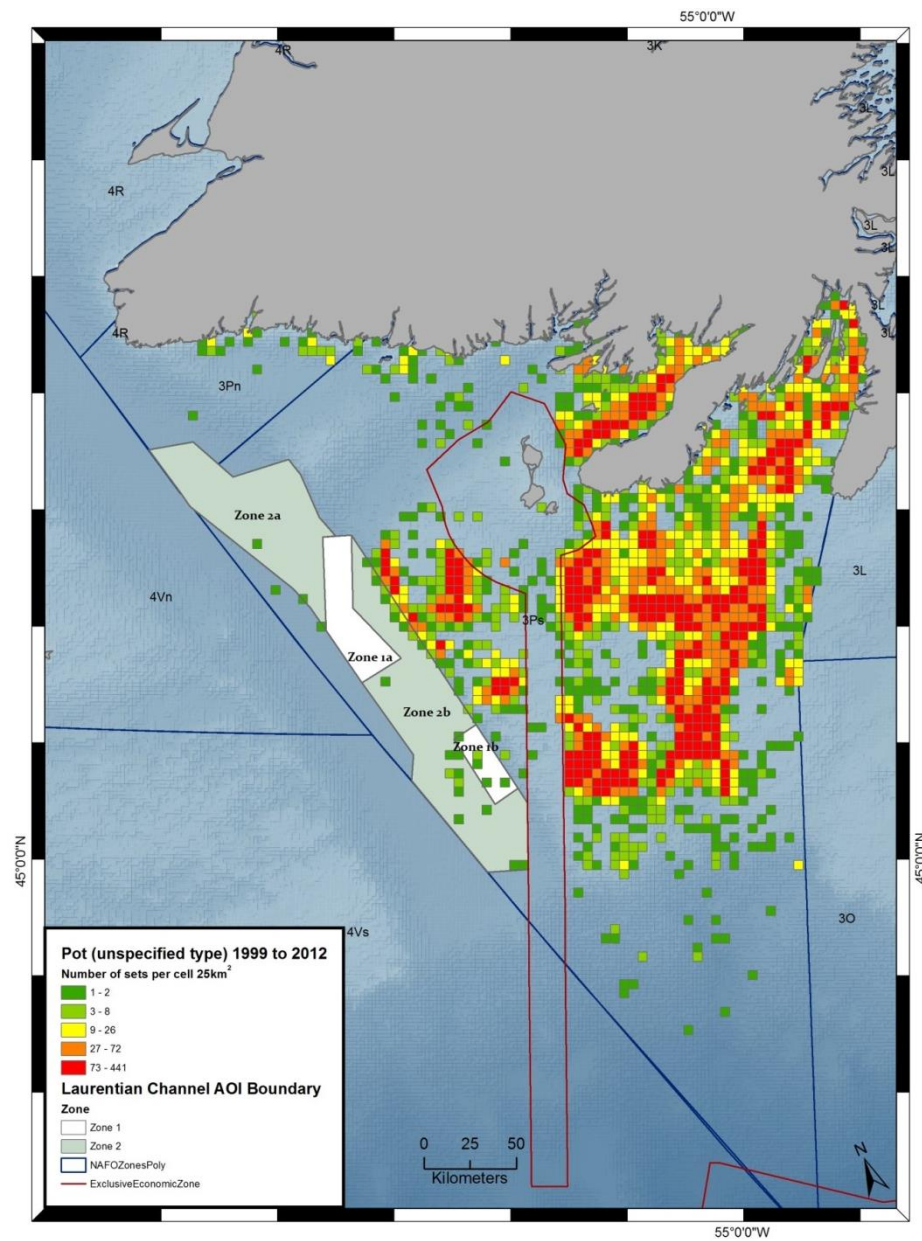
Supplemental Figure 3: Distribution of fishing activity near the Laurentian Channel AOI using bottom trawls from 1999-2012. Source: Fisheries and Oceans Canada.



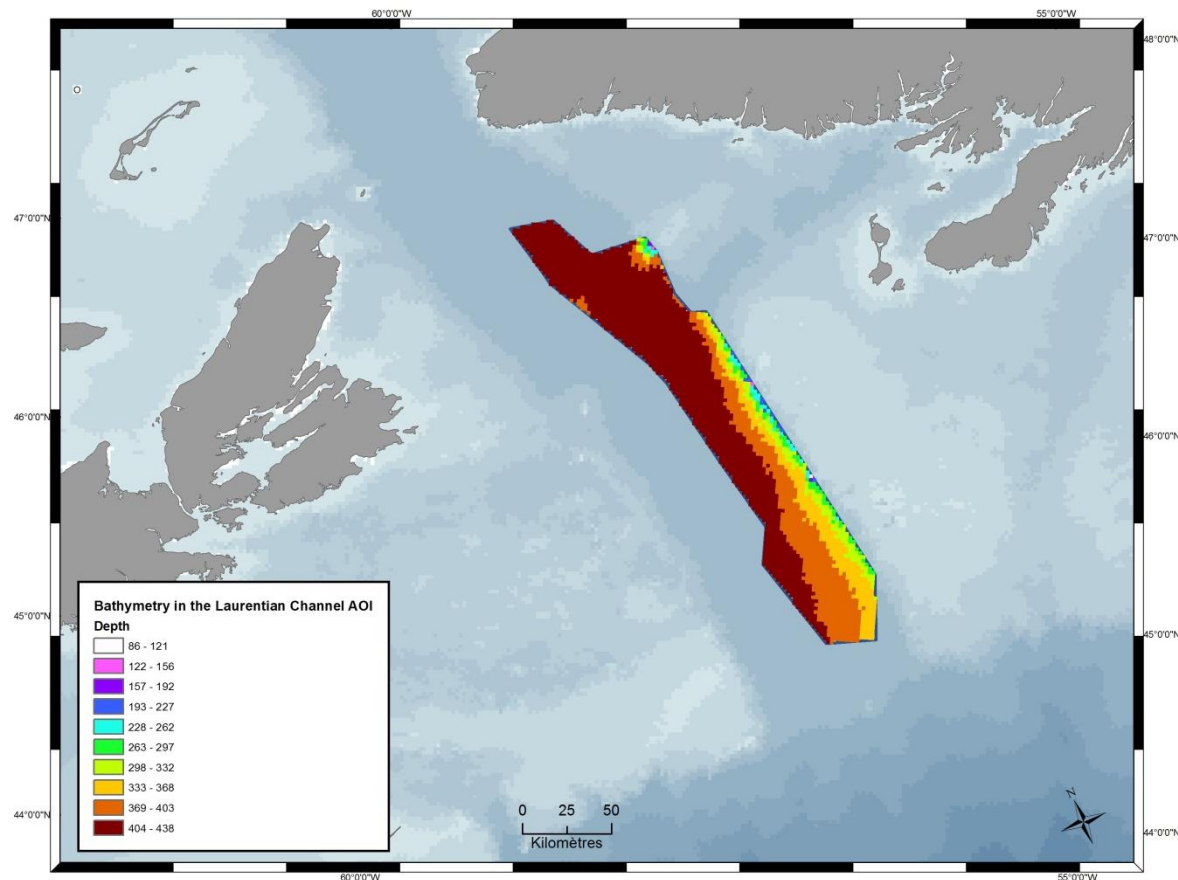
Supplemental Figure 4: Distribution of fishing activity near the Laurentian Channel AOI using Gillnets from 1999-2012. Source: Fisheries and Oceans Canada.



Supplemental Figure 5: Distribution of fishing activity near the Laurentian Channel AOI using longlines from 1999-2012. Source: Fisheries and Oceans Canada.



Supplemental Figure 6: Distribution of fishing activity near the Laurentian Channel AOI using fixed pot gear from 1999-2012. Source: Fisheries and Oceans Canada.



Supplemental Figure 7: Bathymetric features of the Laurentian Channel AIO. Source: Fisheries and Oceans Canada.

Supplemental Table 1. IUCN protected area categories (IUCN-WPCA 2008). Categories are divided into two broad classifications for those who limit human activity and those who manage based on multiple activities and use (grey shaded).

IUCN Category		Primary objective
Ia	Strict Nature Reserve	Strictly protected areas to protect biodiversity and possibly geological / geomorphological features. Human visitation, use and impacts are strictly controlled and limited to ensure preservation of the conservation values. These areas can serve as indispensable reference areas for scientific research and monitoring.
Ib	Wilderness Area	Large or slightly modified areas, retaining their natural character and influence, without permanent or significant human habitation, which are protected and managed so as to preserve their natural condition.

II	National Park	Large natural or near natural areas set aside to protect large scale ecological processes, along with the complement of species and ecosystems characteristic of the area, to provide a foundation for environmentally and culturally compatible spiritual, scientific, educational, recreational and visitor opportunities.
III	Natural Monument	Set aside to protect a specific natural monument, which can be a landform, sea mount, submarine caverns, geological feature such as caves or even a living feature such as an ancient grove. They are generally quite small protected areas and often have high visitor value.
IV	Habitat/Species Management Area	Protect particular species or habitats and management reflects this priority. Regular, active interventions often needed to address the requirements of particular species or to maintain habitats.
V	Protected Landscape/ Seascape	Where the interaction of people and nature over time has produced an area of distinct character with significant ecological, biological, cultural and scenic value; and where safeguarding the integrity of this interaction is vital to protecting and sustaining the area and its associated nature conservation and other values.
VI	Managed Resource Protected Area	Large, with much of the area in a natural condition and where a proportion is under sustainable natural resource management. Exploitation is a main aim of the area.

Supplemental Table 2. Distribution, size and proportion with limited access of ‘very large’ marine protected areas globally. This list is partially built on Toonen et al. (2013)’s Table 1.

Name	Country	Size (km ²)	% no-take
Phoenix Islands Protected Area	Republic of Kiribati	408,250	4
Great Barrier Reef Marine Park	Australia	344,000	33
Palau Marine Sanctuary	Palau	500,000	partial*
Coral Sea	Australia	989,842	51
Papahānaumokuākea Marine National	USA	362,074	partial*

Monument

Mariana Trench Marine National Monument	Common-wealth of Northern Mariana Islands, USA	246,609	100
Chagos	UK overseas territory	640,000	100
Motu Motiro Hiva Marine Park	Chile	411,000	100

* Reserve has a mixed management framework where some limited fishing is permitted

Supplemental Table 3. Examples studies which provide empirical data or analysis pertaining to positive biological responses to protection outlined in Figure 3. Note that these are selected examples and a subset of a much larger literature base. Emphasis was placed on providing studies with a diversity of global locations and focal species.

ID	Species or system	MPA	Citation
1	Spiny lobster (<i>Jasus edwardsii</i>)	Leigh Marine Reserve (New Zealand)	(Kelly et al. 2000, Kelly et al. 2002)
2	Sparid snapper (<i>Pagrus auratus</i>)	Hahei and Tawharanui Marine Reserves (New Zealand)	(Willis et al. 2003a)
3	American lobster (<i>Homarus americanus</i>)	Eastport MPA (Canada)	(DFO 2014b)
4	Lepard coral grouper (<i>Plectropomus leopardus</i>)	Central Great Barrier Reef (Australia)	(Ferreira & Russ 1995)
5	Various fish species	Scandola Marine Reserve (Italy)	(Francour 1991, 1994)
6	Spiny lobster (<i>Palinurus elephas</i>)	Columbretes Islands Marine Reserves (Spain)	(Goni et al. 2006, Goni et al. 2010b)
7	Scallop (<i>Placopecten magellanicus</i>)	George's Bank (US)	(Murawski et al. 2000, Lewis et al. 2001, Murawski et al. 2005)
8	Coral trout (<i>Plectropomus maculatus</i>)	Great Barrier Reef	(Harrison et al. 2012)

	and stripy snapper (<i>Lutjanus carponotatus</i>)	(Australia)	
9	Rock scallop (<i>Spondylus calcifer</i>) and black murex snail (<i>Hexaplex nigritus</i>)	Puerto Peñasco reserve network (Mexico)	(Cudney-Bueno et al. 2009)
10	Coral Habitat	Al Yasat (Qatar)	(Moati et al. 2008)
11	Coral Habitat	Cabo Pulmo National Park (US)	(Aburto-Oropeza et al. 2011)
12	Coastal habitat	Hawaii Islands Reserve Network (US)	(Friedlander et al. 2003)
13	Various fish species	Apo Islands Marine Reserve (Phillipines)	(Russ & Alcala 2011)
14	Various species	Simulation – meta-analysis	(Dichmont et al. 2013)
15	Various fish species	Bonifacio Strait Natural Reserve (Italy)	(Mouillot et al. 2008)
16	Various species	Hawaii Islands Reserve Network (US)	(Friedlander et al. 2010)
17	Mangroves and associated biota	Global review	(McLeod & Salm 2006, McLeod et al. 2009)
18	Coastal habitat and associated biota	Kimbe Bay MPAs (Papua New Guinea)	(Green et al. 2009a)
19	Juvenile Atlantic cod (<i>Gadus morhua</i>)	Mathematical Model and Stellwagen Bank National Marine Sanctuary (US)	(Lindholm et al. 2001)
20	Coral reef habitat	Philippine marine reserve networks	(Oracion et al. 2005)
21	Black drum (<i>Pogonias cromis</i>), Red drum (<i>Sciaenops ocellatus</i>), and	Soufrière Marine Management Area & Merritt Island National	(Roberts et al. 2001)

	Spotted seatrout (<i>Cynoscion nebulosus</i>)	Wildlife Refuge (US)	
22	Blue cod (<i>Parapercis colias</i>)	Kokomohua Marine Reserve (New Zealand)	(Davidson 2001)
23	Various fish species	Maria Island Reserve (Australia)	(Edgar & Barrett 1999)
24	Lingcod (<i>Ophiodon elongatus</i>) and copper rockfish (<i>Sebastes caurinus</i>)	Edmunds Under Water Park (US)	(Palsson & Pacunski 1995)
25	Queen conch (<i>Strombus gigas</i>)	Exuma Cays Land and Sea Park (Bahamas)	(Stoner et al. 2012b)
26	Chilean Abalone (<i>Concholepas concholepas</i>)	Las Cruces MPA (Chile)	(Manriquez & Castilla 2001)
27	Various benthic sessile species	Experimental protected plots (southwest Mediterranean)	(Bevilacqua et al. 2006)
28	Infuana of the brown mussel (<i>Perna perna</i>)	Dwesa and Hlueka MPAs (South Africa)	(Cole et al. 2011)
29	Various reef fish species including groupers (Epinephelinae) and snappers (Lutjanidae)	Mona Island MPA (Puerto Rico)	(Mateos-Molina et al. 2014)
30	Various fish species	Mediterranean MPAs: Cerbere-Banyuls & Carry-le-Rouet (France) and Medes, Cabrera, Tabarca, and Cabo de Palos (Spain)	(Goni et al. 2008)
31	Various rockfish species	Torre Guaceto Marine Reserve	(Guidetti 2006)
32	Spiny lobster (<i>Palinurus elephas</i>)	Columbretes Islands Marine Reserves (Spain)	(Diaz et al. 2011)
33	Benthic invertebrates and suspension	Currambene Creek no-	(Costa et al. 2013)

	feeders	take area (Australia)	
34	Biomass, size and egg production of spiny lobster (<i>Panulirus Cygnus</i>)	Kingsford Reef Marine Sanctuary (Australia)	(Babcock et al. 2007)
35	Size and abundance of Atlantic cod (<i>Gadus morhua</i>) and European lobster (<i>Homarus gammarus</i>)	MPAs of the Skagerrak coast (Norway)	(Moland et al. 2013a)
36	Increased abundance of upright Bryozoans and hydroids	Isle of Man scallop research closures	(Bradshaw et al. 2001)
*	General benefits assumed of MPAs which excluding fishing activity, generally IUCN categories I-III (Supplemental Table 1)		

Supplemental Table 4. List of functional traits assigned to species captured during Northwest Atlantic benthic groundfish surveys (NAFO 2J3KLNO; 1977-2013). Traits are assigned to species, and (subspecies), taxonomic level. Trait data is aggregated from Fishbase (Froese & Pauly 2015) and DFO species databases . VP = vertical position, DN = diet niche, DT = population doubling time (years), ML = maximum length (cm), TL = trophic level and AB = aggregation behaviour, L = large, M = medium, S = small, irreg = irregular. Database compiled for analysis of functional diversity in Pedersen et al. (in prep). *Please inquire with R. Stanley for methods or collection and use of this data.*

Species	VP	DN	DT	ML	TL	BA
<i>A. radiata</i>	demersal	L Benthivore	9.25	105	4	rare
<i>A. denticulatus</i>	benthopelagic	L Benthivore	9.25	180	3.8	rare
<i>A. lupus</i>	demersal	L Benthivore	9.25	150	3.2	irreg
<i>A. minor</i>	demersal	L Benthivore	9.25	180	3.5	irreg
<i>A. rostrata</i>	bathypelagic	M Benthivore	2.9	75	3.6	irreg
<i>A. atlanticus</i>	demersal	S Benthivore	9.25	15	3.4	rare
<i>A. monopterygius</i>	demersal	S Benthivore	2.9	18	3	none
<i>B. euryops</i>	bathypelagic	S Benthivore	2.9	13	3.3	irreg
<i>B. spinicauda</i>	bathydemersal	L Benthivore	9.25	170	4	irreg
<i>B. glaciale</i>	pelagic-oceanic	Planktivores	2.9	10.3	3	irreg
<i>B. saida</i>	demersal	PlankPiscivore	2.9	40	3.1	schools
<i>C. fabricii</i>	bathydemersal	Piscivore	9.25	107	3.9	irregular
<i>C. sloani</i>	bathypelagic	Piscivore	2.9	35	4.2	shoal
<i>C. niger</i>	bathypelagic	S Benthivore	2.9	25	4.2	none
<i>C. harengus</i>	benthopelagic	Planktivores	2.9	45	3.2	schools
<i>C. rupestris</i>	bathypelagic	M Benthivore	9.25	110	3.5	schools

<i>C. microps</i>	bathydemersal	S Benthivore	9.25	30	3.4	none
<i>C. lumpus</i>	benthopelagic	M Benthivore	9.25	61	3.9	irreg
<i>E. cimbrius</i>	demersal	S Benthivore	2.9	41	3.5	rare
<i>E. praecisus</i>	benthopelagic	S Benthivore	2.9	22	3.5	none
<i>E. spinosus</i>	demersal	S Benthivore	2.9	13.2	3.5	none
<i>G. ensis</i>	benthopelagic	Planktivores	2.9	40	3.4	none
<i>G. morhua</i>	benthopelagic	Piscivore	2.9	200	4.4	schools
<i>G. cynoglossus</i>	demersal	M Benthivore	9.25	60	3.1	irreg
<i>H. platessoides</i>	demersal	L Benthivore	9.25	82.6	3.7	irreg
<i>I. spatula</i>	demersal	S Benthivore	2.9	21	3.9	none
<i>L. decagonus</i>	demersal	S Benthivore	2.9	21	3.2	rare
<i>L. ferruginea</i>	demersal	M Benthivore	2.9	64	3.2	irreg
<i>L. gibbus</i>	demersal	S Benthivore	2.9	52	3.3	irreg
<i>L. lampretaeformis</i>	demersal	M Benthivore	2.9	50	3.6	rare
<i>L. esmarkii</i>	bathydemersal	M Benthivore	9.25	75	3.4	rare
<i>L. reticulatus</i>	bathydemersal	M Benthivore	2.9	36	3.5	rare
<i>L. vahllei</i>	bathydemersal	M Benthivore	9.25	52	3.4	rare
<i>M. berglax</i>	benthopelagic	L Benthivore	14	110	4.5	rare
<i>M. senta</i>	bathydemersal	M Benthivore	9.25	61	3.5	irreg
<i>M. villosus</i>	pelagic-oceanic	Planktivores	2.9	20	3.2	schools
<i>M. octodecemspinosus</i>	demersal	M Benthivore	1.25	46	3.5	shoal
<i>M. scorpius</i>	demersal	M Benthivore	2.9	60	3.9	shoal
<i>N. bairdii</i>	benthopelagic	S Benthivore	2.9	40	3.6	none
<i>N. chemnitzii</i>	benthopelagic	L Benthivore	9.25	120	3.5	none
<i>P. copei (copei)</i>	bathydemersal	Piscivore	2.9	17	4	irreg
<i>P. chesteri</i>	benthopelagic	PlankPiscivore	9.25	42	3.2	none
<i>R. hippoglossoides</i>	benthopelagic	Piscivore	9.25	80	4.5	irreg
<i>S. mentella</i>	bathypelagic	PlankPiscivore	14	58	3.7	shoal
<i>S. norvegicus</i>	pelagic-oceanic	PlankPiscivore	9.25	100	4	shoal
<i>S. beanii</i>	bathypelagic	M Benthivore	9.25	78	3.9	irreg
<i>S. boa (ferox)</i>	bathypelagic	Piscivore	2.9	30	3.6	none
<i>S. kaupii</i>	bathydemersal	M Benthivore	2.9	100	4.1	none
<i>T. nybelini</i>	demersal	S Benthivore	2.9	20	3.3	none
<i>U. tenuis</i>	demersal	Piscivore	9.25	133	4.2	shoal

Supplemental Table 5. List of proposed research objectives for the Laurentian Channel AOI (Lewis et al. 2014).

ID	Research Objective
1	Advance the understanding of the location, health and integrity of cold water corals and sponges in the Laurentian Channel MPA.

- 2 Identify important as well as **sensitive marine benthic areas and habitats** in the Laurentian Channel MPA by supporting the conduct of scientific surveys, mapping and habitat association studies.
 - 3 Advance the understanding of **plankton variability** in the area and locations of enhanced productivity supporting benthos, fish and cetaceans.
 - 4 Advance the understanding of **cetacean** distribution, abundance and migration in the Laurentian Channel MPA.
 - 5 Advance the understanding of the spatial and temporal distribution of **sharks** and shark by catch, and quantify shark bycatch across all fisheries for species frequenting the Laurentian Channel MPA.
 - 6 Advance scientific studies contributing to the identification and understanding of significant or critical habitat for **SARA listed species** found in the Laurentian Channel MPA.
-

Supplemental R-code:

We provide this section as a template for how the analysis *could* be conducted. This section should not be employed without explicit understanding of the statistical assumptions and output provided from each analysis. *R* is one of many tools available to evaluate the statistical problems posed in this review, and should not be considered the only option.

Code disclaimer: Note that the following code was compiled by R. Stanley who permits its use and manipulation. The onus is on the user of the code to ensure that all assumptions and data validations are properly employed before reporting results. Please cite this document as acknowledgment of its use.

MPA size distribution plots (Figures 1 & 2):

The following code uses the IUCN Protected Planet database (IUCN 2015) to create visual summaries of the distribution of MPAs globally by designation and size.

```
#load dplyr for filter functions and ggplot2 for plotting
library(dplyr)
library(ggplot2)
library(easyGgplot2)
```

#Function for data labeling

```
recoderFunc <- function(data, oldvalue, newvalue) {  
  if (is.factor(data)) data <- as.character(data)  
  if (is.factor(oldvalue)) oldvalue <- as.character(oldvalue)  
  if (is.factor(newvalue)) newvalue <- as.character(newvalue)  
  newvec <- data  
  for (i in unique(oldvalue)) newvec[data == i] <- newvalue[oldvalue == i]  
  newvec  
}
```

#load in MPA metadata

```
mpaDat=read.csv("Data/general_all-csv.csv",sep=";",header=T)
```

#ensure data is in proper format

```
mpaDat$Area=as.numeric(as.character(mpaDat$Area))  
mpaDat$rep_m_area=as.numeric(as.character(mpaDat$rep_m_area))
```

#filter the data to 'marine' protected areas

```
mpaDat=filter(mpaDat, marine == 1)
```

find out which protected areas have no reported surface area and filter them out

```
mpaDat$areaData=rowSums(cbind(mpaDat$rep_m_area,mpaDat$rep_area))  
mpaDat=filter(mpaDat,areaData>0)
```

#Create a list of designation for marine protected areas. This list would exclude those areas with #no official protection and those in estuarine or partially land based. Note that the database #contains no systematic method of naming the reserves. This list is one possible subset of a #large database of protected and/or monitored areas globally.

```
Designations=c("Marine National Park","National Reserve","Marine Protected Area","Marine Park","Marine  
Sanctuary", "Marine Reserve","Marine Protected Area","Marine Park and Sanctuary","Commonwealth Marine  
Reserve","Parque Nacional Marino","Natural Monument","National Monument","National Marine  
Reserve","Reserve","Marine Reserves","Marine Preserve","Marine National Monument","Nature Reserve and  
Marine Reserve", "Marine Protected Area (CCAMLR)","Baltic Sea Protected Area (HELCOM)","National  
Marine Park", "Marine Protected Area (OSPAR)","National Reserve", "Marine National Reserve","Marine  
Multiple Use Reserve","Nationalpark")
```

filter databased to include only those with the chosen designations

```
mpaDat=filter(mpaDat,desig %in% Designations)
```

Two columns denote area ("rep_area" and "rep_m_area") assign the largest one to "Area"

```
mpaDat$Area=0
  for (i in 1:nrow(mpaDat)){
    if(mpaDat$rep_m_area[i]>mpaDat$rep_area[i]){mpaDat$Area[i]=mpaDat$rep_m_area[i]}
    else {mpaDat$Area[i]=mpaDat$rep_area[i]}
  }
mpaDat$Area=mpaDat$rep_m_area
```

#CREATE FIGURE 1

#define size categories

```
vSmall = 5
Small = 10
Medium = 100
Large = 400
```

#create a vector which categorizes MPAs by the size designations

```
mpaDat$SizeCat="Very Large"
mpaDat[which(mpaDat$Area<vSmall),"SizeCat"]="Very Small"
mpaDat[which(mpaDat$Area>vSmall & mpaDat$Area<Small),"SizeCat"]="Small"
mpaDat[which(mpaDat$Area>Small & mpaDat$Area<Medium),"SizeCat"]="Medium"
mpaDat[which(mpaDat$Area>Medium & mpaDat$Area<Large),"SizeCat"]="Large"
```

#filter for only those MPAs with IUCN categorization

```
ind=which(mpaDat$iucn_cat %in% c("Not Applicable","Not Assigned","Not Reported"))
histDat=mpaDat[-ind,]
histDat$iucn_cat=recoderFunc(histDat$iucn_cat,"Ia","I")
histDat$iucn_cat=recoderFunc(histDat$iucn_cat,"Ib","I")
```

#create the designation of “Conservation” or “Multi-use” based on the IUCN Category

```
histDat$Category="Conservation"
histDat$Category[which(histDat$iucn_cat %in% c("IV","V","VI"))]= "Multi-use"
```

#create histogram. Note some labelling needs to be conducted outside of R

```
p1=ggplot2.histogram(data=histDat,xName='Area',groupName='Category',legendPosition="top",alpha=0.7)+
  geom_vline(xintercept=c(5,10,100,400),lty=2)+
  geom_vline(xintercept=c(2.1,60,11908),lty=1,col="grey45",lwd=1.25)+
  scale_y_log10(breaks = trans_breaks("log10", function(x) 10^x),
    labels = trans_format("log10", math_format(10^x)))+
  scale_x_log10(breaks = trans_breaks("log10", function(x) 10^x),
    labels = trans_format("log10", math_format(10^x)))+
  coord_cartesian(ylim=c(10^0,10^3))+theme_bw()+annotation_logticks()+
```

```

labs(x=expression(paste("MPA Area ( ",km^2," )",sep="")),y="Frequency")+ theme(legend.position="top")

#Save plot
ggsave("Figures/SizeHistogram_IUCN_Protection.jpg",p1)

#CREATE FIGURE 2
#Remove any reported MPAs with no area
mpaDat=mpaDat[-which(mpaDat$Area==0),]

#Vectors of MPA sizes (km2) and names.
Sizes=c(2.1,60,11908)
Names=c("Eastport","Gilbert Bay","Laurentian Channel")

#Fit a empirical cumulative frequency distribution to the size data
fit=ecdf(log10(mpaDat$Area))

#Calculate percentiles for the sizes of the MPAs
Larger=fit(v=log10(Sizes))

#Create dataframe which contains the vector of y-axis locations (t) and labels (l)
Ticks=data.frame(t=round(sort(c(Larger,0,0.25,0.5,0.75,1)),3),
l=round(sort(c(Larger,0,0.25,0.5,0.75,1)),2));Ticks

#Create plot
p2=ggplot(mpaDat,aes(x=Area))+stat_ecdf()+
scale_x_log10(breaks = trans_breaks("log10", function(x) 10^x),labels = trans_format("log10",
math_format(10^.x)))+
annotation_logticks(side="bottom")+
labs(x=expression(paste("MPA Area ( ",km^2," )",sep="")),y="Cumulative frequency")+
geom_vline(xintercept=Sizes,lty=1,col="grey45",lwd=1.25)+
geom_segment(aes(x=0,y=Larger[1],xend=Sizes[1],yend=Larger[1]),lty=2)+
geom_segment(aes(x=0,y=Larger[2],xend=Sizes[2],yend=Larger[2]),lty=2)+
geom_segment(aes(x=0,y=Larger[3],xend=Sizes[3],yend=Larger[3]),lty=2)+
scale_y_continuous(breaks=Ticks$t,labels=Ticks$l)+
annotate("text",x=Sizes[1]+1.5,y=0.25,label=Names[1],angle=270)+
annotate("text",x=Sizes[2]+35,y=0.25,label=Names[2],angle=270)+
annotate("text",x=Sizes[3]+6500,y=0.25,label=Names[3],angle=270)+
theme_bw()+theme(panel.grid.major = element_blank(), panel.grid.minor = element_blank());p

```

#Save plot. Note that the file reference is relative to the current working directory which can be #found with the command `getwd()`
`ggsave(filename = "CumulativeFreq.png",p2)`

Asymmetrical BACI design:

For single species analyses, the linear mixed-effects model (`lmer`) function in the package ‘*lme4*’ can incorporate both random and fixed factors when evaluating potential interactive effects. For example, an analysis of abundance of a species in side vs outside the MPA over time would be coded as:

Code1: `M1 = lmer(Abundance ~ Designation + Loc + Designation*Loc+ (1|Site), data=Dataset)`

where *Dataset* denotes the data with column names corresponding to the variables in the model. Similarly, the PERMANOVA function (`adonis`) in the ‘*vegan*’ package handles fixed and random effects when evaluating interactions using dissimilarity matrices. For example an analysis of a fish assemblage response to protection would be coded as:

Code 2: `Response = Dataset[,c("Sp1", "Sp2" ... , "Spn")]`

Code 3: `M2 = adonis(Response ~ Designation + Loc + Designation*Loc+ Loc:Site, data=Dataset)`

where *Response* represents the abundance of each species defined by columns headers (e.g. SP1) in the *Dataset* with column names corresponding to the variables in the model. In both examples a significant interaction (*Designation*Loc*) would indicate a reserve effect. Pairwise post-hoc tests of the interaction term test for pre and post-designation significant differences within versus outside the MPA.

Modified asymmetrical BACI design with spatial variant (spillover):

Code 4: `M3= lmer(Abundance ~ Designation*Loc*Dist + (1|Site), data=Dataset)`

where pairwise tests on significant three-way (*Designation*Loc*Dist*) and two-way (*Designation*Loc*) interactions would elucidate the effect of protection and proximity to the MPA.

Modified asymmetrical BACI design with spatial variant and offset:

Code 5: `M4 = lmer(Abundance ~ Designation*Loc*Dist + (1|Site)+offset(gear), data=Dataset)`

Where offset is a numeric or ordinal variable which accounts for the variation in the relative gear or capture efficiency.

General linear model using control-impact design:

Code 6: `M5 = glm(Abundance ~ location + (1|Site)+offset(gear), data = Dataset)`

General linear model using control-impact design with temporal variant:

Code 7: `M6 = glm(Abundance ~ location*year + (1|Site)+offset(gear), data = Dataset)`

FishBase database trait query for Newfoundland MPA focal species:

Code 8: `data(fishbase)`

Code 9: `Fish=c("Gadus morhua", "Malacoraja senta", "Centroscyllium fabricii", "Lamna nasus", "Anarhichas denticulatus")`

Code 10: `mpafish = findSpecies(Fish, fish.data)`

Code 11: `getDepth(fish.data[mpafish])`

where Code 8 brings the FishBase database into the R workspace, Code 9 creates a list the genus' and species' of interest, Code 10 identifies the position of these species in the FishBase database, and Code 11 returns a summary of depth information for the species defined in Code 9. There are a variety of tools like 'getDepth' (Code 11) to extract trait data of interest from the FishBase database. The FishBase database can also be manually queried for available data using the species names:

Code 12: `fish.data[findSpecies("Gadus morhua", fish.data)]`

where Code 12 returns all the available data on FishBase for Atlantic cod (*Gadus morhua*).

Functional diversity calculations:

The calculation of functional diversity variables are done contemporaneously in the FD package. The FD package also permits missing values, allowing inclusion of species with incomplete trait databases in functional diversity metrics whatever trait data are available:

Code 13: `FD_PA = dbFD(x = Trait_db, a = SurveyData, w.abund=F)`

Code 14: `FD_Abun = dbFD(x = Trait_db, a = SurveyData, w.abund=T)`

where *Trait_db* is a trait database with each row representing a species and each column a numeric or categorical representation of the trait, *SurveyData* are the results from surveys where each row is a trawl set and each column is a species (absence of a species is coded as 0). *FD_PA* is based on the presence of a trait and *FD_Abun* is weighted by the observed abundance of a trait in a trawl set. Code 13 and 14 return all associated functional diversity metrics outlined in table 5 and by (Gower & Legendre 1986, Mason et al. 2005, Laliberte & Legendre 2010).

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