INTEGRATING BENTHIC HABITAT MAPPING AND SEASCAPE ECOLOGY INTO MARINE CONSERVATION PRIORITIZATION

by

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Abstract

Advances in seafloor mapping have allowed for the production of fine-scale seafloor landscape (i.e., benthoscape) maps that are analogous to terrestrial land cover maps, providing the foundation for assessing the spatial configuration of seafloor habitat patches. While many species rely on large, well-connected patches for foraging and migration, variability in patch size and configuration can be difficult to incorporate into Marine Protected Area (MPA) design. In this thesis, I developed a novel method that considers the spatial arrangement of benthic habitat patches in MPA design. I applied the approach to the Eastport MPA and surrounding region in Newfoundland, Canada by first quantifying the composition and configuration of the benthoscape using multibeam echosounder, seafloor video surveys, and patch size and connectivity metrics. Using a reserve design algorithm, I then compared outputs that included and excluded the prioritization of benthoscape connectivity. The approach presented in this thesis results in the preferential selection of large patches within the home-range of a given species, which can be important for reducing fragmentation in conservation prioritization solutions and better supporting species and ecological processes. This approach offers potential benefits for the conservation of coastal and marine regions by increasing our understanding of how we can incorporate broad scale patterns into on-the-ground conservation decision making.

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List of Symbols, Nomenclature and Abbreviations

ANOSIM	Analysis of Similarity
BLM	Boundary Length Modifier
CBD	Convention on Biological Diversity
CCGS	Canadian Coast Guard Ship
CHS	Canadian Hydrographic Service
CPAWS	Canadian Parks and Wilderness Society
dB	Decibel
deg	Degree
DFO	Department of Fisheries and Oceans
EBSA	Ecologically and Biologically Significant Area
GIS	Geographic Information System
GMS	Gravelly Muddy Sand
Iso	Iterative self-organizing
kHz	Kilohertz
MBES	Multibeam Echosounder
MPA	Marine Protected Area
NL	Newfoundland and Labrador
nMDS	non-metric Multidimensional Scaling
SCP	Systematic Conservation Planning
NSERC	Natural Sciences and Engineering Research Council
SIMPER	Similarity Percentage

PRIMER	Plymouth Routines in Multivariate Ecological Research
PU	Planning Unit
QPS	Quality Positioning Services

Co-Authorship Statement

The student's contributions to the thesis manuscripts are as follows:

- Led the finalization of research questions
- Conducted analyses of video and imagery data from 2016 surveys in Newman Sound
- Conducted all data analyses used for all chapters
- Drafted all the thesis chapters and lead author on the two papers (Chapters 2 & 3)

Co-supervisor Dr. Rodolphe Devillers (Geography, Memorial University of Newfoundland) co-authored Chapters 2 and 3, contributed to survey design, interpretation of results and helped improve manuscript drafts. Co-supervisor Dr. Craig Brown (Applied Research, Nova Scotia Community College) co-authored Chapters 2 and 3, contributed to interpretation of results and helped improve manuscript drafts. Committee member Dr. Evan Edinger (Geography and Biology, Memorial University of Newfoundland) co-authored Chapter 2, contributed to survey design, interpretation of results and revisions to manuscript draft. Alison Copeland (Department of Environment and Natural Resources, Government of Bermuda and former Master's student at Memorial University of Newfoundland Geography) co-authored Chapter 2, led acquisition, analysis and interpretation of video and imagery from 2005 surveys in Newman Sound and provided minor revisions to Chapter 2. Committee member Dr. Yolanda Wiersma (Biology, Memorial University of Newfoundland), contributed to project conception for the thesis.

This thesis is based on the following manuscripts:

Chapter 2:

Proudfoot, B., Devillers, R., Brown, C.J., Edinger, E. & Copeland, A. (*submitted to the Journal of Coastal Conservation, June 2018*) Seafloor mapping to support conservation planning in an ecologically unique fjord in Newfoundland and Labrador, Canada.

Chapter 3:

Proudfoot, B., Devillers, R. & Brown, C.J. (*in prep*) Integrating fine-seafloor mapping and seascape ecology metrics into marine conservation prioritization.

Chapter 1 – Introduction

1.1 Context

1.1.1 Marine Protected Areas

Conserving Earth's biodiversity in the shadow of intensifying impacts from anthropogenic activities has never been more pressing. Impacts related to pollution (Islam and Tanaka 2004), overexploitation of fisheries resources (Pauly et al. 2005), resource extraction (Niner et al. 2018), invasive species (Bax et al. 2003) and coastal development have resulted in rapid habitat and biodiversity loss (Halpern et al. 2008). Recent reports indicate that global wildlife populations have declined by 60 percent on average in less than 50 years (WWF 2018). Marine Protected Areas (MPAs) are spatial management tools that can mitigate the effects of some of these impacts, help protect and restore biodiversity and habitats while supporting conservation and sustainable use of marine resources (Green et al. 2014; Lester et al. 2009).

Canada has committed to protecting 10% of its coastal and marine waters by 2020 through an ecologically representative and well-connected system of protected areas (Aichi Target 11; CBD-UNEP, 2010). Despite progress in recent years (DFO 2017), Canada has yet to reach this target. A very low proportion of existing Canadian MPAs have a strong level of protection (0.01% are no-take MPAs; Marine Conservation Institute 2018) despite evidence that no-take MPAs provide higher conservation benefits compared to MPAs providing partial protection (Lester and Halpern 2008). As new protected areas are rapidly being proposed and designated, understanding how existing protected areas contribute to these conservation goals is important, particularly when protecting the full range of marine and coastal environments is a primary goal. Furthermore, designing ecologically sound MPAs that support biodiversity and resilience in marine ecosystems is a significant challenge, not only to identify areas that are important to protect, but also in terms of the social and economic considerations that are involved (Pascual et al. 2016; Agardy et al. 2016).

1.1.2 Systematic conservation planning and conservation prioritization

Methods for designing and prioritizing individual sites for inclusion in MPAs and MPA networks can be guided by systematic conservation planning approaches (SCP; Margules and Pressey 2000). SCP, a field that emerged from terrestrial conservation planning, provides a framework for making repeatable, transparent and efficient conservation decisions (Margules and Pressey 2000) and is often used in marine conservation planning. One of the many key elements of SCP that is particularly important to consider in MPA design is representativity, which refers to the goal of reserves sampling across the full range of variation of biodiversity and habitats in a region (Margules and Pressey 2000). Other fundamental goals of SCP include comprehensiveness, adequacy and efficiency. Comprehensive reserve and reserve networks aim to protect the full range of biodiversity features, where biodiversity features can include species, habitats and ecological processes (Kukkala and Moilanen 2013). The key distinction between comprehensiveness and representativity is that comprehensiveness focuses on defining the full range of biodiversity feature that are to be protected while representativity considers the degree to which the biodiversity features are protected within conservation solution (protected area network) (Kukkala and Moilanen 2013). Representativity is often discussed in the context of quantitative targets or proportions of biodiversity features included in the conservation solution (Kukkala and Moilanen 2013). Adequacy refers to the goal of protecting enough to ensure the persistence of biodiversity features (Kukkala and Moilanen 2013). Finally, efficiency refers to the goal of meeting conservation objectives at a minimal cost (Kukkala and Moilanen 2013). Cost can refer to monetary costs associated with acquiring, operating and monitoring the protected area, the monetary value corresponding to the displacement of human activities (e.g. resource extraction) that would potentially accompany implementation, as well as social and cultural values and local importance of an area (Ban and Klein 2009).

Evidently, designing reserves that balance these principles can pose challenges for conservation planners when the goal is to maximize conservation benefits while minimizing impacts to resource users. Fortunately, decision support tools such as Marxan (Ball and Possingham 2000), Zonation (Moilanen et al. 2009) and C-Plan (Pressey et al. 2009) can aid in generating MPA and MPA network design scenarios while meeting SCP goals. Marxan is widely utilized by many in the field of conservation planning. The tool uses spatial data layers to represent conservation and cost features, and a simulated annealing algorithm to identify near-optimal scenarios for meeting user defined conservation targets for the least cost (Ball and Possingham 2000). Marxan is the most widely used SCP tool in the world, and is currently being used by Fisheries and Oceans Canada (DFO) to aid in the creation of bioregional MPA networks in Canada's three oceans (DFO 2018). While tools such as Marxan, Zonation and C-Plan can help at various stages of the MPA design processes, they require rich spatial data representing conservation and cost features.

1.1.3 From landscape ecology to seascape ecology

While SCP elements such as representativity, comprehensiveness, adequacy and efficiency are central to conservation decision-making, the spatial arrangement of habitat patches and its influence on ecological processes has also become a principal element of conservation planning in in recent years (Pittman et al. 2017). However, SCP tools such as Marxan do not consider the spatial arrangement of conservation features and habitat patches, and tend to focus primarily on the core SCP elements (representativity, adequacy, comprehensiveness and efficiency). Assessing the spatial configuration of habitat patches typically requires spatial data at scales relevant to management decisions (Stevens and Connolly 2004; Ardron et al. 2010). The ability to produce fine-scale spatial data representing habitats in regions that are important for conservation provides important baseline information for designing and monitoring MPAs. This spatial information also permits the assessment of spatial arrangement, composition and configuration of seafloor landscapes, which are important for structuring habitats and understanding movement and migration patterns for many organisms. This relationship between spatial pattern (arrangement and composition of patches within a landscape) and ecological processes that shape the distribution and abundance of species is the essence of landscape ecology (Turner et al. 2015). In recent years, seascape ecology (Pittman 2017) – the application of landscape ecology concepts to marine environments – has become increasingly important for supporting ecologically sound conservation practices, much like its terrestrial counterpart (Boström et al. 2011). Landscape ecology methods and metrics include those related to connectivity and fragmentation as well spatial pattern metrics such as patch size, shape and proximity to neighbouring patches. Connectivity, in particular, has recently become a central focus of marine conservation planning (Olds et al. 2016).

In a broad sense, ecological spatial connectivity refers to the transfer and movement of organisms, species, genes and nutrients among spatially separated populations, communities or ecosystems (Carr et al. 2017). Connectivity can be quantified by combining oceanographic modelling and measures of pelagic larval duration (PLD) to identify the spatial location of larvae sources and sinks (Carr et al. 2017; Coleman et al. 2017). Connectivity can also be identified by the post settlement migration of individuals in shallow, biogenic habitats such as coral reefs, mangrove forests and eelgrass beds (Grüss et al. 2011; Weeks et al. 2017). From a conservation planning perspective, supporting these complex connectivity processes involves not only protecting large regions that encompass critical habitat, but also habitat corridors, larval sources and sinks, nursery and spawning grounds and regions that serve as donors and recipients of nutrient transfers (Burt et al. 2014).

Seascape connectivity is one form of ecological spatial connectivity that is analogous to landscape connectivity in the sea. Seascape connectivity can refer to structural connectivity (i.e., physical linkages within a seascape), potential connectivity (i.e., a measure of connectivity that incorporates limited or assumed information on species mobility) or actual connectivity (i.e., a measure that uses spatial information on species movement to quantify connectivity; Grober-Dunsmore et al. 2009).

Including connectivity processes in protected area design can positively influence reserve outcomes by improving fish abundance within protected area boundaries (Olds 2012a, 2012b), as well as increasing productivity and biodiversity (Olds et al. 2016). As such, methods and applications for integrating connectivity information with widely used conservation prioritization tools (i.e., Marxan) are being developed and provide operational frameworks for integrating various forms of connectivity into conservation planning (e.g. Beger et al. 2010; Weeks et al. 2017; Daigle et al. 2018).

Connectivity studies are increasingly common in marine systems (Magris et al. 2014; Olds et al. 2016), in part fueled by advances in remote sensing and oceanographic modelling – technologies that offer valuable insights into processes that operate below the surface of the ocean (Olds et al. 2016). The increasing use of genetic approaches in marine ecology also advance our understanding of connectivity (Riginos and Liggins 2013). The majority of seascape connectivity studies have been conducted in shallow environments, and focus on connectivity between patches of coral reefs, mangroves or seagrass beds (Olds et al. 2016; Wedding et al. 2011; Pittman et al. 2011) – areas that provide shelter and habitat for many juvenile species. However, connectivity research is not limited to shallow biogenic habitats and ecosystems. Discrete oceanographic "patches," defined by variables such as temperature, salinity and dissolved oxygen are also being considered in marine connectivity studies (Guidetti et al. 2013).

In contrast, seascape connectivity research in benthic ecosystems that rely on acoustic mapping remains largely unstudied even though processes such migration, dispersal, reproduction and range expansion are also occurring in these ecosystems (Comeau et al. 1998; Comeau and Savoie 2002; Hovel and Wahle 2010).

1.1.4 Fine-scale seafloor mapping to support seascape ecology and conservation planning

High quality seascape maps are increasingly valuable in light of recent efforts towards supporting landscape-scale ecological processes such as migration, foraging and connectivity in marine conservation (Burt et al. 2014; Magris et al. 2014; D'Aloia et al. 2017). As a result, seascape maps have become central to many marine conservation and management activities (Brown et al. 2012; Copeland et al. 2013; Buhl-Mortensen et al. 2015; Novaczek et al. 2017a). Imagery acquired from satellites and unmanned aerial vehicles (UAVs) as well as Light Detection and Ranging (LiDAR) data can be used to map intertidal and shallow subtidal coastal zones (Reshitnyk et al. 2014; Jalali et al.

2015). However, these light-based techniques are limited in their ability to survey subtidal habitats and ecosystems at greater depths than the shallow subtidal zone.

Advances in acoustic (sonar) seafloor mapping and sampling techniques have allowed for the production of continuous seafloor maps that are essentially analogous to digital terrain models in terrestrial environments (Brown et al. 2011). Single-beam and sidescan sonars, seismic profilers and multibeam echosounders (MBES) are now integral to habitat and substrate mapping exercises that aim to map environments that are beyond the depths at which light can penetrate. Seascape maps can inform marine conservation (Novaczek et al. 2017b; Young and Carr 2015) particularly in terms of identifying vulnerable or threatened habitats (Rengstorf et al. 2013), restoration activities (Walker and Alford. 2016) and fisheries management initiatives (Brown et al. 2012; Smith et al. 2017; Walton et al. 2017). The spatial structure of the seascape can be represented as a two or three dimensional continuous surface, or more often as a two-dimensional classification of benthic habitat (Pittman 2017). Mapping the benthoscape (i.e., the component of the seascape that relates to the benthic environment; Zajac et al. 2000) can support the identification of the extent, location and composition of seafloor features and biodiversity, which provides important baseline information for designing and monitoring MPAs (Lacharité and Brown *in press*).

Benthoscape maps and species specific habitat maps are typically developed by interpreting continuous coverage environmental seafloor data, such as bathymetry and reflectivity (backscatter), often collected using MBES (Brown et al. 2011). Bathymetric derivatives such as slope, rugosity, aspect and curvature can also be extracted from MBES data, and have been shown to be important variables for predicting substrate and species distributions (Monk et al. 2010; Brown et al. 2011; Lecours et al. 2017). The strength of the backscatter return signal is also related with seafloor substrate composition (Kostylev et al. 2003; McGonigle and Collier 2014). Backscatter intensity (ranging from strong to weak) can capture variability in substrate characteristics with strong backscatter signals typically associated with hard, consolidated substrates such as bedrock, boulder and cobble while weak signals are typically associated with soft, muddy and sandy substrates (Lurton and Lamarche 2015). Recording and interpreting backscatter intensity can be complex. A number of factors, including system-specific settings and environmental variability can result in backscatter intensity values that are not typically calibrated across surveys (see Lurton and Lamarche 2015 and Lacharité et al. 2017 for comprehensive discussions of backscatter measurements and challenges). This poses challenges in terms of extrapolating the substrate and habitat predictions across multiple MBES coverages, and the use of such data for long term monitoring and change detection. Additionally, high costs associated with collecting MBES data pose further challenges. However, novel methods for creating seabed maps from multiple MBES coverages are being developed and refined (Lacharité et al. 2017) and contribute substantially to our ability to produce fine-scale seafloor maps for a variety of ocean management and conservation needs.

Benthoscape maps produced using these techniques yield the spatial information required for exploring and testing landscape ecology and connectivity concepts in benthic environments that require acoustic techniques to map. Methods and metrics include those related to connectivity and fragmentation and spatial pattern metrics such as patch size, shape and proximity to neighbouring patches.

1.1.5 Integrating benthoscape mapping, connectivity analyses and conservation prioritization

Despite evidence that including connectivity positively influences conservation outcomes (Olds et al. 2016), cases in marine contexts where connectivity is integrated with widely used conservation prioritization tools (e.g., Marxan) are rare and dominated by studies in shallow coastal systems that can be mapped using optical remote sensing techniques (e.g. Crouzeilles et al. 2015; Magris et al. 2016; Weeks et al. 2017). Recently proposed applications and methods for including connectivity in conservation prioritization continue to advance the field (e.g. Weeks et al. 2017; Daigle et al. 2018). MarxanConnect is one such application that allows users to incorporate estimates of directional demographic and landscape connectivity in Marxan conservation prioritization (Daigle et al. 2018). Another recently proposed method integrates connectivity analyses and Marxan conservation prioritization in a relatively data-limited context (Weeks et al. 2017). In terms of benthic species and ecosystems, benthoscape maps and species-specific habitat maps are particularly valuable for achieving positive conservation outcomes (Ferrari et al. 2018). However, methods are lacking for incorporating benthoscape mapping, configuration and connectivity into widely used conservation prioritization tools. Of the limited number of studies that aim to link connectivity and conservation prioritization, the majority tend to focus on larval connectivity (White et al. 2014; Magris et al. 2016; D'Aloia et al. 2017) and post settlement migration of individuals in shallow, biogenic habitats such as coral reefs, mangrove forests and eelgrass beds (Grüss et al. 2011; Weeks et al. 2017) while little attention has been paid to benthic environments that require acoustic techniques to map. While Marxan focuses on questions of comprehensiveness, representativity, adequacy and efficiency, exploring how the current Marxan workflow could be modified to enable the procedure to distinguish between large, well connected and isolated fragments of benthic habitat is valuable.

This project addresses this research gap through the development of a method that integrates benthic habitat mapping and landscape connectivity and fragmentation analysis into the standard Marxan workflow so that the spatial arrangement of habitat patches can be considered in conservation prioritization. In terms of spatial conservation prioritization initiatives, filling this research gap is important for reducing habitat fragmentation and better supporting species and ecological processes.The method was tested through a case study in Newman Sound, an ecologically diverse and well-studied coastal region in of Newfoundland and Labrador.

1.2 Research Questions

To address this research gap, this thesis aims to answer the following research questions:

1. What is the composition and spatial configuration of the Newman Sound benthoscape?
2. How can landscape ecology metrics be applied to benthoscape maps to describe and measure seafloor structural and species specific potential connectivity?
3. How can landscape ecology metrics be used by conservation prioritization tools so that MPA design considers benthoscape composition and configuration?

1.3 Research Objectives

The overarching goal of this thesis is to develop a new method for incorporating seascape connectivity metrics related to patch size and connectivity into a conservation prioritization tool, while simultaneously meeting benthic habitat and/or substrate representativity targets. This goal and the above research questions will be answered by:

1. Using the benthoscape mapping approach (Brown et al. 2012) to map the seafloor in Newman Sound, NL, using newly acquired and archival seafloor video survey and multibeam echosounder data.
2. Applying patch size, proximity and species specific potential connectivity metrics to benthoscape classes to assess benthoscape composition and configuration at a broad seascape scale.

3. Developing a method for integrating fine-scale benthic mapping, species-specific potential connectivity and spatial pattern metrics into widely used conservation prioritization tools.
4. Comparing conservation prioritization scenarios when the spatial configuration of the benthoscape and its potential effect on ecological processes is considered.

1.4 Method Summary

A general overview of the methods applied in this research project is described below.

1.4.1 Study area

This research was conducted in Newman Sound, a coastal fjord located in Bonavista Bay in Newfoundland and Labrador (NL). Due to the high diversity and abundance of ecologically unique areas, including tidal flats, eelgrass and rhodolith beds, and species-rich submerged fjord walls, Newman Sound has been described as a special marine area (CPAWS-NL 2018). Newman Sound is a well-studied region with high conservation potential, and thus offers a suitable scenario for applying the benthoscape mapping approach and for developing a method for integrating fine-scale benthoscape mapping, configuration and connectivity into conservation prioritization. Furthermore, one of the two MPAs in NL, the Eastport MPA, is located in the region and provides context for discussing potential adaptive management scenarios that could help shift from single species to biodiversity conservation. The Eastport MPA consists of two closures: Duck Island and Round Island. Much work has been done in terms of identifying the fine scale benthic habitats within the Eastport MPA (Novaczek et al. 2017a) and preliminary mapping in Newman Sound (Copeland 2006). This, coupled with additional ground truthing data acquired during this project, provides an ideal situation for integrating benthoscape configuration and connectivity metrics into conservation prioritization using fine scale data in a coastal context.

1.4.2 Benthoscape mapping

The seafloor adjacent to the Round Island closure was mapped using the benthoscape approach (Brown et al. 2012). The approach involves using multibeam echosounder (MBES) data, seafloor imagery and a pixel-based classification method. The abundance of benthic fauna was also recorded for each seafloor image. Relationships between seafloor substrates and species presence/absence was explored using multivariate statistical analyses with the aim of capturing the potential impact of seafloor structure (depth, slope and predicted substrate) on benthic community composition.

1.4.3 Connectivity and Marxan conservation prioritization

The next stage of this research involved using the benthoscape map produced to quantify the composition and configuration of seafloor habitats (benthoscape units) adjacent the MPA using landscape ecology concepts and metrics. This included measuring patch size, shape and structural connectivity metrics. The metrics were then captured in a data layer used by the Marxan SCP tool, so that large, well-connected patches were favored during site selection.

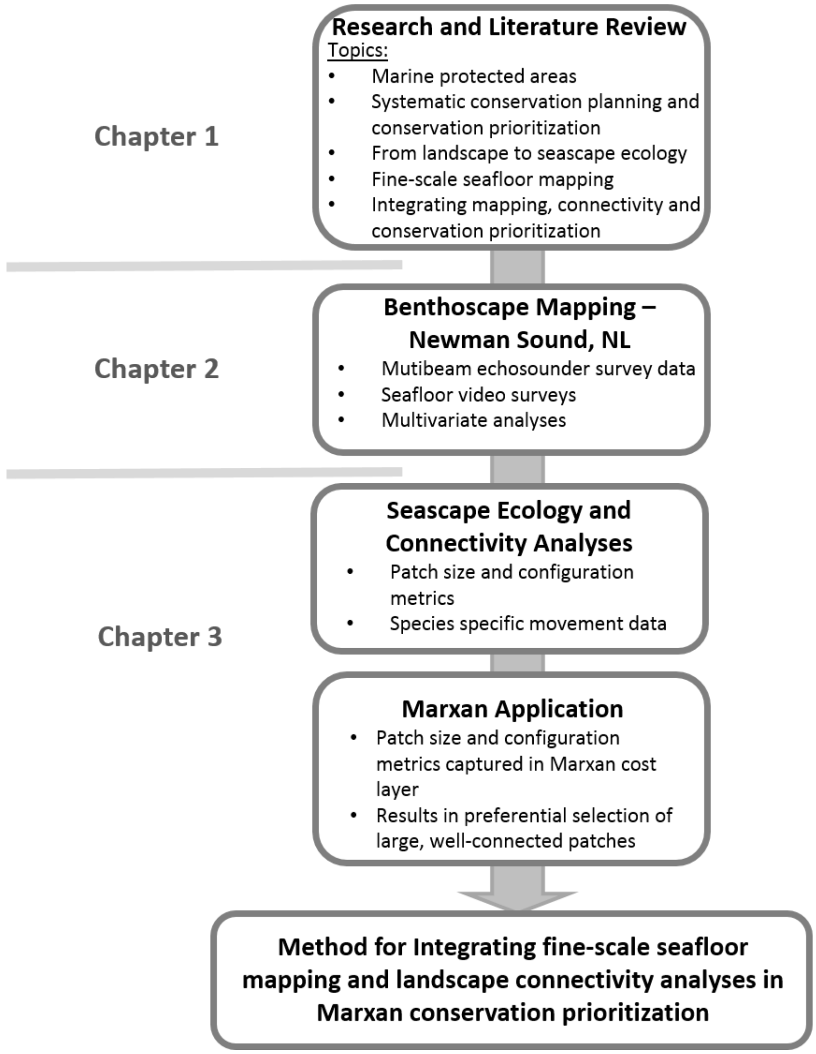

Figure 1-1: General methods and techniques applied in this thesis

1.5 Thesis Outline

This thesis is presented in manuscript format and includes two research chapters as submitted to the journals for publication. Chapter 2 describes the method and results related to the benthoscape mapping activities in Newman Sound. The resulting benthoscape map provides the foundation for assessing spatial arrangement and landscape ecology metrics described in Chapter 3. Chapter 3 demonstrates the newly developed method for integrating benthoscape mapping, connectivity and conservation prioritization. Chapter 4 provides a general summary and discussion of the limitations and future applications of this research.

While the benthoscape mapping and subsequent connectivity and conservation prioritization method was developed and tested in the context of Newman Sound, the method developed in this research can be applied to other regions and protected areas in Canada where fine-scale habitat or substrate maps are available.

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Chapter 2 – Seafloor mapping to support conservation planning in an ecologically unique fjord in Newfoundland and Labrador, Canada

2.1 Abstract

Purpose: As human impacts continue to threaten coastal habitats and ecosystems, marine benthic habitat and substrate mapping has become a key component of many conservation and management initiatives. Understanding the composition and extent of marine habitats can inform marine protected area (MPA) planning and monitoring, help identify vulnerable or rare habitats and support fisheries management. To support conservation planning in Eastern Canada, we mapped the seafloor of Newman Sound, identifying the benthoscape classes (i.e., discrete biophysical seafloor classes) of this ecologically diverse and unique fjord in Newfoundland and Labrador (NL).

Methods: Mapping was achieved using multibeam echosounder (MBES) data collected using multiple platforms, seafloor videos and an unsupervised pixel-based classification method.

Results: Seven benthoscape classes were identified within the extent of the MBES coverages. Multivariate statistical analyses indicate that two benthoscape classes - mixed boulder and mud - support distinct epifaunal communities, and also capture the changes in benthic community composition between hard/shallow substrates and soft/deep substrates.

Conclusions: Our results illustrate how benthoscape maps can inform marine spatial planning and conservation in the Newman Sound region, support monitoring and also calls for adaptive management of the adjacent Eastport MPA.

2.2 Introduction

Coastal areas are among the most highly threatened and vulnerable regions of the world (Waycott et al. 2009; Halpern et al. 2015; Li et al. 2018). Climate change, natural resource extraction, coastal development and land-based pollution are just a few of many stressors that can lead to habitat degradation and loss in coastal areas (Halpern et al. 2008; Grech et al. 2011; Lefcheck 2017).

Protecting coastal areas requires knowledge of the habitats and ecosystem characteristics of the region. However, it can be difficult to assess impacts and monitor changes without baseline information on the composition and extent of coastal habitats and ecosystems. As a result, habitat and substrate maps have become central to many coastal conservation and management activities (Brown et al. 2012; Copeland et al. 2013; Buhl-Mortensen et al. 2015; Novaczek et al. 2017a).

Advances in acoustic (sonar) seafloor mapping and sampling techniques have allowed for the production of fine-scale generalized biophysical maps of the seafloor (i.e., benthoscape maps – Brown et al. 2012). Specifically, multibeam echosounders (MBES) offer a survey technique capable of collecting continuous coverage baseline information pertaining to seafloor characteristics (Brown et al. 2011). Seafloor bathymetry measured by MBES can be used to understand the geomorphology of the seafloor through the production of bathymetric derivatives such as slope, rugosity, aspect and curvature (Brown et al. 2011). MBES backscatter, a measure of the acoustic signal strength returning from the seafloor, has also proven valuable in understanding seafloor characteristics within a region. Backscatter strength can capture variability in substrate composition, with strong backscatter signals typically representing hard, consolidated substrates such as bedrock, boulder and cobble, while weak signals correspond to soft, muddy substrates (Lurton and Lamarche 2015). Using a combination of bathymetry, bathymetric derivatives and backscatter have proven valuable for predicting substrate and species distribution (Brown et al. 2011; Monk et al. 2010; Lecours et al. 2017). Recording backscatter intensity is complex. A combination of factors, including system-specific settings and environmental variability can result in backscatter intensity values that are not typically calibrated across surveys (see Lurton and Lamarche 2015 and Lacharité et al. 2017 for comprehensive discussions of backscatter measurements and challenges). This can present further challenges in terms of extrapolating substrate and habitat predictions across multiple MBES coverages. However, novel methods have been developed and refined for producing seamless biophysical seafloor maps using multiple MBES coverages from an area (Lacharité et al. 2017).

Benthoscape maps and species-specific habitat maps produced using these techniques can inform marine conservation and marine spatial planning activities (Novaczek et al. 2017b; Young and Carr 2015). Specifically, these maps can help identify vulnerable or threatened habitats (Rengstorf et al. 2013), support and monitor restoration activities (Walker and Alford 2016) and inform fisheries management (Brown et al. 2012; Smith et al. 2017; Walton et al. 2017). Identifying the extent, location and composition of seafloor features and biodiversity in the form of habitat maps also provides important baseline information for assessing change, particularly as impacts from climate change and anthropogenic activities continue to threaten the resilience of coastal areas.

Here, we used MBES bathymetry, backscatter and derivatives from two uncalibrated multibeam systems to create a benthoscape map of a coastal fjord with high conservation potential. We then discuss how the map provides important baseline information to support conservation planning in the region.

Newman Sound is a fjord located in Bonavista Bay on the northeast coast of the island of Newfoundland, Canada. The western part of the fjord is located within the boundaries of Terra Nova National Park, one of two national (i.e., federal) parks on the Island of Newfoundland. Due to the high diversity and abundance of ecologically unique areas, including tidal flats, eelgrass and rhodolith beds, and species rich submerged fjord walls, Newman Sound has been listed in an expert process as a ‘special marine area’ (CPAWS-NL 2017). ‘Special marine areas’ recognize special or representative marine features identified during a series of workshops where scientists, provincial and federal government and community experts together identified areas in the province of Newfoundland and Labrador of higher conservation value. The inner basin of Newman Sound, makes up approximately 70% of the Terra Nova Migratory Bird Sanctuary (11.8 km^2), and provides important habitat for a high diversity of migratory shorebirds, seabirds and resident waterfowl (Environment and Climate Change Canada 2017). Newman Sound also supports sizeable eelgrass (*Zostera marina*) beds, which serve as important refuge and nursery areas for juvenile Atlantic cod (*Gadus morhua*) (Linehan et al. 2001; Cote et al. 2004; Rao et al. 2014) and other fish and invertebrate species (Joseph et al. 2006; Cote et al. 2013).

The Round Island closure, one of two closed areas of the Eastport Marine Protected Area (MPA), is located in Newman Sound (Fig. 2-1), adjacent to the area mapped in this study. Eastport is one of two Fisheries and Oceans Canada (DFO) Oceans Act MPAs in the province of Newfoundland and Labrador. This small (2.1 km^2) no-take (i.e. closed to all fishing) MPA was established in 2005, based on a voluntary fishing closure initiated by the local community in 1997. The primary goal of the Eastport MPA is to protect American lobster (*Homarus americanus*), a species fished commercially in the region. Reports of high lobster catches informed MPA boundary placement. However, no habitat mapping or biodiversity surveys were done prior to MPA establishment. Recent habitat mapping and characterization within the boundaries of the MPA have determined that the MPA does little to conserve habitats and biodiversity representative of the broader region of Newman Sound (Novaczek et al. 2017a). While the Eastport MPA was primarily designed to help protect a single species, its small size offers limited protection of ecologically diverse and unique areas in Newman Sound. A recent ecosystem goods and services study used biological indicators (estimates of lobster abundance; catch per unit effort; estimates of female lobsters that are carrying eggs; size distribution) to examine the impact of the MPA on the local lobster fishery (Lewis et al. 2017). The study determined that the Eastport MPA has little to no effect on the enhancement of the local fishery as the majority of biological indicators show no significant improvement when compared to regional scale patterns. The small size of the Eastport MPA is frequently cited as limiting the fisheries enhancement and biodiversity conservation benefits of the MPA (Lewis et al. 2017; Novaczek et al. 2017a; Stanley et al. 2018), leading to calls for adaptive management that could result in expanding or changing the MPA boundaries to include more diverse habitats (Novaczek et al. 2017a). Adaptive management allows regulations and boundaries to be improved as new data and knowledge accumulates and as ecological systems change through time (Wilhere 2002). While identifying the appropriate MPA size is complex, general guidelines recommend that if an MPA is intended to conserve biodiversity and support climate change resilience it should be moderate to large in size, ideally 4-20 km in dimension (Green et al. 2014).

In addition to creating a benthoscape map of Newman Sound, this research also discusses how the map provides important baseline information that can support the calls for the potential adaptive management of the Eastport MPA to include representative biodiversity and unique and vulnerable habitats of the wider geographic region. We illustrate scenarios for extending the boundaries of the Round Island closure into the area mapped in this study and the resulting increases in protection of benthoscape classes. The benthoscape map can also be used to identify variability in benthoscape pattern and benthic biodiversity across a range of spatial scales. This information can allow researchers to investigate how seafloor patterns can be most accurately used as surrogates for biodiversity in the region. Additionally, coastal seafloor maps such as the one created in this study can support future research questions and policy development related to habitat limitations for species in response to climate change and for identifying critical nursery, spawning and potential settlement areas for threatened or at-risk species.

2.3 Methods

2.3.1 Study Area

Newman Sound (Fig. 2-1) is one of several deep fjords in the region and is separated from Bonavista Bay by a shallow sill (Cumming et al. 1992). The maximum depth of the outer basin is 349 m, while the inner basin has a maximum depth of 63 m. Several rivers and streams flow into the inner basin of Newman Sound where mudflats and estuarine vegetation are common. The adjacent Terra Nova National Park protects 400 km^2 of sheltered inlets, islands, forest and bog habitat that supports a variety of terrestrial mammals including the endemic Newfoundland marten (*Martes americana atrata*; Parks Canada 2018).

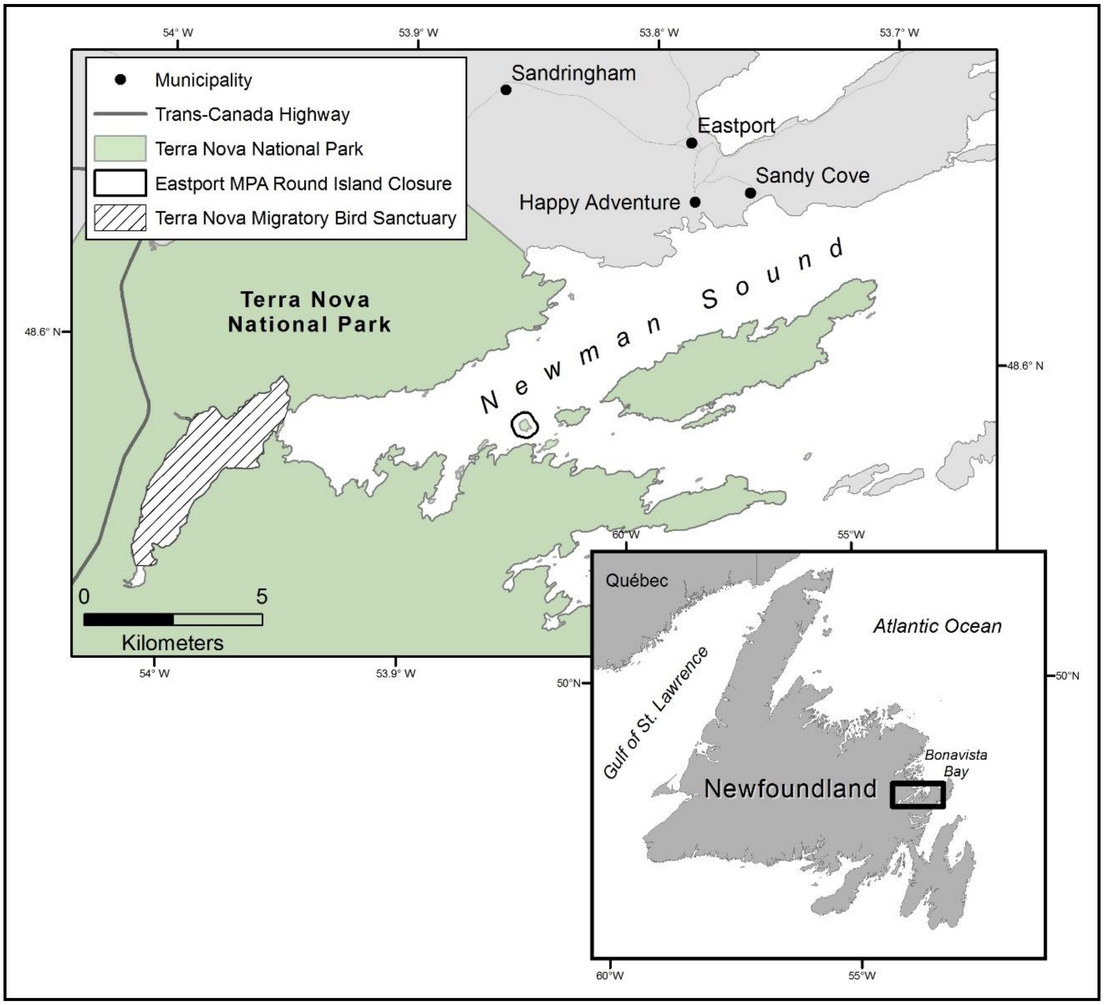

Figure 2-1: Newman Sound, Newfoundland and Labrador. Note that the Terra Nova National park does not extend into the marine area.

2.3.2 Multibeam Echosounder Surveys

In 2003, an acoustic multibeam echosounder (MBES) survey was conducted in Newman Sound by the Canadian Hydrographic Service (CHS; Fig. 2-2). Bathymetric and backscatter data were collected using two Kongsberg MBES systems: an EM1002 (95 kHz) system operated aboard the CCGS Matthew, and an EM3000 (300 kHz) system operated aboard the CCGS Plover. The EM1002 system primarily surveyed the deeper regions of the fjord, while the EM3000 system surveyed the shallow and narrow region separating the inner and outer basins (Copeland et al. 2006). Bathymetric data were post-processed by the Geological Survey of Canada to generate a 10m resolution surface for the region. Uncalibrated raw backscatter data (.all format) from the two MBES systems were post-processed independently for this study in QPS FMGT software v7.7.8 using default settings and Adaptive AVG algorithm that reduces noise in terrains with significant slope variation. Both MBES backscatter coverages were rendered as 10m resolution mosaics (Fig. 2-3).

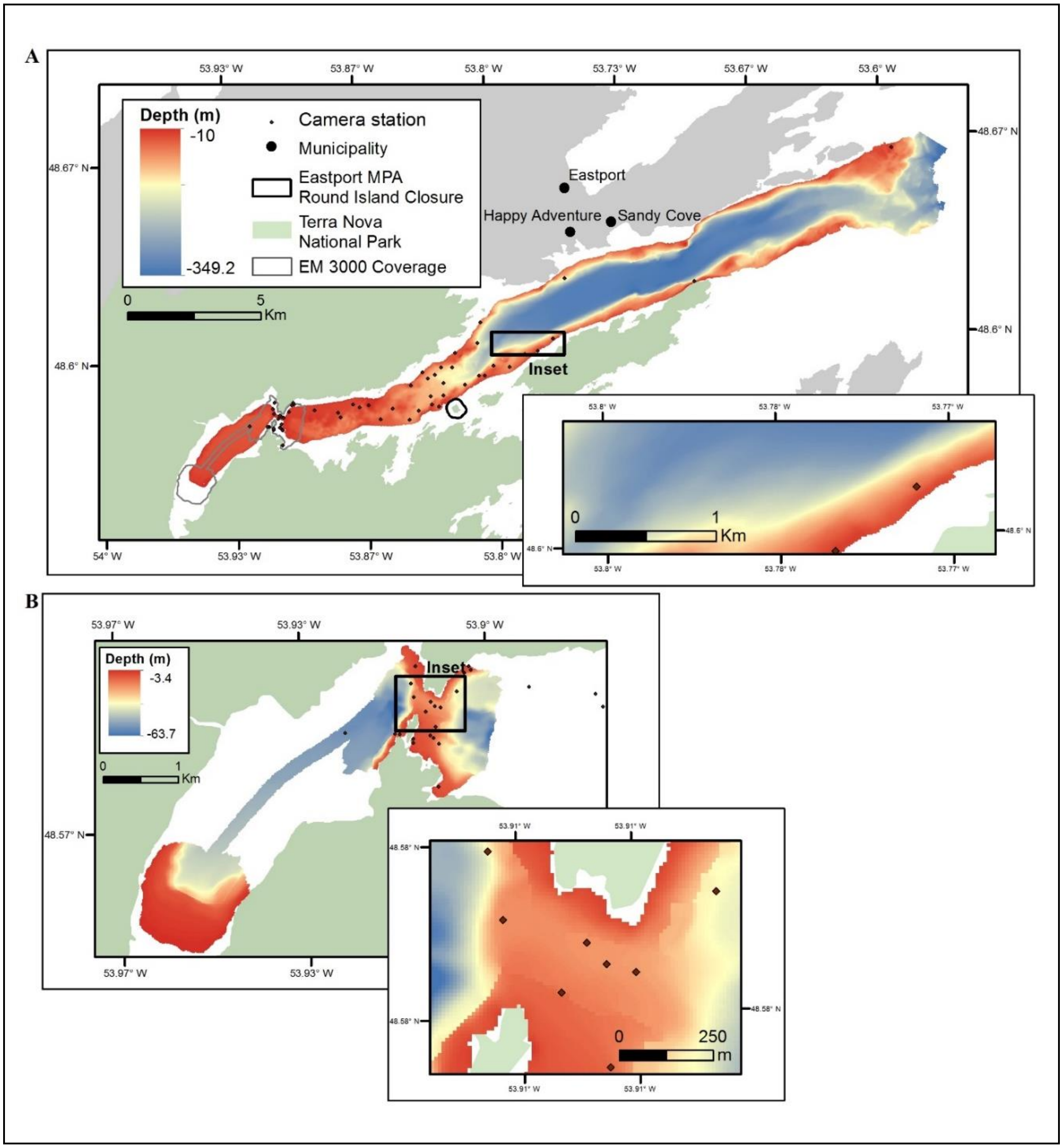

Figure 2-2: Bathymetry data collected during EM1002 (A) and EM3000 (B) multibeam echosounder surveys (located in the western part of the study area). Inset maps illustrate areas with interesting bathymetric features: a large change in depth indicating a high slope region of the fjord (A) and the narrow region separating the inner and outer basins of Newman Sound (B)

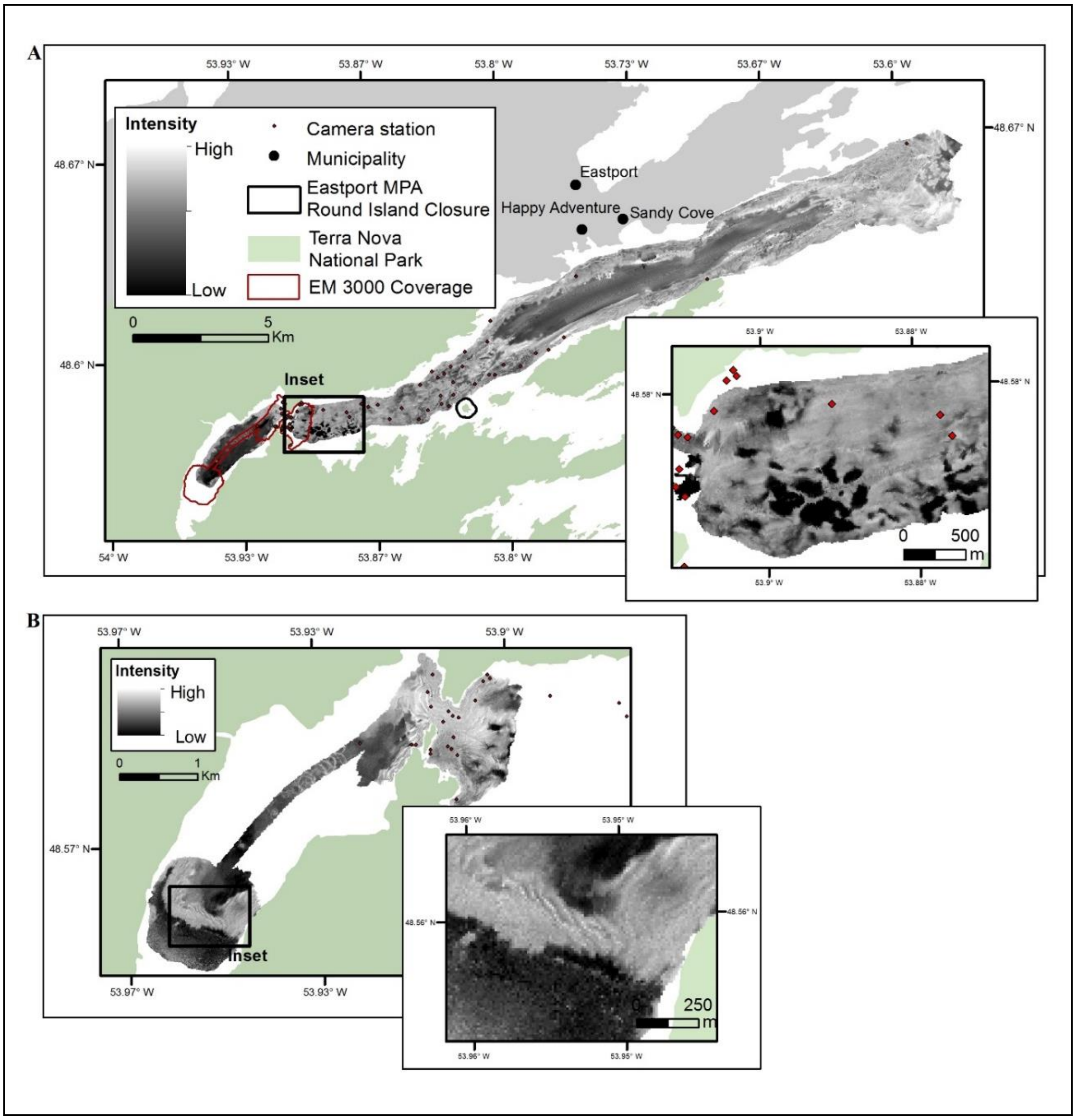

Figure 2-3: Backscatter data collected during EM1002 (A) and EM3000 (B) multibeam echosounder surveys (located in the western part of the study area). Inset maps illustrate areas with interesting backscatter features: patches of shallow seafloor substrate with low intensity backscatter signals surrounded by relatively high intensity backscatter signals (A) and a region where backscatter intensity abruptly transitions from strong to weak (B)

2.3.3 Benthic Ground Truthing Video Surveys

Benthic video surveys were conducted from October 3-13th, 2016 from a 40ft inshore fishing vessel, the 'Jamie Tim N' Trevor', based out of Happy Adventure, NL. Survey stations were selected using a backscatter stratified random sampling design (N=28), constrained by the depth range of the camera system (max depth 170m). Ground truthing efforts focused on the area mapped by the EM1002 system, as extensive archival video data already existed for the shallow region mapped by the EM3000 system (Copeland 2006; Fig. 2-2B). Although a temporal difference exists between the MBES survey, archived video surveys (2003) and recent video surveys (2016) it is unlikely that the MBES bathymetry, backscatter and derivatives would have changed significantly within the sheltered geographic region at the resolution at which the benthoscape map is generated (10m).

Video footage was obtained using a custom drop camera system that recorded in both standard and high definition. The camera system was equipped with mounted LED (Light Emitting Diode) lights and two red lasers positioned 5cm apart to allow for scale measurements. Camera and vessel positions were recorded using a handheld Garmin eTrex10 WAAS (Wide Area Augmentation System) enabled GPS (Global Positioning System), while video transects ran for approximately 4 minutes as the vessel drifted. Positional accuracy for the WAAS enabled GPS was < 3m (Garmin Ltd 2005). High definition seafloor video footage was recorded with a downward facing GoPro Hero 3 Black Edition, and an approximate 1m distance between the camera and the seafloor was maintained through an on-board live feed from a downward facing standard definition 200m-tethered Deep Blue Pro camera.

Still images were extracted from the high definition GoPro footage at 2 second intervals with each transect yielding between 2 and 56 usable images. Images were deemed unusable if they had an obstructed field of view (e.g., disturbed fine sediment), absent scaling lasers, poor illumination or were blurred. An additional 39 classified standard definition images from a previous habitat mapping study (Copeland 2006) were incorporated into the analysis. In total, 238 images from 57 stations were used to classify the seafloor of Newman Sound. Image location was recorded using the timestamps from the high definition footage and continuous GPS overlay of vessel position. Image area was measured using Image J Software, and substrates were classified based on biophysical characteristics.

2.3.4 Image Classification

Seafloor images were classified into benthoscape classes based on their biophysical characteristics, including the dominant sediment type and the presence/absence of encrusting coralline algae. This approach is comparable to the classification of terrestrial landscapes from remote sensing datasets (see Zajac et al. 2003; Zajac 2008; Brown et al. 2012). Each image was classified into one of eight benthoscape classes that captured the range of variability across ground-truthed images (Table 2-1; Fig. 2-4). The eight benthoscape classes were: (A) bedrock, (B) deep pebble/cobble, (C) gravelly muddy sand, (D) mixed boulder, (E) mud, (F) shallow pebble/cobble, (G) rhodolith and (H) sand. The classification mirrored that of ground truthing surveys (Copeland 2006) to ensure that the two datasets were comparable.

Table 2-1: Benthoscape classes used to characterize the seafloor in Newman Sound. Depth, slope and backscatter measurements were extracted when image locations were overlaid on MBES bathymetry (10m resolution)

Benthoscape	Biophysical characteristics	Mean depth(m) [range]	Mean Slope(deg) [range]	Mean Backscatter(dB) [range]	# of images
Bedrock	Solid exposed bedrock of fjord walls	83 [163–15]	32 [6–65]	-14.25 [-25 – -9]	30
Deep pebble/cobble	>50% cobbles/gravel. Encrusting coralline algae absent	106 [161–72]	18 [12– 23]	-13 [-20 – -6]	16
Gravelly Muddy Sand	Mixed gravel, mud, sand	46 [68–7]	7 [1–14]	-14 [-21 – -8]	44
Mixed Boulder	Boulder >25%. Mixed cobble/ gravel/sand. Coralline algae present	20 [42–7]	6 [1–19]	-12 [-15 – -9]	20
Mud	Mud	113 [159–47]	8 [0–27]	-16 [-26 - -8]	85
Shallow pebble/cobble	>50% cobbles/gravel Encrusting coralline algae present	50 [77–33]	10 [2–23]	-12 [-14 – -9]	25
Rhodolith	>50% Rhodolith coverage	17 [20–14]	2 [1-7]	-17 [-22 – -9]	8
Sand	Sand	16 [56–7]	3 [0–11]	-8* [-9 – -5]*	10

*indicates backscatter measurements obtained from EM3000 MBES. All others measured from EM1002 MBES.

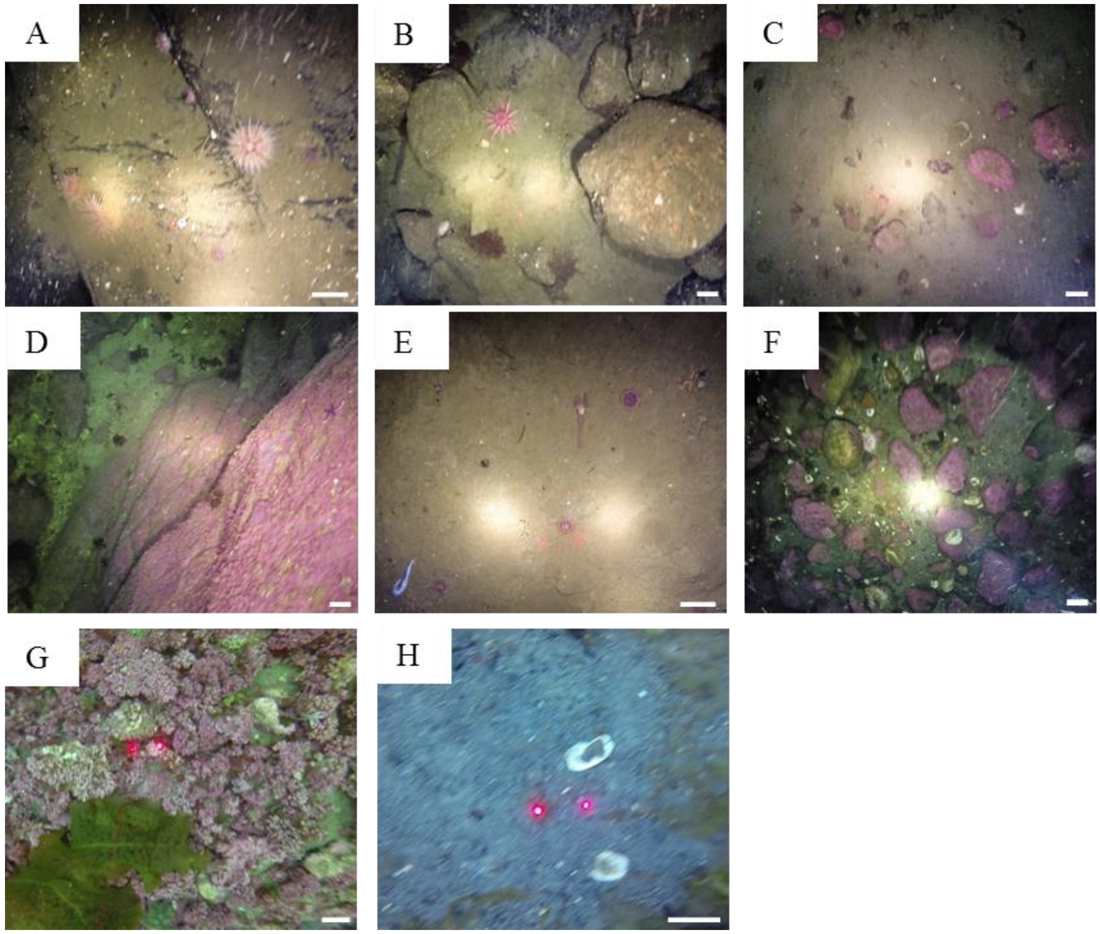

Figure 2-4: Images representing benthoscape classes in Newman Sound: (A) bedrock, (B) deep pebble/cobble (C) gravelly muddy sand, (D) mixed boulder, (E) mud, (F) shallow pebble/cobble, (G) rhodolith, and (H) sand. Scale bar = 5cm

2.3.5 Unsupervised Segmentation of MBES Data

Bathymetry, seafloor slope and backscatter have been used extensively in benthic mapping studies due to their spatially continuous nature in terms of map coverage, and also their important role in separating distinct benthic habitats (Brown et al. 2011; Ierodiaconou et al. 2011; Che Hasan et al. 2014; Hill et al. 2014). These variables have been used specifically to map coastal fjord environments (Cochrane et al. 2011; Copeland et al. 2013), with slope being of particular interest in a context where steep walls and current-winnowed gravel have been shown to support unique and diverse benthic communities (Dale et al. 1989). In this study, the two MBES datasets were segmented separately based on bathymetry, backscatter and slope using the ISO Cluster Unsupervised Classification Tool (ArcGIS 10.3.1). The tool segments a series of input raster bands by combining an iterative-self-organizing (ISO) algorithm and maximum likelihood classification. The ISO segmentation was limited to areas within the depth range of the ground-truthed video surveys (≤170 m). This method has been shown to be an effective way of segmenting MBES data (Calvert et al. 2014). The ISO Cluster unsupervised classification tool allows the user to select the number of output classes. The optimal number of classes for both the EM1002 and EM3000 MBES coverages was determined by using the maximum number of classes possible while maintaining at least two ground-truthed images within each class. This was not possible for one class within the EM3000 coverage due to the limited number of samples within the inner basin of Newman Sound.

The locations of classified ground-truthed images were then overlain on the map derived from MBES segmentation and an error matrix was generated to determine the accuracy of the classification. The error matrix compares observed substrate classes (from image analysis) with the classes derived from MBES segmentation. Three standard measures of accuracy were calculated: (1) overall accuracy measures the percentage of correctly classified reference pixels when overlain on the MBES segmentation, (2) benthoscape class-specific measures of accuracy, measuring the probability that a reference pixel is correctly classified (Producer's Accuracy) and (3) how likely a map user is to encounter a correctly predicted benthoscape (User's Accuracy) (Diesing et al. 2016). A methodological diagram (Fig. 2-5) illustrates the general workflow followed for integrating the two MBES datasets and comparing the final Newman Sound benthoscape classification with epifaunal assemblage analysis (described below).

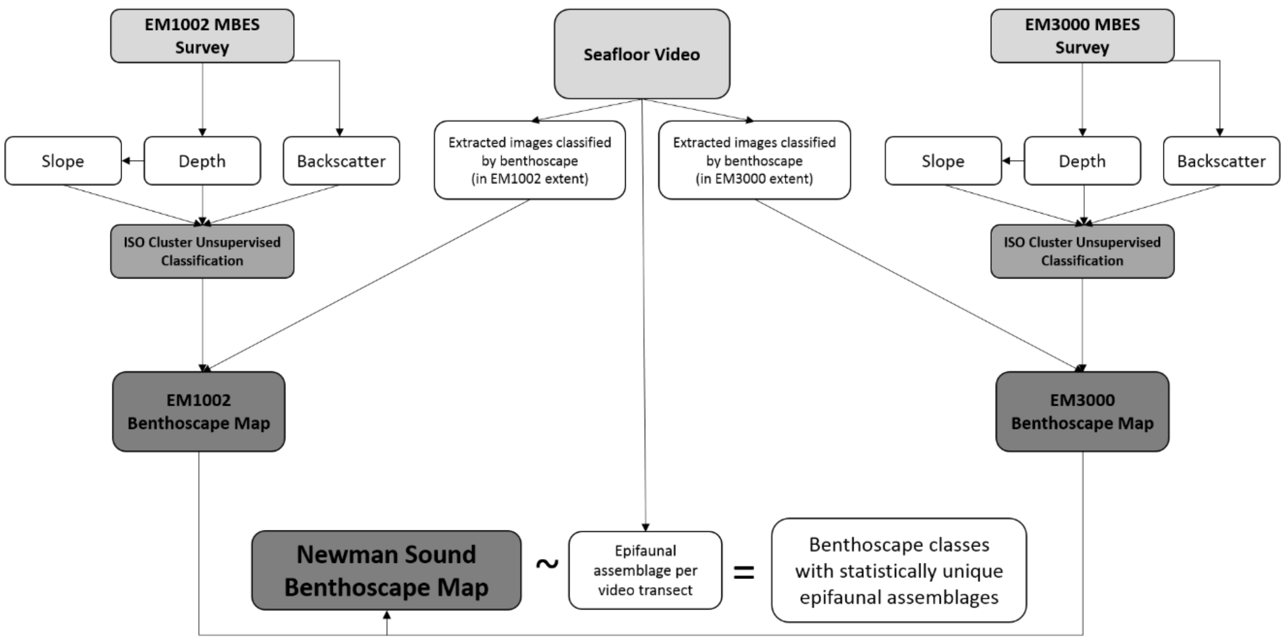

Figure 2-5: Methodological diagram representing the workflow followed for producing benthoscape map of Newman Sound using two multibeam echosounder datasets and seafloor video surveys

2.3.6 Epifaunal Assemblage Analysis

The abundance of benthic fauna was also recorded for the entire transect length using continuous video footage, and we identified organisms to the lowest possible taxonomic level. To explore potential relationships between benthoscape class and organism assemblages, multivariate statistical analyses were performed using the software package PRIMER (Plymouth Routines in Multivariate Ecological Research) v7 (Clark and Gorley 2015). In order to allow for the integration of the archival data, abundance data were presence-absence transformed and combined for the entire transect, using the mid-point to represent each transect position. Transect length in all samples in the archived and recent video footage varied (mean transect length = 110m). Transect length from the archived video data varied between approximately 30 – 500m while transect length from recent video footage varied between approximately 60 – 200m. As such, species presence/absence was not standardized by transect length. Nonetheless, incorporating this archival video footage provides insight into the potential relationship between species composition and benthoscape class, although it is likely that species richness and abundance estimates standardized by transect length would more adequately capture these patterns. Transects were excluded from the analysis if they crossed the boundary between two predicted benthoscape classes. A Bray-Curtis Similarity matrix was generated, and non-metric multi-dimensional scaling (nMDS), analysis of similarity (ANOSIM) and similarity percentage (SIMPER) tests were run in PRIMER to assess the potential impact of benthoscape class on benthic community composition.

2.4 Results

2.4.1 Unsupervised Substrate Segmentation of MBES Data

Unsupervised classification of the EM1002 MBES coverage resulted in thirteen substrate classes characterized by their depth, slope and backscatter values. The classification is based solely on the inputs derived from the MBES surveys (bathymetry, backscatter and slope) and classes are not defined using the benthoscape classes from ground-truthing surveys (Fig. 2-4). The thirteen substrate classes are assigned to a benthoscape class based on the dominant *in-situ* image class (described below). The classes derived from the unsupervised segmentation of the EM1002 coverage which covers the majority of the fjord, aligns with what would be expected in terms of the geomorphology of fjord environments: separate classes that encompass flat deep basins, shallow sills and steep fjord walls (Fig. 2-6A; Syvitski and Shaw 1995).

Four substrate classes were identified in the EM3000 coverage, with segmentation revealing a flat, inner basin with more variable substrate classes along the perimeter and at the shallow sill separating the inner and outer regions of the fjord (Fig. 2-6B).

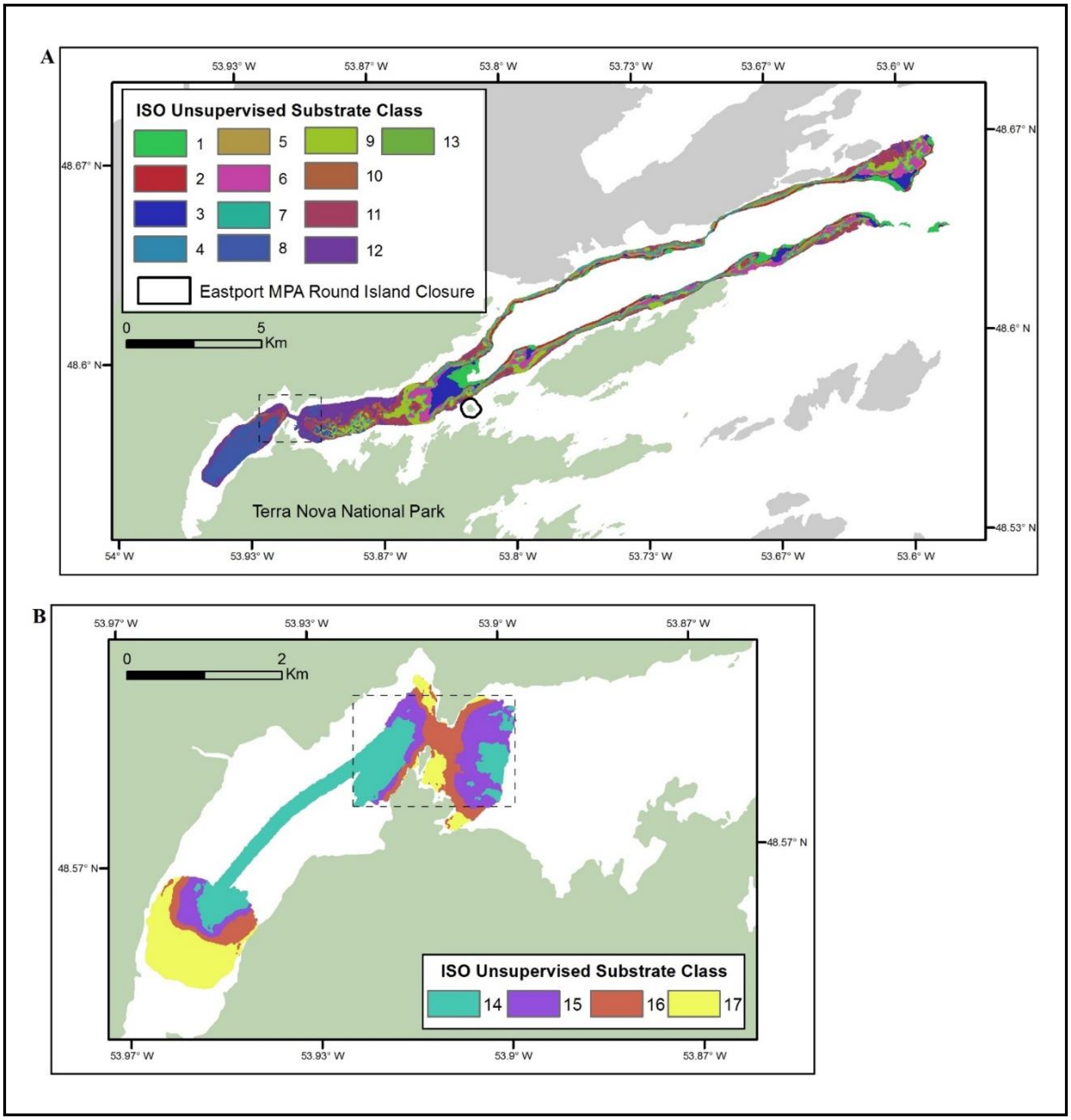

Figure 2-6: Results of ISO unsupervised segmentation using bathymetry, backscatter and slope for EM1002 MBES coverage within the depth range of the drop camera system (max depth 170m) (A) and EM3000 coverage (B). Dashed line indicates region where two coverages overlap

The error matrix compared the MBES segmentation against the classified images (Table 2-2). Ground-truthed images obtained where the two MBES coverages overlapped (the narrow region between the inner and outer basins) were assigned to MBES segmentation classes from the EM1002 coverage, as the EM1002 segmentation provided more variability of classes within the overlapping region (Fig. 2-7) and thus more classes to compare against ground-truthed images.

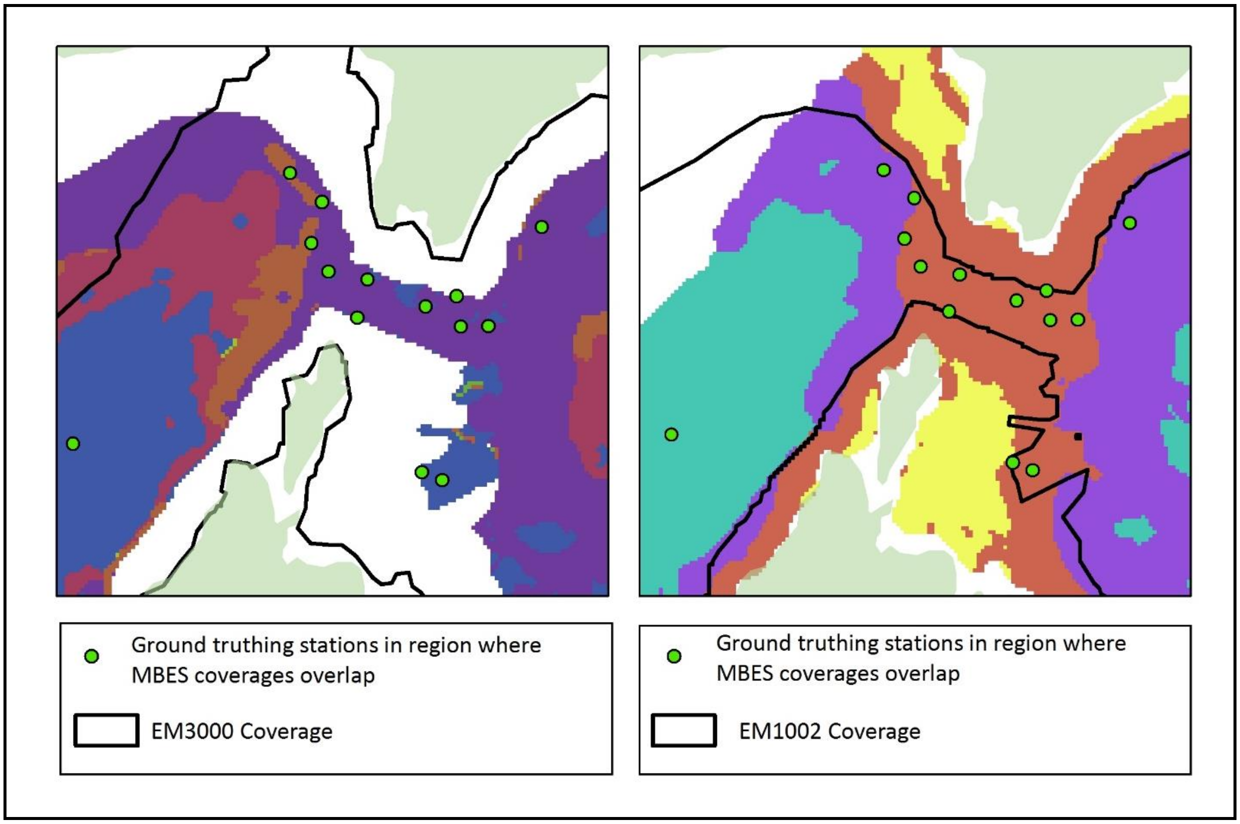

Figure 2-7: Locations of ground truthing in region where EM1002 (left) and EM3000 (right) MBES coverages overlap. Colours on each map represent the same unique substrate classes as Fig. 2-6

The seventeen classes derived from the two MBES coverages were reduced to seven by merging segmented classes based on their dominant *in-situ* image class. Several benthoscape classifications from the images corresponded closely with a single MBES segmentation class, while others required merging. Images classified as deep pebble/cobble corresponded to MBES segmentation class 7 (50.0% agreement) – see

Table 2-2 for details on each class, which was typically located at the outer edge of regions classified as mud. Gravelly muddy sand (GMS) images corresponded to MBES segmentation class 11 (75.0% agreement). Images classified as mixed boulder corresponded to MBES segmentation classes 12, 15 and 16 (80.0% agreement) and were predominant in shallow water regions towards the head of the fjord. Images classified as rhodolith were common in MBES segmentation classes 15 and 16, and were only found at the narrow region between the inner and outer fjord basins. Rhodolith was not acoustically distinguishable from mixed boulder and without additional ground truthing surveys in other regions we cannot assume that the rhodolith bed is uniform across the extent of classes 12, 15 and 16. Previous studies have, however, identified the extent of a rhodolith bed at the narrow region separating inner and outer Newman Sound (Copeland 2006). Mud corresponded to MBES segmentation classes 1, 2, 3, 5, 6, 8, 9 and 14 (98.8% agreement) and was found in the flat inner and outer central basins of the fjord. Substrate class 14 was validated by a single archived image (mud) however the low slope (mean = 1.1 degrees) and low (relative, uncalibrated) backscatter return in the substrate class suggests that mud is an appropriate classification. Bedrock corresponded to MBES segmentation classes 4 and 13 (53.3% agreement) and was predominant in deep, high slope regions toward the periphery of the fjord. Shallow pebble/cobble did not clearly correspond to any MBES segmentation class and was acoustically indistinguishable from gravelly muddy sand (GMS). The number of shallow pebble/cobble and GMS images differed by a single image when overlain on MBES segmentation class 10 (shallow pebble/cobble = 11 images; GMS = 12 images). As a result, MBES segmentation class 10 was retained to capture the transition between GMS and more complex mixed boulder substrates. Images classified as sand corresponded to map class 17 (60.0% agreement) and exhibited a patchy distribution (Fig. 2-6).

Table 2-2: Error Matrix for final benthoscape classification of Newman Sound, NL

	MBES Segmentation (ISO Unsupervised Classification) Classes								
Ground-truth (benthoscape class)	4+13	7	11	12+ 15+ 16	1+2+3 + 5+6+8 + 9+14	10	17	Row total (no. of images)	Producer's accuracy (%)
Bedrock	16	3	0	5	5	1	0	30	53.3
Deep pebble/cobble	0	8	0	0	8	0	0	16	50.0
Gravelly Muddy Sand	0	0	21	4	5	12	2	44	75.0
Mixed Boulder	0	0	0	16	0	2	2	20	80.0
Mud	0	0	1	0	84	0	0	85	98.8
Shallow pebble/ cobble	0	0	7	5	2	11	0	25	44.0
Rhodolith	0	0	0	6	2	0	0	8	n/a
Sand	0	0	0	1	2	1	6	10	60.0
Grand total (no. of pixels)	16	11	29	37	108	27	10	238	
User's accuracy (%)	100	72.7	72.4	43.2	77.8	85.2	60.0		
								Overall accuracy	73.1%

The final benthoscape map of Newman Sound waters shallower than 170m depth (Fig. 2-8) was produced by combining the two MBES coverages. In the region where the two MBES coverages overlapped, the EM 1002 coverage was used as it provided a greater number of MBES segmentation classes for ground-truthing comparison and the majority of the other ground truthing stations were also located in the EM 1002 extent. The final map indicates that the region is dominated by mud (54.22%) and mixed boulder (12.85%). Areas classified as mixed boulder occurred in some of the shallowest regions of the MBES coverage. Based solely on the results of this mapping project, the amount of mixed boulder in Newman Sound is likely underestimated because the MBES coverage does not extend to shallow regions adjacent to the shoreline.

The greatest amount of confusion occurred between GMS/shallow pebble/cobble and rhodolith/mixed boulder. Images classified as rhodolith and pebble cobble were acoustically indistinguishable from other benthoscape classes, and were thus not included in the final benthoscape map of Newman Sound except where shallow pebble/cobble and GMS were combined to capture the transition between GMS and more complex substrates. The overall accuracy of the final map after merging the two MBES coverages was 73.1% (Table 2-2).

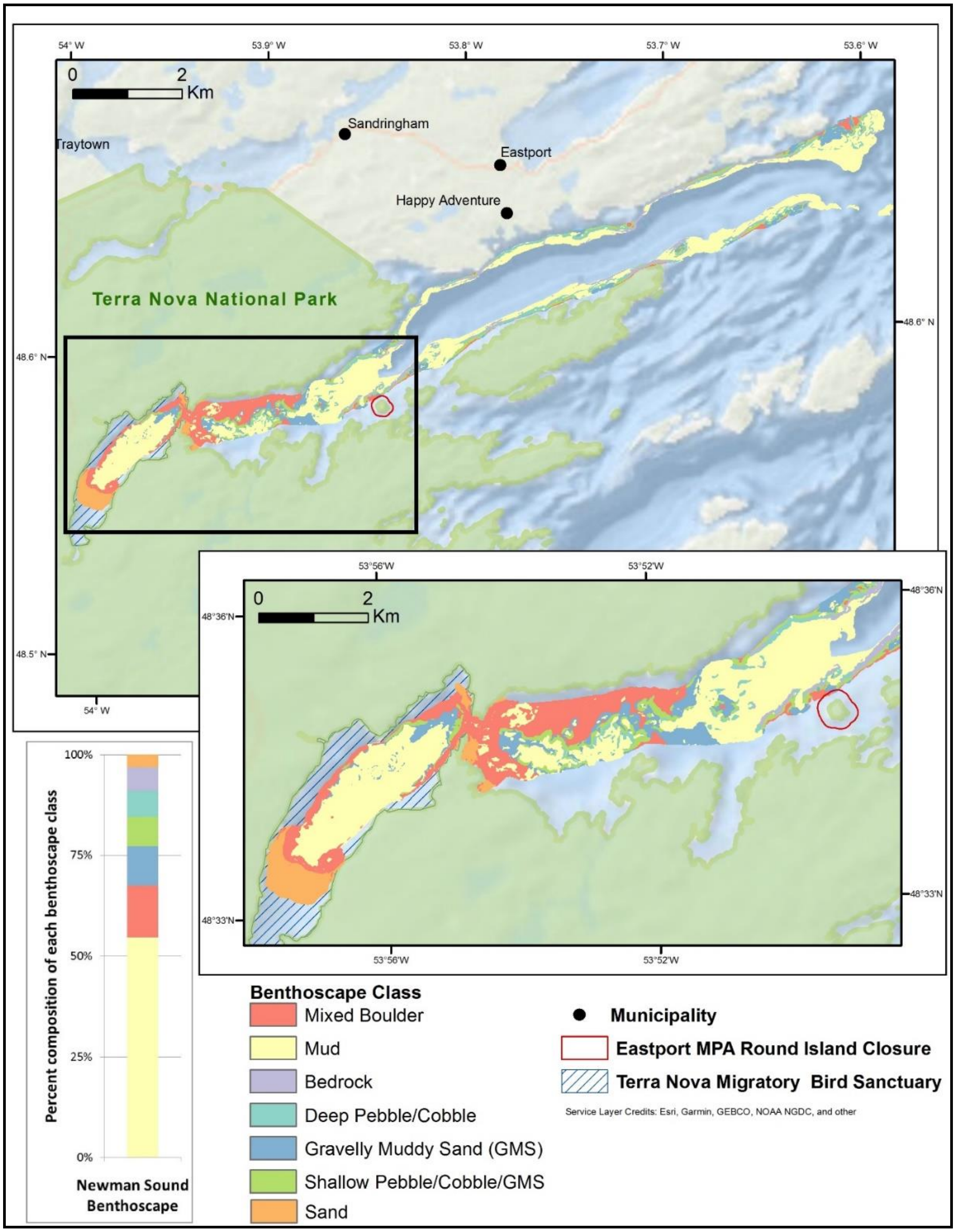

Figure 2-8: Final benthoscape map of Newman Sound (depth <170m). Bar chart illustrates the percent contribution of each benthoscape class in the mapped area

2.4.2 Epifaunal Assemblage Analyses

A number of distinct patterns emerged in the nMDS plot of benthic community data in Newman Sound (n=55) (Fig. 2-9). The nMDS plot represents the dissimilarity of epifaunal assemblages in multidimensional space in a reduced number of dimensions to facilitate visualization and interpretation. However, because species data for the entire length of each transect had to be merged into a single point to allow for the incorporation of archival data, the sample size was much smaller and may not fully capture the potential impact of benthoscape class on community composition. The nMDS plot does however suggest a gradual shift from soft substrates (mud) on the right side of the ordination plot (Fig. 2-9) to more complex and consolidated substrates (mixed boulder) on the left.

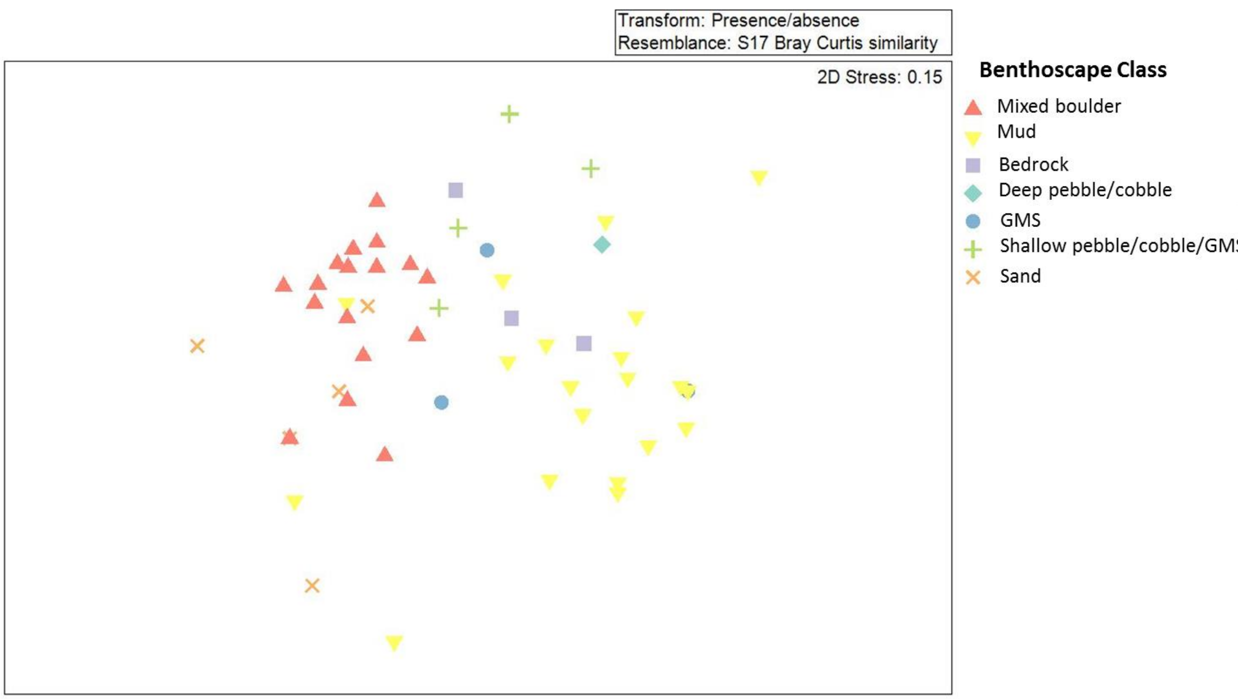

Figure 2-9: nMDS ordination plot of epifaunal assemblage data extracted from seafloor video surveys in Newman Sound.

ANOSIM results (Table 2-3) reveal significantly different species assemblages between mixed boulder and mud (R = 0.619, p < 0.001), mixed boulder and bedrock (R = 0.620, p < 0.004) mixed boulder and GMS (R = 0.648, p = 0.003), mixed boulder and shallow pebble /cobble/GMS (R= 0.606, p = 0.003) and mud and sand (R= 0.549, p = 0.001). Due to the low representativity of some benthoscape classes in the multivariate dataset, pairwise ANOSIM results for several benthoscape classes are not valid because the number of possible permutations was too low.

Table 2-3: Analysis of similarity of observed species between benthoscape classes. Global R = 0.465.

Benthoscape Class	**Mixed Boulder**	**Mud**	**Bedrock**	**Deep pebble/ cobble**	**GMS**	**Shallow pebble /cobble / GMS**
Mud	0.619**					
Bedrock	0.620**	0.113				
Deep pebble/cobble	-	-	-			
GMS	0.648**	0	-	-		
Shallow pebble/cobble/GMS	0.606**	-	-	-	-	
Sand	0.296	0.549**	-	-	-	-

- number of possible permutations too low; ** significant at p < 0.005

SIMPER results indicate taxa that contribute most towards the similarity of samples within each benthoscape class and the overall average similarity of epifaunal composition within each class (Table 2-4). Average similarity within each benthoscape class was generally low, ranging from 33% to 56% (Table 2-4).

Table 2-4: Results from SIMPER analyses of epifaunal assemblage data (presence-absence transformed). The main characterizing species from each benthoscape class are listed. Similarity percentage, cumulative similarity percentage for each species and the overall average similarity between samples from within each benthoscape class are listed.

Benthoscape Class	Species	%	Cumulative %	Average similarity
Mixed Boulder	*Asterias vulgaris*	28.5	28.5	56%
	Strongleyocentrotus droebachiensis	28.1	56.6	
	Metridium senile	21.0	77.1	
Mud	*Ptychogastria polaris*	22.1	22.1	41%
	Ophiopholis aculeata	20.3	42.4	
	Chionoecetes opilio	17.2	59.6	
	Strongleyocentrotus droebachiensis	13.9	73.5	
Bedrock	*Strongleyocentrotus droebachiensis*	27.7	27.7	44%
	Ophiopholis aculeata	27.7	55.4	
	Chionoecetes opilio	8.4	63.9	
Deep Pebble/Cobble	-	-	-	-
Gravelly Muddy Sand	*Strongleyocentrotus droebachiensis*	25.0	25.0	33%
	Chionoecetes opilio	25.0	50.0	
	Ophiopholis aculeata	25.0	75.0	
Shallow Pebble/Cobble/GMS	*Strongleyocentrotus droebachiensis*	62.5	62.5	37%
	Urticina felina	11.3	73.8	
Sand	*Echinarachinus parma*	33.2	33.2	47%
	Asterias vulgaris	30.6	63.7	
	Psuedopleuronectes americanus	16.6	80.3	

In terms of pairs of benthoscape classes with distinct epifaunal assemblages (Table 2-3), dissimilarity in faunal composition was driven by the presence of several key species. Dissimilarity in epifaunal assemblages in mixed boulder and mud benthoscape classes (average dissimilarity = 78.1%) was primarily driven by plumose anemones (*Metridium senile*) and northern seastars (*Asterias vulgaris*), species commonly found on mixed boulder substrates, and epibenthic trachymedusae (*Ptychogastria polaris*), common on mud substrates. Plumose anemones, snow crab (*Chionoecetes opilio*) and daisy brittle stars (*Ophiopholis aculeata*) were the highest contributors to dissimilarity between mixed boulder and GMS faunal compositions (average dissimilarity = 69.3%). Dissimilarity between mixed boulder and bedrock (average dissimilarity = 62.92%) was primarily driven by the presence of daisy brittle stars, and unidentified white anemones. Differences in faunal composition between mixed boulder and shallow pebble/cobble/GMS (average dissimilarity = 63.6%) was driven by plumose anemones, northern seastars and dahlia anemones (*Urticina felina*). Common sand dollars (*Echinarachnius parma*), northern seastars and the hydrozoan *P. polaris* were the highest contributors to dissimilarity between mud and sand benthoscape classes (average dissimilarity = 83.64%).

2.5 Discussion

Seven benthoscape classes and two statistically distinct epifaunal assemblages were identified in Newman Sound, NL by integrating two MBES datasets with new and archived seafloor video surveys. The final predicted benthoscape map (Fig. 2-8) contributes to seafloor mapping efforts in Newfoundland and Labrador and improves our understanding of regional seafloor substrates and habitats in Bonavista Bay, an area that has been prioritized for conservation regionally.

2.5.1 Epifaunal Assemblage Analysis

Mud and mixed boulder benthoscape classes supported statistically distinct epifaunal assemblages. However, due to underrepresented benthoscape classes in the multivariate analyses, it is possible that the species rich bedrock walls and deep pebble/cobble benthoscape classes also support distinct communities. Furthermore, species abundances may be in fact more important for detecting patterns and variation in epifaunal assemblage across benthoscape classes as habitat distinction within the boundaries of the MPA was often driven by the abundance of green urchins (*Strongleyocentrotus droebachiensis*), northern seastars (*Asterias vulgaris*) and brittle stars (*Ophiopholis* sp.; (Novaczek et al. 2017a)). Transforming data to presence/absence can make detecting patterns in species assemblages difficult, particularly in the case of Newman Sound where generalist species such as green urchins and brittle stars were found across all substrates. However, these analyses do provide insight into the potential relationships between species composition and benthoscape class. ANOSIM results revealed distinct differences in epifaunal assemblages between mixed boulder and mud benthoscape classes ($R = 0.619$, $p < 0.001$) – a pattern likely driven by contrasting sediment composition (hard vs. soft substrates) and depth (shallow vs. deep). This pattern, along with the low similarity of epifaunal assemblage composition within each benthoscape class (33-56%) aligns with other studies that aim to classify seafloor habitats based on benthic community assemblages (McGonigle et al. 2009; Brown et al. 2012; Lacharité and Brown *in press*). While these results fail to capture gradients of epifaunal assemblages in transitional habitats, a higher density sampling effort measuring species abundance could potentially capture this variation.

2.5.2 Mapping for Conservation Planning

Seafloor mapping has become increasingly important for informing conservation planning. Design, management and monitoring of MPAs and other area-based conservation measures as well as the identification of Ecologically and Biologically Significant Areas (EBSAs) can all be informed by seafloor maps. Recent habitat characterization and mapping activities in the nearby Eastport MPA determined that the MPA does little to conserve habitats and biodiversity representative of the broader region of Newman Sound (Novaczek et al. 2017a). Novaczek et al. (2017a) identified four distinct benthic substrates within the boundaries of the MPA: “shallow rocky,” “sand and cobble” and “boulder/bedrock” and “sand.” Several of the benthoscape classes identified in this study are either under-represented or unrepresented within the boundaries of the Eastport MPA. The MPA is dominated by shallow rocky substrates, which correspond to the ‘mixed boulder’ classification in this study (48.9% of the combined closures; 86.3% of the Round Island closure). The shallow pebble cobble and GMS substrates identified in this study are similar to the “sand and cobble” habitat identified by Novaczek et al. (2017a). No sandy substrate was predicted to be protected within the boundaries of the Round Island closure, while sand/cobble and bedrock features combined made up only a small portion of substrates within the closure (13.7%; Novaczek et al. 2017a). Furthermore, 95.7% of the Round Island closure is shallower than 80m and within the photic zone (Novaczek et al. 2017a). Consequently, deep-water substrates identified in this study (mud and deep pebble/cobble) are unrepresented.

Extending the boundaries of the Round Island closure into the area mapped in this study would result in at least some protection of nearly all benthoscape classes identified in Newman Sound (Fig. 2-10). This would increase the protection of benthoscape classes and their associated biodiversity that are under or unrepresented based on the current boundaries of the MPA, and result in a more ecologically representative MPA - a key element of effective conservation planning. Five boundary expansion scenarios and the amount (km^2) of each benthoscape class included in each scenario are illustrated in Fig. 2-10. These incremental boundary expansions are useful for determining the proportion of each benthoscape class that would be included in potential adaptive management scenarios, and how the MPA could better represent benthoscape scale patterns in Newman Sound. For each boundary expansion scenario, it is only possible to quantify the amount of each benthoscape class within the mapped area. Mapping the additional benthoscape classes outside the extent of the MBES (i.e. currently unmapped) would be an important next step if these boundary expansion scenarios were considered. Alternatively, expanding the boundaries in a NE-SW direction (i.e., only into the area mapped in this study) could also be considered.

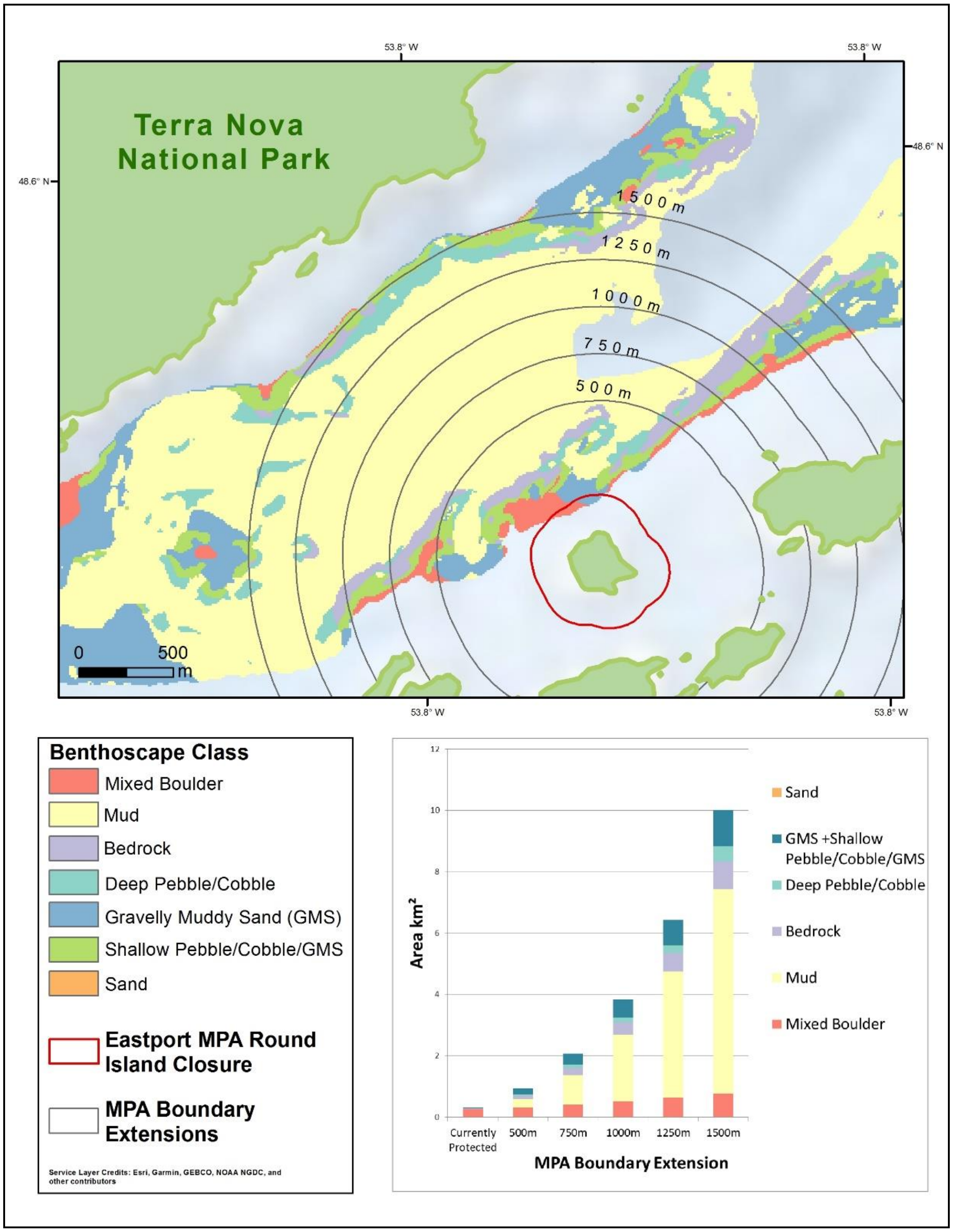

Figure 2-10: Boundary expansion scenarios for the Round Island closure. Bar graph illustrates the amount (km^2) of each benthoscape class that would be included in various scenarios. Area measurements only include the region mapped in this study. Current protection levels obtained from Novaczek et al. 2017a

The final benthoscape map produced in this study also identifies the location and extent of additional shallow rocky substrate (mixed boulder benthoscape) in Newman Sound. This substrate is commonly associated with juvenile and adult lobster habitat and dominates the MPA (Novaczek et al. 2017a). Increasing the amount of lobster habitat protected within the boundaries of the MPA could potentially result in enhancements to the local fishery by broadening the area in which MPA benefits, such as increased lobster density and a broadened size structure that includes larger, more fecund lobsters, have been observed (Stanley et al. 2018).

The inability to acoustically distinguish rhodolith and mixed boulder benthoscape classes, means that alternative methods such as SCUBA surveys are likely required to delineate the extent rhodolith beds in the region. Previous research (Copeland 2006) used SCUBA surveys to identify the extent of a rhodolith bed located at the shallow narrow region separating the inner and outer basins of Newman Sound. Rhodolith beds form complex habitats that support a high diversity of invertebrates, fish and algae (Steller et al. 2003; Copeland et al. 2008; Hernandez-Kantun et al. 2017 and references therein) and also provide refuge and feeding grounds for juvenile Atlantic cod (Kamenos et al. 2004). Rhodolith beds are sensitive to bottom contact fishing activities (Hall-Spencer and Moore 2000; Kamenos et al. 2003), waste build-up from aquaculture operations (Hall-Spencer et al. 2006) and are particularly vulnerable to impacts from ocean acidification (Martin and Hall-Spencer 2017 and references therein). The rhodolith bed in Newman Sound is also in close proximity to sizeable eelgrass (*Zostera marina*) beds located in the inner sound

(Rao et al. 2014). However, neither eelgrass nor rhodolith beds are protected by the Eastport MPA despite their importance in providing habitat and refuge for a wide range of species.

Newman Sound is an ecologically unique and well-studied region in Newfoundland and Labrador. However, the Eastport MPA protects very little of its diversity despite our knowledge of the importance of conserving representative biodiversity, benthoscape classes and unique habitats such as eelgrass and rhodolith beds. The small size of the Eastport MPA is a clear limitation. However, the benthoscape map produced in this study coupled with extensive research on eelgrass beds in the region and our knowledge of the location of the rhodolith bed provide valuable information that can inform the potential adaptive management and expansion of the MPA.

2.5.3 Mapping to Support Future Research Activities in Newman Sound

A recent study in Newman Sound used nearshore baited cameras to examine how seabed habitat influences fish, Atlantic rock crab and American lobster community composition (Dalley et al. 2017). The authors identified four common nearshore substrates: bedrock (high relief boulder and bedrock outcrops), sand-pebble, cobble and eelgrass, in addition to one anthropogenic habitat (wharf). Significant differences in species diversity and relative abundance were observed between sand-pebble and bedrock substrates. Dissimilarity was driven by a variety of fish species, including cunner (*Tautogolabrus adspersus*), age-1 Greenland cod (*Gadus microcephalus ogac*), winter flounder (*Psuedopleuronectes americanus*) and Atlantic cod (*G. morhua*). American lobsters were observed primarily on bedrock substrates, where Atlantic rock crabs were absent (Dalley et al. 2017). The baited cameras used by Dalley et al. (2017) were limited to the depth at which complementary beach seining sampling was done (max depth 3m), which is shallower than the MBES data used in this study. Although it is difficult to compare the results of baited vs. non-baited sampling methods, the final benthoscape map produced in this study could inform future studies that examine how fish communities change in response to substrate beyond the near-shore environment.

The Atlantic cod population in Newman Sound has been extensively studied in terms of habitat use and movement (Cote et al. 2001; 2004), predator-prey dynamics (Linehan et al. 2001; Gorman et al. 2009), and relationships between cod density and eelgrass patchiness (Thistle et al. 2010). This research was almost exclusively done in shallow regions of the inner basin of Newman Sound. As such, our understanding of cod movement and habitat associations in deeper regions of Newman Sound is limited. The Newman Sound benthoscape map provides valuable information that can inform future studies on Atlantic cod movement and variability. Specifically, the benthoscape map can provide insight as to how distribution and movement patterns relate to seafloor characteristics and benthoscape classes.

This benthoscape map provides important baseline information for management and monitoring activities related to the Eastport MPA. An additional objective of this study was to demonstrate the application of a seafloor mapping method that combines MBES backscatter data from two sources, a pixel based unsupervised classification and seafloor image analysis. Methods for integrating MBES backscatter from multiple sources and incorporating archival data are valuable, particularly due to high costs associated with at-sea surveying. Maps that describe seafloor substrates and habitats are crucial for effective marine conservation and management, and can facilitate implementation, design and management of marine conservation objectives, particularly in coastal environments where anthropogenic stressors and threats are concentrated.

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Chapter 3 – Integrating fine-scale seafloor mapping and seascape ecology metrics into marine conservation prioritization

3.1 Abstract

Marine protected area (MPA) planning often relies on a number of scientific principles with the goal of ensuring that the area selected for conservation is likely to be effective. Capturing ecological processes, such as seascape connectivity in MPA planning has become a central focus in recent years. However, studies that aim to integrate seascape connectivity and conservation prioritization are dominated by studies in shallow, biogenic habitats while little attention has been paid to benthic seascapes (benthoscapes) that require acoustic techniques to map. Advances in acoustic seafloor mapping strategies allow for the production of fine-scale maps that yield the spatial information required to extend connectivity and conservation prioritization research into these environments. Here, we propose and test a method that combines benthoscape mapping, connectivity analysis and conservation prioritization. Using a case study in Eastern Canada, we quantified the composition and configuration of the benthoscape using patch size, and connectivity metrics. We then used Marxan, a widely used conservation prioritization tool, to set representativity targets and compared outputs that included and excluded the prioritization of benthoscape connectivity. Results illustrate how larger, well-connected patches of seafloor substrate and habitat can be preferentially selected in Marxan analyses when benthoscape configuration is considered. We also demonstrate the flexibility of the method for including species-specific movement data when available. This illustrates how

benthic habitat mapping can be coupled with connectivity analyses to extend the field that links seascape connectivity and conservation prioritization into regions that require acoustic techniques to map, with the goal of better supporting biodiversity conservation in these environments.

3.2 Introduction

Direct and indirect human stressors continue to have profound impacts on biodiversity. Recent reports indicate that global wildlife populations have declined by 60 percent on average in less than 50 years (WWF 2018). In response to this decline, calls for increased protection of coastal and marine areas have become widespread (e.g. CBD-UNEP 2010). Marine habitats and biodiversity face threats from activities such as pollution, overfishing, marine shipping, resource extraction, as well as from terrestrial based activities such as agricultural runoff and coastal development (Halpern et al. 2008). Marine Protected Areas (MPAs) are area-based conservation tools that have been successful in mitigating some of the impacts of these activities, helping to protect biodiversity, and contributing to marine ecosystem resilience (Lester et al. 2009; Green et al. 2014). Designing MPAs is challenging, not only in terms identifying and prioritizing areas important to protect, but also in terms of the social and economic considerations that are involved (Pascual et al. 2016; Agardy et al. 2016). Prioritizing individual sites for protection can be guided by systematic conservation planning (SCP): a process providing a framework for making repeatable, transparent and efficient conservation decisions (Margules and Pressey 2000; Wiersma and Sleep 2016). Key principles of SCP include comprehensiveness, adequacy, representativity and efficiency (Margules and Pressey 2000). Representativity refers to the need for MPAs to sample the full range of biodiversity and habitats in a region (Margules and Pressey 2000) and its importance is widely discussed in the field of MPA and MPA network design (Rice and Houston 2011).

Area-based marine conservation planning methods are continually improving, aiming to expand their traditional capabilities to capture representative biodiversity and habitats, as well as important ecological processes such as migration, foraging and connectivity (Burt et al. 2014; Magris et al. 2014; D'Aloia et al. 2017). Connectivity in a broad sense is the exchange of individuals, genes, energy or materials across habitat patches, communities or ecosystems (Daigle et al. 2018). Connectivity can be quantified by identifying the location of larvae sources and sinks through oceanographic modelling and pelagic larval duration (PLD) analyses (Treml et al. 2012) and also by identifying post settlement migration patterns of individuals (D'Aloia et al. 2017; Weeks et al. 2017). Incorporating connectivity processes into MPA planning, although still rarely done in practice, has been shown to enhance conservation outcomes (Martin et al. 2015; Olds et al. 2016) leading to calls for better integration between connectivity and conservation prioritization (Huntington et al. 2010; Olds et al. 2016). Connectivity processes have received increased attention in recent years in part due to Aichi Target 11 of the Convention on Biological Diversity (CBD), which states that signatory countries protect at least 10% of coastal and marine waters through ecologically representative and well-connected area-based conservation measures (CBD-UNEP 2010; Rees et al. 2018).

Seascape connectivity is one form of connectivity and can be considered analogous to landscape connectivity in the terrestrial environment (Olds et al. 2017). Seascape connectivity can refer to structural connectivity (i.e., physical linkages within a seascape as portrayed by a map of the habitat of interest), potential connectivity (i.e., a measure of connectivity that incorporates limited or assumed information on species mobility), or actual connectivity (i.e., a measure that uses spatial information on species movement to quantify connectivity; Grober-Dunsmore et al. 2009). Seascape connectivity has been quantified using various techniques, including larval flow and modelling (D'Aloia et al. 2017), acoustic tagging to identify ontogenic shifts, pathways and migratory bottlenecks (Nagelkerken et al. 2015), and also by quantifying spatial pattern metrics such as patch size, proximity and nearest neighbor (Wedding et al. 2011). Seascape connectivity influences the spatial distribution of benthic flora and fauna and supports post-settlement processes such as migration, foraging and spawning for a variety of species (Meynecke et al 2008; Grober-Dunsmore et al. 2009; Bostrom et al. 2011; Nagelkerken et al. 2015). Seascape maps are integral for quantifying connectivity by providing foundational information for assessing spatial patterns of various seascapes and habitat types. The ability to map and identify migration patterns, movement corridors and nursery and spawning grounds offers value for designing and establishing protected areas in marine and terrestrial systems (Rudnick et al. 2012; Nagelkerken et al. 2015; Magris et al. 2018). Additionally, many widely used conservation prioritization tools require spatial data layers representing conservation features – data that are provided through detailed, land/seascape scale mapping. Marxan (Ball et al. 2009) is the most widely used conservation prioritization tool for generating protected area design scenarios in both marine and terrestrial systems (Ardron et al. 2010). Marxan aims to address the problem of selecting a set of proposed areas meeting quantitative conservation targets (e.g. X% of species Y or habitat Z) at a minimal cost (e.g. forgone opportunities).

Despite evidence that considering connectivity improves conservation outcomes (Olds et al. 2016), cases where connectivity is integrated with widely used conservation prioritization tools (i.e., Marxan) are rare and dominated by studies in shallow coastal and terrestrial systems that can be mapped using optical remote sensing techniques (e.g. Crouzeilles et al. 2015; Magris et al. 2016; Weeks et al. 2017). In contrast, the vast majority of deeper seafloor regions are unmapped, particularly at scales suitable for seascape connectivity analyses. Without appropriate spatial information, it can be difficult to incorporate complex connectivity metrics into conservation prioritization tools. However, advances in acoustic seafloor mapping and sampling techniques have allowed for the production of fine-scale seascape maps that are essentially analogous to terrestrial land cover maps. Mapping the benthoscape (i.e., the component of the seascape that relates to the benthic environment; Zajac et al. 2000) provides a broad landscape perspective that yields useful information for extending connectivity analyses into the MPA designation process in these deeper benthic environments.

Benthoscape maps also provide high quality and fine-scale information on the location and extent of seafloor habitats and substrates, which can be used to assess whether MPAs and MPA networks protect representative habitats (e.g. Evans et al. 2015; Novaczek et al. 2017). However, using benthoscape maps to identify areas for protection without considering spatial configuration and connectivity may result in scenarios where representativity targets are met but the actual habitat and substrate included in each protected area does not consider the size and connectivity of the habitat patches themselves. It is well documented that the habitat within a MPA should support regular movements of targeted species (Moffitt et al 2009; Metcalfe et al. 2015), however by not considering the spatial configuration of habitat patches in MPA design, we risk decreasing MPA effectiveness by potentially limiting the protection of large, well-connected patches.

Spatial pattern metrics that measure and describe patch size, shape and configuration can be derived from landscape-scale maps and are used extensively in ecology and conservation studies in terrestrial and shallow water regions (Wedding et al. 2011; Turner and Gardner 2015). The general lack of fine-scale continuous benthoscape data precludes their use in deeper regions of the ocean. With a growing acknowledgement of the value of benthic habitat and substrate maps for informing conservation planning and monitoring (Ferrari et al. 2018; Hogg et al. 2018; Lacharité and Brown *in press*), developing methods for extending seascape connectivity analyses and spatial pattern metrics into regions that rely on acoustic techniques to map is pertinent. This is particularly relevant as many conservation objectives focus on static benthic features while little attention has been paid to connectivity processes in these environments.

Recent studies have proposed tools and methods for including connectivity in conservation prioritization (e.g. Weeks et al. 2017; Daigle et al. 2018). MarxanConnect is an example of recent tool that helps incorporate estimates of directional demographic connectivity and landscape connectivity in Marxan conservation prioritization (Daigle et al. 2018). Another recently proposed method integrates connectivity analyses and Marxan conservation prioritization in a relatively data-limited context (Weeks et al. 2017). Studies that link connectivity and conservation prioritization tend to focus on maximizing larval connectivity (White et al. 2014; Magris et al. 2016) and post settlement migration of individuals in shallow, biogenic habitats such as coral reefs, mangrove forests and eelgrass beds where habitat patch configuration and composition can be mapped using optical remote sensing techniques (Olds et all. 2016; Weeks et al. 2017). However, these tools are limited in their ability to consider benthoscape structure and configuration in the selection of potential protected areas. In general, seascape connectivity research that relies on benthoscape mapping remains largely unstudied despite the fact that processes such as migration, dispersal, reproduction and range expansion are occurring in these deeper benthic ecosystems (Comeau et al. 1998; Comeau and Savoie 2002; Hovel and Wahle 2010).

To address this gap, this research presents a new method for meeting benthic habitat and/or substrate representativity targets while simultaneously incorporating spatial pattern metrics related to patch size and connectivity into the conservation prioritization tool Marxan. We test the method through a case study in Newman Sound, a coastal region of the Canadian province of Newfoundland and Labrador. The goal is to integrate benthoscape mapping, spatial pattern analyses and Marxan conservation prioritization to develop a method where large well-connected patches of seafloor habitat within ecologically meaningful threshold distances can be preferentially selected.

3.3 Methods

3.3.1 Marxan conservation prioritization

Marxan is the most widely used conservation prioritization tool, helping select a set of potential protected areas that meet user-defined targets for conservation features (i.e., habitats, species), while attempting to minimize the displacement of, and impact on, resource users (i.e., resource harvest value, area, cultural value; Ball et al. 2009). The tool uses multiple spatial data layers representing conservation features and a single cost feature. Cost layers representing economic costs, such as fishing effort, are commonly used in Marxan prioritization to help reduce potential impacts of the proposed protected areas on resource users (e.g. Leathwick et al. 2008; Mazor et al. 2014). A common way to run a Marxan analysis in the absence of reliable socio-economic data is to assign a uniform cost value to all Marxan planning units (PUs; see Fig.1). This area based, uniform-cost approach assumes that minimizing the total area in a reserve design scenario in turn minimizes the cost to implement, monitor and manage the reserve (Ardron et al. 2010). In this context, any PU containing a habitat or species of interest would be equally likely to be selected by Marxan, regardless of its location in the study area or spatial relationship to other PUs. Marxan can favor the selection of PUs that are adjacent by using a Boundary Length Modifier (BLM). Increasing the value of the BLM in turn increases the compactness of the proposed network. While the use of BLM helps reduce the network fragmentation, it does not consider the spatial configuration of the habitat patches themselves (e.g., patch size and proximity).

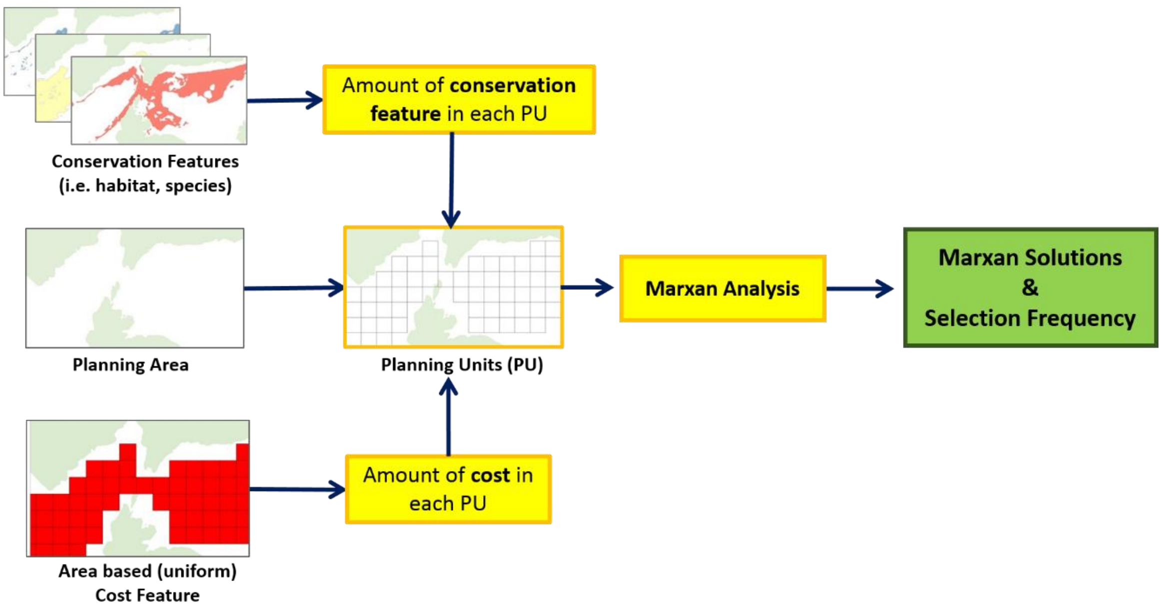

Figure 3-1: General Marxan workflow using a uniform, area-based cost layer. The selection frequency is a surface generated by overlaying the top 100 Marxan solutions where values represent the frequency at which each planning unit was selected

When area is used as a cost, meeting conservation targets in Marxan could result in solutions where representativity targets are met but the habitat or substrate within the solutions is composed of isolated fragments. Effectively, the spatial arrangement of selected PUs is not informed by knowledge of species home range movements or the spatial configuration and connectivity of habitat patches within the PUs themselves. This lack of a broad, benthoscape-scale perspective when aiming for representativity of benthic habitats in conservation planning can have important implications for protecting benthic species and ecological processes.

The method developed and tested in this study involves creating a Marxan cost layer (described below) where patches of benthic habitat are weighted based on their spatial attributes, configuration, and connectivity. The approach can support planners in achieving representative conservation while considering the spatial configuration and connectivity of seafloor habitat and substrate patches as well as species movement and habitat use. This method includes measures of structural connectivity (physical linkages between patches) and species-specific potential connectivity between habitat and/or substrate patches. This method requires continuous spatial data on the seafloor, which is becoming increasingly available due to advances in seafloor mapping technologies.

3.3.2 Marxan conservation prioritization using benthoscape cost

In this study, we propose a new method that helps capture benthoscape connectivity measures as a cost layer in the Marxan process. Benthoscape classes are acoustically distinguishable biophysical seafloor classes derived using the benthoscape mapping approach (Brown et al. 2012). The method presented here requires maps of each benthoscape class to represent conservation features. Patches belonging to each class are extracted using a Geographic Information System (GIS) approach. For benthoscape classes where species-specific connectivity and movement data are available, connectivity refers to both structural and species-specific potential connectivity. Benthoscape cost is calculated using a combination of patch area (*Pa*) measurements and neighbourhood analyses, typical metrics used in landscape ecology. Metrics that quantify patch size and proximity are valuable in MPA planning scenarios where a primary goal is to protect areas that encompass large, high quality patches of habitat within the home range or dispersal distance for focal species (Olds et al. 2016). Neighbourhood analyses include calculating the number of patches (*#P)* and mean distance between patches $\overline{(x\ dP)}$ within a threshold distance. Threshold distance can be set for each habitat layer based on existing species-specific movement data (i.e., home range, dispersal distance). Benthoscape cost for each patch when species-benthoscape associations and connectivity/movement data are available was calculated as:

$$Benthoscape\ cost = (\log(Pa) * x) + (\#P * y) + (\bar{x}\ dP * z),$$

where *x*, *y* and *z* are weights applied to each patch metric based on their importance for the species of interest. These weights should be carefully considered and informed by research and expert opinions. *Pa* values are then rescaled to values between 0-1 and inverted to allocate lower costs to larger patches (i.e., as larger patches are more desirable for selection as part of the network because they can better support the movements of target species). Values for both #*P* and $\bar{x}\ dP$ are rescaled as benthoscape costs between 0-1. Values for #*P* are inverted so that patches with more neighbouring patches within the threshold distance are assigned a lower cost. Values for $\bar{x}\ dP$ are not inverted, resulting in patches with closer neighbouring patches within the threshold distance having lower costs.

For benthoscape classes where species-specific connectivity and movement data are unavailable, connectivity refers to structural connectivity only and is calculated as:

$$Benthoscape\ cost = (\log(Pa) * x),$$

where *Pa* is patch area. *Pa* values are also rescaled as a benthoscape cost value between 0-1 and inverted. The inversion results in larger patches having lower benthoscape costs. Cost layers associated with each benthoscape class, whether they capture species-specific potential connectivity or structural connectivity, are then merged to produce a continuous benthoscape cost layer. This layer is then used for Marxan conservation prioritization. Fig. 3-2 illustrates the modified Marxan workflow when benthoscape cost replaces an area-based, uniform cost layer.

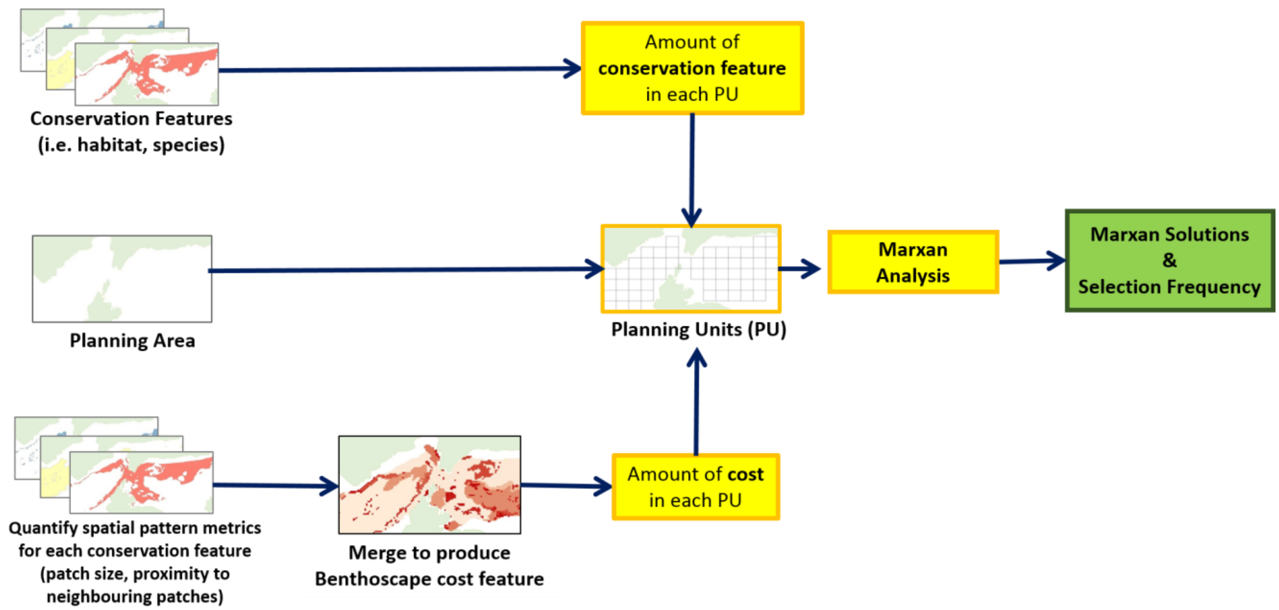

Figure 3-2: General Marxan workflow using a benthoscape cost layer. The selection frequency is a surface generated by overlaying the top 100 Marxan solutions where values represent the frequency at which each planning unit was selected

By replacing an area-based, uniform cost layer with our derived benthoscape cost, we can achieve representation targets for conservation features while minimizing benthoscape cost (i.e., maximizing seascape connectivity). Effectively, the method allows the Marxan procedure to distinguish between patches of the same benthoscape class, and preferentially select PUs that are within large benthoscape patches in close proximity to neighbouring patches over small isolated fragments. This in turn reduces the degree of fragmentation of conservation features in Marxan outputs while also prioritizing habitat or substrate patches within ecologically meaningful threshold distances. While this approach may help reduce network fragmentation, a role often played by Marxan's boundary length modifier (BLM), applying a benthoscape cost means that the habitat or substrate within the solutions themselves is composed of larger, well-connected patches.

In doing so, we effectively provide a benthoscape-scale perspective when aiming for representative benthic conservation which can have important implications for protecting species and ecological processes.

3.3.3 Application of the approach: The Case Study of Newman Sound, Newfoundland and Labrador

3.3.3.1 Study Area

The proposed approach has been tested for the area of Newman Sound. Newman Sound is a coastal fjord located in Bonavista Bay on the East coast of the island of Newfoundland, Canada (Fig. 3-3). This site has been selected due to its ecological significance in the region, the large amount of existing biophysical and ecological data, and the diversity of benthoscape classes. Additionally, no socioeconomic cost layer (i.e., map of fisheries catches) was available for Newman Sound, making the application of the benthoscape cost approach appropriate.

The maximum depth of the outer basin of the fjord is 349m, while the inner basin has a maximum depth of 63m. Several rivers and streams flow into the inner basin of Newman Sound where mudflats and estuarine vegetation are common. The adjacent Terra Nova National Park protects 400km^2 of sheltered inlets, islands, forest and bog habitat (Parks Canada, 2018).

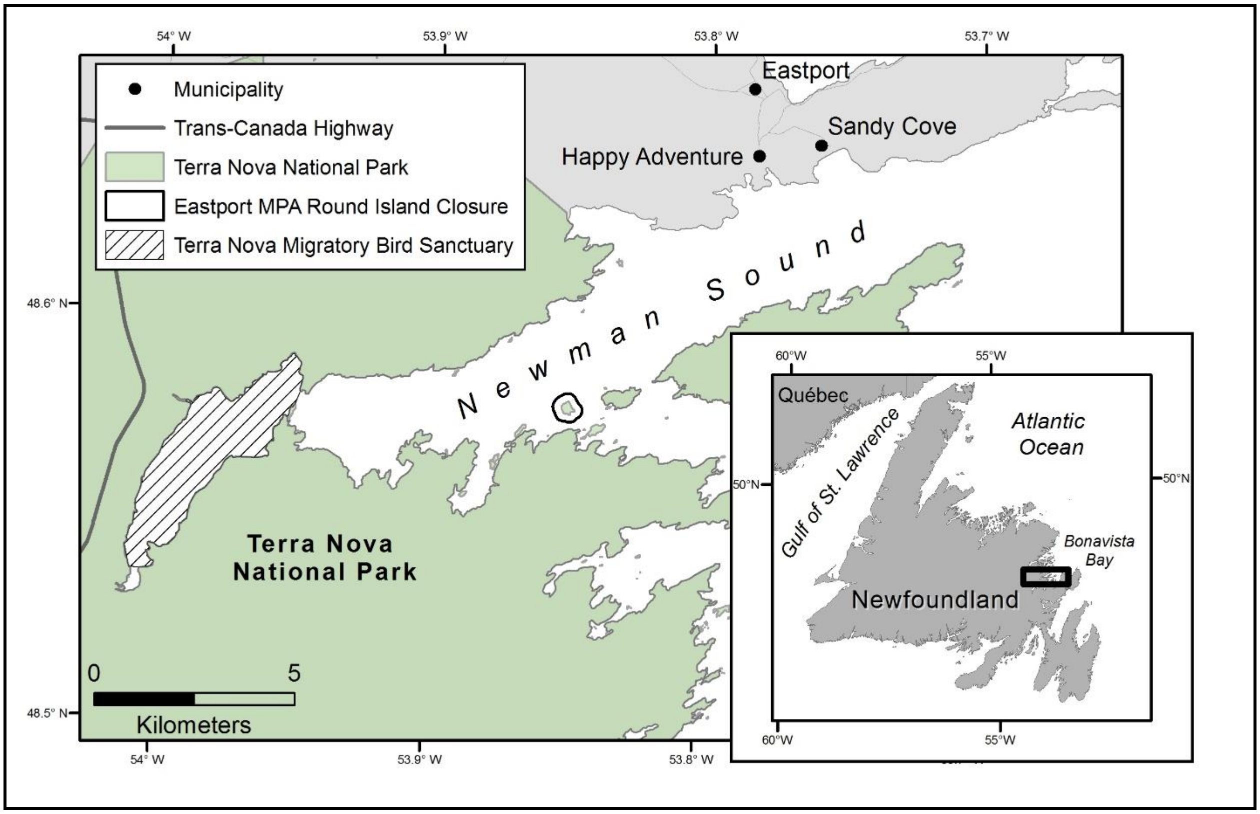

Figure 3-3: Case study area: Newman Sound, Newfoundland and Labrador

Due to the high diversity and abundance of ecologically unique habitats in the area, including tidal flats, eelgrass and rhodolith beds, as well as species-rich submerged fjord walls, Newman Sound has been listed by a NL non-for profit environmental organization as a special marine area (CPAWS-NL 2017). Newman Sound also supports sizeable eelgrass (*Zostera marina*) beds, which serve as important refuge and nursery areas for juvenile Atlantic cod (*Gadus morhua*) (Linehan et al. 2001; Cote et al. 2004; Rao et al. 2014) and other fish and invertebrate species (Joseph et al. 2006; Cote et al. 2013).

Newman Sound also includes part of one of the two Fisheries and Oceans Canada (DFO) Oceans Act MPAs in Newfoundland and Labrador: the Eastport MPA. This MPA is composed of two closures, with the Round Island closure being located in the southern part of Newman Sound (Fig. 3-3). Eastport is a small (2.1 km^2) no-take MPA established in 2005, based on a voluntary fishing closure initiated by the local community in 1997. The primary goal of the Eastport MPA is to protect American lobster (*Homarus americanus*, hereafter lobster), a species fished commercially in the region. Reports of high lobster catches informed MPA boundary placement, although no habitat mapping or biodiversity surveys were done prior to establishment. Recent habitat mapping and characterization within the boundaries of the MPA have determined that the MPA has a limited capacity to conserve habitats and biodiversity representative of the broader region (Novaczek et al. 2017). While the Eastport MPA was primarily designed to help protect a single species, its small size offers limited protection of ecologically diverse and unique areas in Newman Sound.

The seafloor landscape (i.e., benthoscape) in Newman Sound was characterized and mapped using the benthoscape approach of Brown et al. (2012), utilizing multibeam echosounder (MBES) and seafloor video data (Proudfoot et al. submitted; Chapter 2). The seafloor mapping study identified seven distinct benthoscape classes in Newman Sound: mixed boulder, mud, bedrock, deep pebble cobble, gravelly muddy sand (GMS), shallow pebble cobble/GMS and sand.

To test our method, we selected two of the benthoscape classes identified in Newman Sound as Marxan conservation features: 1) mixed boulder; 2) gravelly muddy sand (GMS). The mixed boulder benthoscape class was selected because it is comprised of complex boulder features with high macroalgal cover – characteristics commonly associated with juvenile and adult lobster habitat (Novaczek et al. 2017). In addition to this species-benthoscape class association, species-specific home range movement data are also available. Lobster tagging studies done in the region demonstrate limited lobster movement (Rowe et al. 2001). Rowe et al. (2001) determined that 58.7% of tagged lobsters were recaptured in close proximity to where they were captured and tagged, and of the more mobile lobsters, 77% were recaptured within 1km of their original tagging location. Although lobsters are known to travel distances in other regions of Atlantic Canada (Campbell and Stasko 1986; Comeau and Savoie 2002), based on Rowe et al. (2001) we used 1km as the threshold distance for lobster movement and for weighting patches of mixed boulder.

The GMS benthoscape class was selected based on previous studies examining benthic fauna in Newman Sound have shown that the GMS benthoscape class supports a distinct community of ophiuroids, infaunal bivalves and polychaete worms (Copeland 2006). GMS patches are prioritized based on size to demonstrate the benthoscape cost scenario where species-specific movement data are unavailable.

We contrast outputs that use the uniform cost approach (Fig. 3-1) against outputs that use the benthoscape cost approach (Fig. 3-2) to demonstrate how large, well-connected patches of these two benthoscape classes within ecologically meaningful distances can be preferentially selected in Marxan prioritization. We also compare the calibrated BLM scenario with a scenario that excludes the use of the BLM parameter (BLM=0). We examine "Best Solution" outputs (the solution that meets conservation targets most efficiently) for both scenarios. This allows us to contrast results of using the benthoscape cost approach to select planning units located in large, well-connected patches and increasing the BLM parameter to enhance the clustering and compactness of the PUs.

3.3.3.2 Creation of the benthoscape cost layer in Newman Sound

For each patch of mixed boulder in Newman Sound where species specific home-range data are available (e.g., American lobster; 1km):

$$Benthoscape\ cost = (\log(Pa) * 0.7) + (\#P * 0.15) + \ (\bar{x}\ dP * 0.15),$$

where *Pa* is patch area, *#P* is number of patches within the 1km threshold distance, and $\bar{x}\ dP$ is the mean distance between patches within the 1km threshold distance. ArcGIS 10.5 was used to derive patch size and proximity metrics. In the case of lobster in Newman Sound, *Pa* of mixed boulder habitat was weighted more heavily than *#P* and $\bar{x}\ dP$ due to the relatively sedentary nature and small home range of lobsters in the region (Rowe et al. 2001) and the assumption that larger patches of mixed boulder support a higher density of lobsters by increasing available habitat for recruitment and shelter (Wahle and Steneck 1991). For this case study, tests were conducted to determine an appropriate weighting scheme that ensured that across all patches, variation in patch size, the number of neighbouring patches and the distance between neighbouring patches together corresponded to ecologically meaningful benthoscape cost values. Assigning a weight of 0.7 to the patch size parameter ensures that small patches in regions with a high quantity of near neighbours (i.e., highly fragmented regions) are not assigned lower benthoscape cost values than large patches due to the quantity of neighboring patches. The remaining weight was divided equally among the parameters associated with neighboring patches. In cases where this approach may be used to support on the ground conservation decision-making, these weights should be carefully considered and informed by expert opinions. Furthermore, these weights can vary depending on study area and species that are the focus of the conservation effort. Through this case study, our intent is to demonstrate the flexibility of our method for including and weighting species-specific connectivity and movement data when available.

Pa values were rescaled to values between 0-1 and inverted. #*P* and $\bar{x}\ dP$ values were rescaled as between 0-1. #*P* values were inverted and values for $\bar{x}\ dP$ were not inverted. The equation results in benthoscape cost values for patches of mixed boulder ranging from low (0; large patches in close proximity to neighbouring patches) to high (1; small isolated patches).

For each patch of GMS in Newman Sound where information on associated species and movement is unavailable:

$$Benthoscape\ cost = (\log(Pa) * 1),$$

where *Pa* is patch area. *Pa* values were rescaled to values between 0-1 and inverted resulting in larger patches having lower benthoscape cost values.

The above calculations result in benthoscape class-specific cost layers. Cost layers for all conservation features (i.e., benthoscape classes) were then merged to produce a seamless benthoscape cost layer for Newman Sound that was then used in the Marxan conservation prioritization. Fig. 3-4 illustrates the general workflow described above for assigning benthoscape cost, using mixed boulder and gravelly muddy sand (GMS) as examples to demonstrate the method's flexibility in incorporating movement data where available.

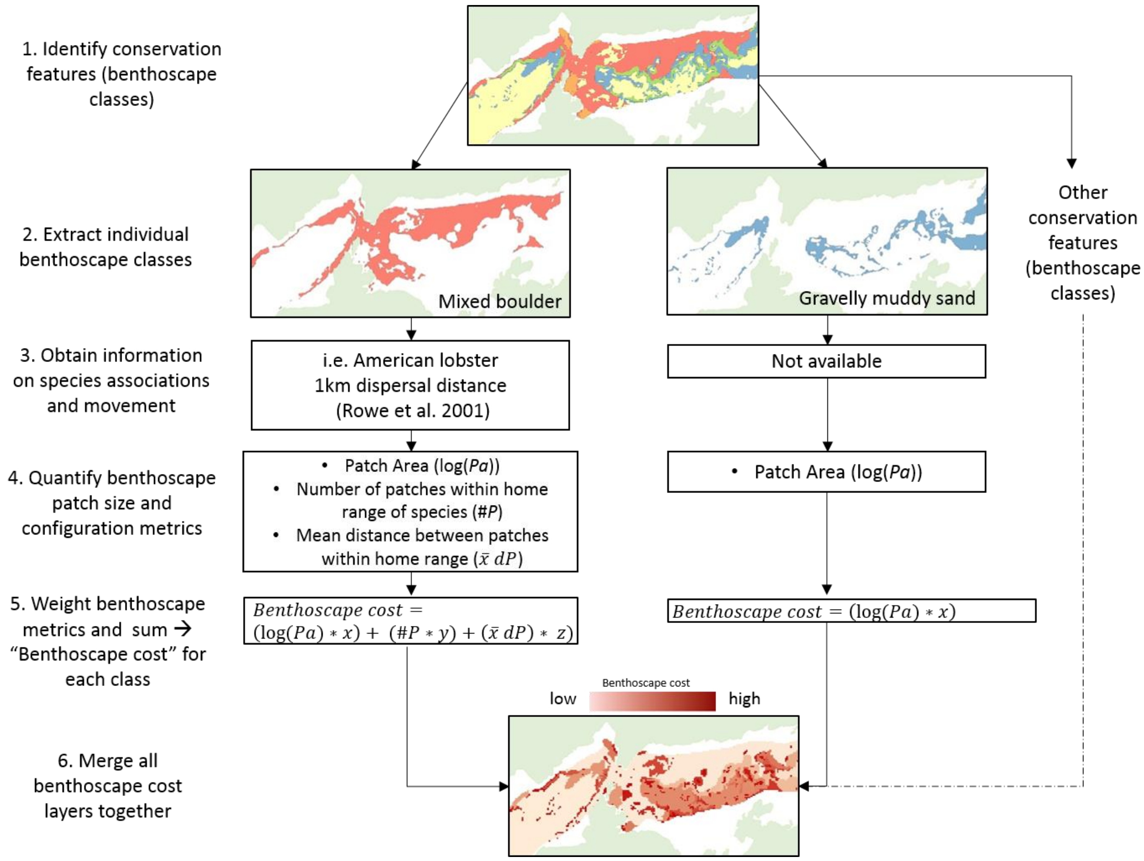

Figure 3-4: General workflow for deriving benthoscape cost

3.3.3.3 Marxan Application

Through this method we compare two Marxan spatial prioritization scenarios in Newman Sound, NL: 1) the reference "uniform cost" scenario where PUs (20 x 20m in size) are assigned an equal cost and 2) the "benthoscape cost" scenario where we use our derived benthoscape cost layer to assign a cost to each PU. Representation targets were set to include 10% of all benthoscape classes within protected areas in line with the Aichi Target 11, where signatory countries, including Canada aim to protect 10% of coastal and marine areas by 2020 through ecologically representative and well connected area-based conservation measures (CBD-UNEP 2010). While this target was set to test the approach, a higher target may be required in an actual conservation planning process due to the high ecological value of the region. The boundary length modifier (BLM), a Marxan parameter that can be modified to produce more clustered solutions remained constant (0.1) in both scenarios. The BLM value was determined through a calibration process that identified the highest BLM value (highest degree of clustering) with a minimal impact on the overall cost (Ardron et al. 2010). All other Marxan parameters were consistent across scenarios (1,000,000 iterations; 100 repetitions).

3.4 Results

3.4.1 Creation of the benthoscape cost layer

The Newman Sound benthoscape and geomorphology are typical of a submerged fjord environment, characterized by flat deep basins defined by shallow sills and steep fjord walls (Proudfoot et al. submitted). Benthoscape cost patterns reflect this structure, with low benthoscape cost values in large continuous patches and variable benthoscape cost values reflecting the more diverse patch sizes in the shallower regions of the fjord (Fig. 3-5).

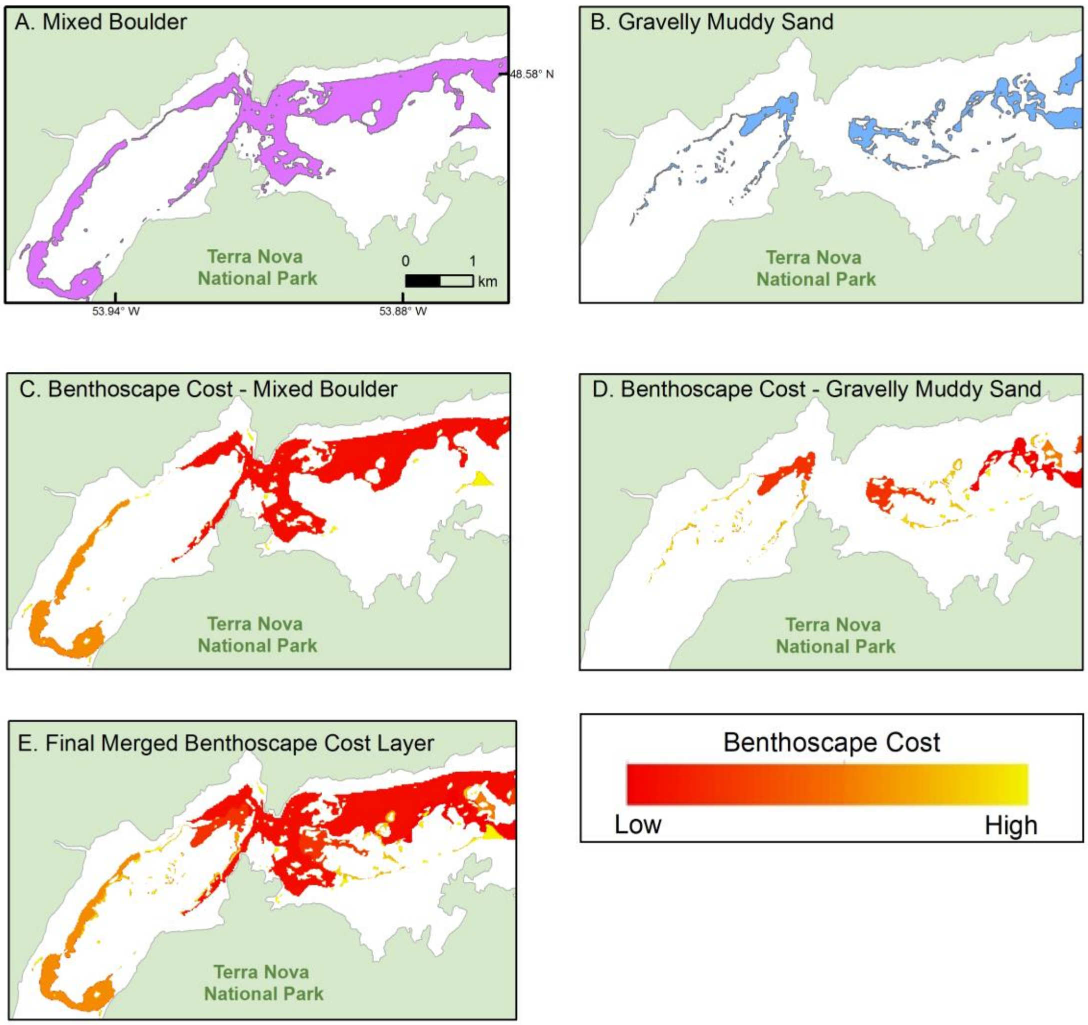

Figure 3-5:Extent of mixed boulder (A) and GMS (B) benthoscape classes in the inner region of Newman Sound adapted from Proudfoot et al. (submitted). Benthoscape cost layers for mixed boulder (C) and GMS (D) benthoscape classes show cost values ranging from 0-1 within each class. Final benthoscape cost layer after C and D are merged (E)

Benthoscape cost patterns for patches within each benthoscape class are spatially variable. As expected from the method used, high benthoscape cost values tend to be associated with highly fragmented regions of the benthoscape (see Fig. 3-5D for examples).

Large patches of mixed boulder contribute to the low benthoscape cost values in the outer region of the fjord (Fig. 3-5C). Low benthoscape cost values are also assigned to patches of mixed boulder that are in close proximity to neighbouring patches within the threshold distance of 1km lobster home range. These regions where benthoscape cost is low are preferentially selected in Marxan prioritization.

3.4.2 Impact of using benthoscape cost vs. uniform cost on Marxan conservation prioritization

Clear differences can be observed in Marxan outputs when comparing PU selection frequencies from the uniform cost (Fig. 3-6A, Fig. 3-6B) with the benthoscape cost scenarios (Fig. 3-6C, Fig. 3-6D). By comparing differences in selection frequencies in the two scenarios (Fig. 3-6E, Fig. 3-6F), we can identify areas that are selected more or less frequently in the benthoscape cost scenario (red areas were selected more frequently; blue areas were selected less frequently). PUs that are within small, isolated patches were selected less frequently, while PUs within large structurally connected patches were selected more frequently.

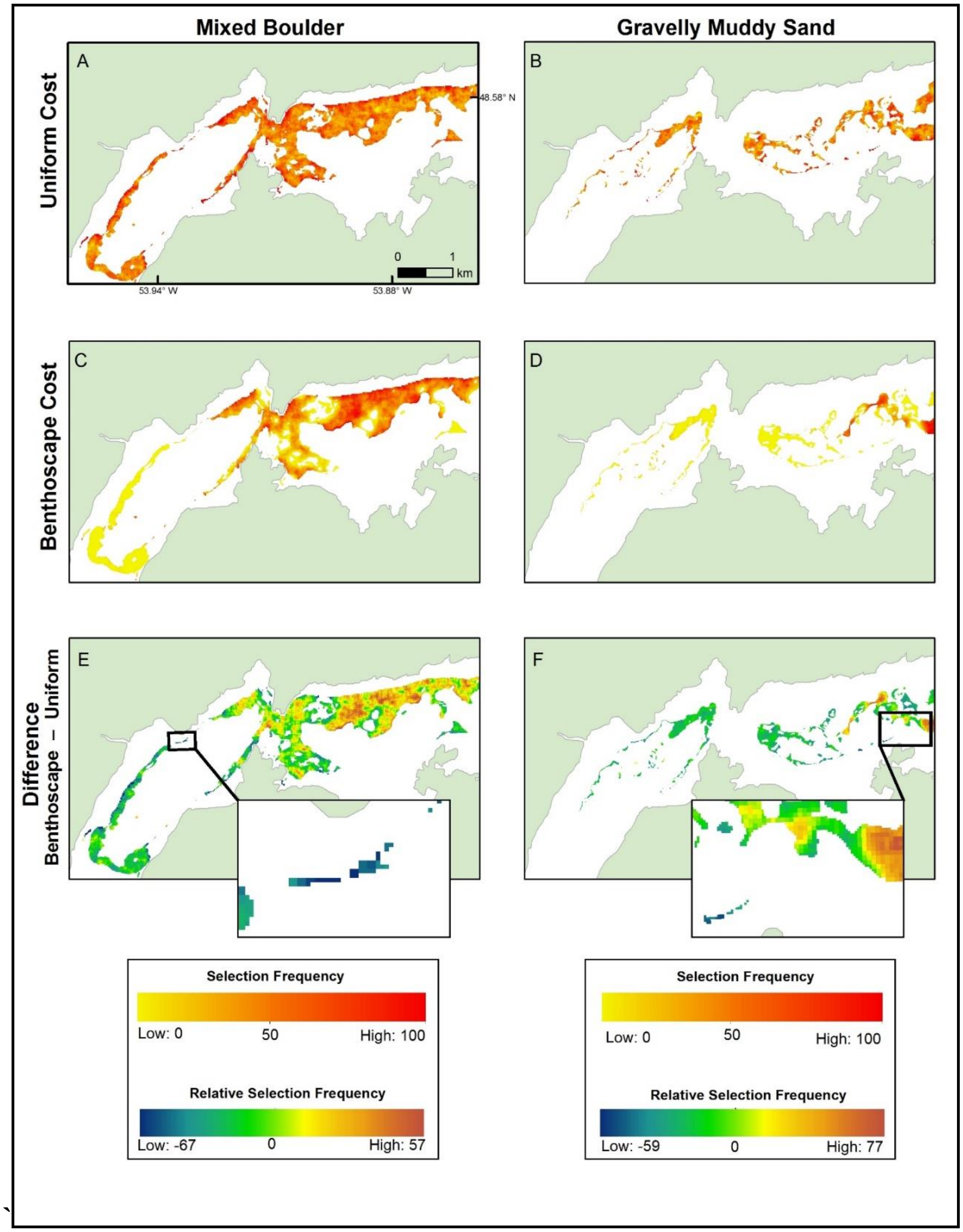

Figure 3-6: Marxan outputs comparing selection frequencies of planning units under two cost scenarios. Uniform cost: mixed boulder (A) and gravelly muddy sand (B). Benthoscape cost: mixed boulder (C) and gravelly muddy sand (C). Planning units selected more or less frequently when benthoscape cost is considered: mixed boulder (E) and gravelly muddy sand (F)

By examining benthoscape-class specific selection frequencies, it is clear that large, structurally connected patches of GMS are preferentially selected in Marxan prioritization while small, patches are avoided (Fig. 3-6D). In the case of mixed boulder, large patches within the 1km threshold distance of lobster movement are preferentially selected in Marxan prioritization, while small isolated fragments are avoided (Fig. 3-6C). Differences in the total area of mixed boulder and GMS included in the 'best solution' (Fig. 3-7A & B) between the uniform and benthoscape cost scenarios were minimal. The total area of mixed boulder in the best solution increased slightly (+0.03%) by replacing the uniform cost layer with the benthoscape cost (0.324km^2 in the benthoscape cost scenario vs. 0.323km^2 in the uniform cost scenario). The total area of GMS in the best solution decreased slightly (-0.05%) in benthoscape cost scenario (0.174 km^2 in the benthoscape cost scenario vs. 0.175km^2 in the uniform cost scenario).

Figure 3-7 illustrates the difference between using the BLM parameter to increase the compactness of the PUs (Fig. 3-7A & B) and using no BLM (Fig. 3-7C & D) in both the benthoscape cost and uniform cost scenarios. In the benthoscape cost scenarios (Fig. 3-7A & C), it is clear that PUs are concentrated in benthoscape patches that are larger and less fragmented. In contrast, applying a uniform cost in both BLM scenarios resulted in PUs that are essentially randomly distributed throughout the planning area at two levels of compactness (Fig. 3-7B & D). By using the benthoscape cost approach and simultaneously increasing the BLM, we can not only increase the compaction of the individual reserves within the network (increased BLM), but also ensure that the individual reserves encompass large, well-connected patches of habitat.

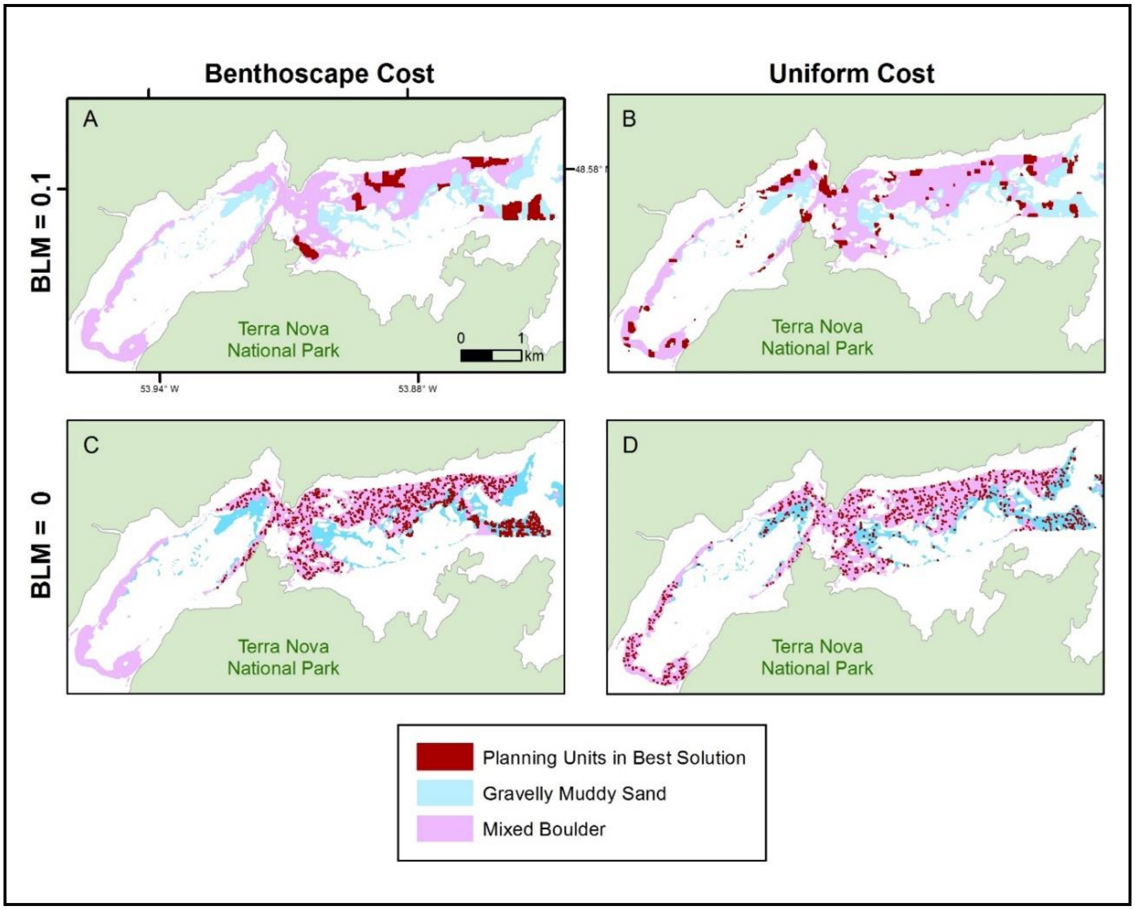

Figure 3-7: Marxan Best Solution outputs under two BLM and two cost scenarios. (A) Benthoscape cost and BLM=0.1. (B) Uniform cost and BLM=0.1. (C) Benthoscape cost and BLM=0. (D) Uniform cost and BLM=0

3.5 Discussion

Using Newman Sound as case study, this research demonstrates a method for using a benthoscape mapping approach (Brown et al. 2012) as a foundation for quantifying seascape connectivity and spatial pattern metrics (e.g. patch size, proximity) in marine environments and incorporating this information into Marxan prioritization. The method is applicable in any area where fine-scale seafloor habitat data are available. This method results in the selection of larger and better connected patches of seafloor habitat and substrate being preferentially selected and also allows for a high degree of flexibility when species specific movement and connectivity data is available.

3.5.1 Benthoscape cost vs. uniform cost: impact and implications

When an area-based uniform cost layer was used to prioritize 10% of mixed boulder and GMS benthoscape classes in Newman Sound, all PUs were assigned an equal/uniform cost. This resulted in essentially a random selection of PUs (Fig. 3-6A & B; Fig. 3-7B & D). Effectively, a PU that contains a single, isolated fragment of habitat was considered equal in conservation priority to a PU that was within a large, well-connected patch of habitat. In contrast, by applying a benthoscape cost we were able to identify and prioritize the selection of large, well-connected patches for protection and exclude small isolated habitat fragments.

Large habitat patches typically support more species and provide more within-patch heterogeneity (McArthur and Wilson 1963; Turner and Gardner 2015). This is an important consideration in MPA network planning where the key goal is ensuring protection of areas encompassing large, high quality patches of habitat within the home range or dispersal distance for focal species (Olds et al. 2016). Applying a benthoscape cost approach resulted in a preferential selection of PUs located in larger patches (e.g. Fig. 3-6C, Fig. 3-6D) that are within the dispersal distance of a focal species (e.g. Fig. 3-

6C). This can allow planners to have more confidence that the habitat and substrate within selected PUs is comprised of patches that support more species, individuals and ecological processes. Furthermore, prioritizing PUs located in larger benthic habitat and substrate patches increases the likelihood that adjacent spillover habitat outside reserve boundaries is available, which could potentially enhance fisheries that target mobile benthic invertebrates and demersal fish if enhancing fisheries is a goal of the conservation initiative (e.g. Freeman et al. 2009).

Using a benthoscape cost layer as opposed to an area-based cost layer effectively imposes constraints on the Marxan selection process. When constraints are applied to conservation prioritization, more PUs are often required to meet conservation targets, which can increase the total ‘cost’ of the reserve network and have a greater impact on the displacement of human interests (Ardron et al. 2010). However in the case of Newman Sound, differences in total area of mixed boulder and GMS included in the ‘best solution’ between the uniform and benthoscape cost scenarios were very small. This suggests that trade-offs such as increasing the size of the overall reserve network may be negligible when a benthoscape cost layer replaces an area-based cost layer. However, this may not be the case in more complex scenarios that involve large numbers of conservation features within a larger geographic planning area. As such, additional testing is required to determine how Marxan outputs would vary as the complexity of the planning scenario increases. Our results also demonstrate that applying a benthoscape cost approach to prioritize large, well-connected patches is not analogous to increasing Marxan's BLM parameter to produce more clustered solutions.

3.5.2 Conservation planning in Newman Sound – limitations

This study intended to illustrate how fine-scale seafloor habitat and substrate configuration can be incorporated into protected area planning. The application of the method to Newman Sound aimed to provide a proof of concept, but is not intended to provide direct advice on conservation planning in this region, which would require a more thorough planning exercise. Newman Sound was a suitable case study for developing and testing our method due to the availability of a fine scale benthoscape map (Proudfoot et al. submitted; Chapter 2). An additional limitation is that the coverage of the benthoscape map is limited to the area that was acoustically mapped.

In Newman Sound, several habitats with high conservation potential were not included due to difficulties in accessing nearshore and intertidal environments as well as discriminating them acoustically. The inner basin and other regions of Newman Sound outside the extent of the benthoscape map support sizeable eelgrass (*Zostera marina*) beds, which serve as important refuge and nursery areas for juvenile Atlantic cod (*Gadus morhua*) (Linehan et al. 2001; Cote et al. 2004; Rao et al. 2014) and other fish and invertebrate species (Cote et al. 2013; Joseph et al. 2013). The location and extent of eelgrass beds throughout Newman Sound have not yet been mapped, and therefore eelgrass beds could not be included as a conservation feature with an associated benthoscape cost.

Previous research has identified the presence of a rhodolith bed located at the shallow narrow region separating the inner and outer basins of Newman Sound (Copeland 2006). Rhodolith beds form complex habitats that support a high diversity of invertebrates, fish and algae (Steller et al. 2003; Copeland et al. 2008; Hernandez-Kantun et al. 2017 and references therein) and also provide refuge and feeding grounds for juvenile Atlantic cod (Kamenos et al 2004). Rhodolith beds are sensitive to bottom contact fishing activities (Hall-Spencer and Moore 2000; Kamenos et al. 2003), waste build-up from aquaculture operations (Hall-Spencer et al. 2006) and are particularly vulnerable to impacts from ocean acidification (Martin and Hall-Spencer 2017 and references therein).

Rhodolith beds are also not included as a conservation feature in this study because rhodolith could not be acoustically distinguished from the mixed boulder benthoscape class (Proudfoot et al. submitted; Chapter 2). However, the ecologically unique and sensitive nature of rhodolith and eelgrass beds suggests that they should be included in any future conservation activities in the region.

While the method presented here was developed and tested in the context of Newman Sound, it is in theory applicable to any site where fine-scale seafloor spatial data is available (i.e., habitat, substrate, benthoscape). However, as in all conservation prioritization scenarios, care must be taken to ensure that conservation features and their associated targets are ecologically meaningful, and their subsequent protection will offer the highest benefits to species and ecosystems (Tear et al. 2005; Agardy et al. 2016;

Wiersma and Sleep 2018). Furthermore, more studies examining direct relationships between benthoscape structure and behavior and movement for specific species could better inform benthoscape cost allocation. For example, in parts of the Northeast Atlantic, heterogeneous regions of the seafloor have been shown to be associated with greater abundances of Atlantic cod (Elliot et al. 2017). In this case, assigning a low benthoscape cost to patches in heterogeneous regions of the benthoscape might better protect Atlantic cod habitat in Marxan planning scenarios. Another example of the flexibility of the benthoscape cost approach depending on the species or habitat of interest is to use concepts related to core area and edge effects to assign a lower benthoscape cost to the interior region or core area of patches (Turner and Gardner 2015). These metrics can be important factors that influence habitat quality, or the distribution, abundance of a species of interest (Pittman and McAlpine 2004) and can be preferentially selected in conservation planning scenarios by applying the benthoscape cost approach.

3.5.3 Future directions linking seafloor mapping and connectivity in conservation prioritization

Marxan is a widely used decision support tool that is used to generate protected area network design scenarios in both marine and terrestrial systems (Ardron et al. 2010). Despite the fact that connectivity is known to positively influence conservation outcomes (Olds et al. 2016), using high quality, fine scale seafloor maps in Marxan analyses to apply connectivity metrics is rare. This research illustrates how both fine-scale benthic mapping and connectivity analyses can be integrated into Marxan analyses using relatively simple landscape ecology metrics and methods.

MarxanConnect, a recently developed application for operationalizing connectivity in conservation prioritization allows users to incorporate estimates of directional demographic and landscape connectivity in conservation planning (Daigle et al. 2018). High quality seafloor maps of habitats and ecosystems mapped using acoustic techniques can be applied to MarxanConnect's landscape connectivity function. The function uses a resistance connectivity matrix (Zeller et al. 2012) that requires information on the degree to which different habitat and substrate types facilitate or impede movement. In Newman Sound, detailed information on species-habitat associations and species-specific movement across substrate types was unavailable and as such, the use of the landscape functionality in MarxanConnect would require uninformed assumptions about how species move across various substrate and benthoscape types. However, future studies focused on explicitly understanding how benthoscape structure influences populations, communities and species movement in Newman Sound could then support this functionality.

Spatial and temporal oceanographic information is currently unavailable for Newman Sound. If this data were to become available, future studies could identify larval connectivity patterns within the region. This information could be used to calculate benthoscape cost values that consider larval connectivity in addition to post settlement-adult movement. Methods for incorporating both adult movement and larval connectivity in MPA planning have been demonstrated (i.e., D'Aloia et al. 2017). However adult home range movement often informs minimum MPA size, as opposed to influencing the selection and inclusion of individual habitat patches within the dispersal distance of a species of interest (Moffitt et al. 2009; Metcalfe et al. 2015; D'Aloia et al. 2017). Furthermore, downscaling oceanographic data to match the resolution of fine-scale benthoscape maps could aid in delineating benthoscape classes that are defined by both benthic and oceanographic variables. This could provide additional information that could be used in conservation prioritization, namely how larval connectivity models relate to benthic habitat patch size and configuration.

3.5.4 Benthoscape cost approach - limitations

The method demonstrated in this research provides a starting point for considering benthic landscape (benthoscape) composition and configuration when species specific movement and resistance to movement data is limited. Additionally, the method also allows for flexibility as data specific to the relationships between benthoscape pattern and ecological processes (e.g. migration, foraging, connectivity) become available.

In conservation planning, minimizing conflict with resource users is key (Douvere 2008). Although utilizing a benthoscape cost layer precludes the use of a true socioeconomic cost layer, further analyses and testing to explore how a benthoscape cost and socioeconomic cost layer could be combined could potentially address this issue. This is an important future research direction as the benthoscape cost layer, while ecologically meaningful, cannot be used for decision making without an additional step that considers displacement of human interests in the form of a socioeconomic cost layer. One approach could be to explore how socioeconomic cost layers could be weighted based on benthoscape cost values so that benthoscape cost is integrated with the socioeconomic cost layer. However, extensive testing and sensitivity analyses would be required to determine how the combined cost layers influence site selection.

Nonetheless, a benthoscape cost layer could potentially substitute an area-based cost layer in scenarios where socioeconomic cost layers are unavailable.

An additional limitation of the approach presented here is the use of a benthoscape classification scheme to determine benthoscape cost and identify conservation features. The limitation stems from the fact that the relationship between the habitat of a particular species and a benthoscape class is rarely 1:1 (Zajac et al. 2003; Brown et al. 2012). For example, mobile species may utilize soft sediment regions and rocky reefs during different life stages (e.g., Ortiz and Tissot 2008) or times of day (e.g., Mason and Lowe 2010). Thus, basing benthoscape cost and conservation features on a classification scheme could present an oversimplification of how species interact with the benthoscape. Alternatively, a species distribution modelling approach (SDM; Wilson et al. 2007) would provide more detail relating to species habitat preferences across a benthoscape and could potentially provide a more detailed representation of benthoscape cost based on the importance of a particular region for a species of interest.

3.5.5 Applications in terrestrial contexts

The benthoscape cost approach demonstrated in this study is not limited to benthic landscapes. It is potentially adaptable to any region, including terrestrial and shallow water environments, where high quality spatial data (i.e., habitat, land cover) and information on species movement and habitat use is available and accessible. As in marine contexts, supporting connectivity and movement processes for species is paramount in terrestrial conservation planning (Correa Ayram et al. 2016), and the application of the benthoscape cost approach in terrestrial contexts would effectively be seamless. Incorporating connectivity processes in terrestrial conservation planning typically involves identifying habitat corridors and migratory movement pathways which can be measured directly, or inferred from least cost path and least cost corridor analyses (Correa Ayram et al. 2016). However, in conservation planning scenarios where this information is unavailable, landscape maps could potentially be used in conjunction with the benthoscape cost approach, and thus discussed as a “landscape cost,” to preferentially select large, well-connected patches within the home range or dispersal distance of a species of interest. The transferability and flexibility of the benthoscape cost approach across marine and terrestrial systems offers a potential positive contribution to terrestrial conservation planning, particularly in scenarios where species-specific movement data may be limited.

3.6 Conclusions

Using a fine-scale seafloor map of a coastal fjord with high habitat and ecological diversity in Eastern Canada (Proudfoot et al. submitted; Chapter 2), this paper proposed and tested a new method for integrating benthoscape mapping, seascape connectivity and spatial pattern metrics into conservation prioritization. The vast majority of studies that link seascape connectivity and conservation prioritization have been done in shallow biogenic environments such as coral reefs, mangrove forests and eelgrass beds (Wedding et al. 2011; Olds et al. 2016). Advances in acoustic mapping techniques can support greater integration between landscape ecology and conservation prioritization (Olds et al. 2016) by providing foundational information for assessing spatial arrangement of seafloor habitats and substrates.

By creating and applying an alternative Marxan cost layer where PUs were weighted to prioritize large habitat patches that are within the spatial extent of species dispersal abilities we favor the selection of solutions that better support species and ecological processes. In doing so, conservation planners can incorporate spatial pattern metrics and broader scale seafloor patterns and complexity into conservation prioritization. A broader landscape perspective is known to be important when designing and evaluating MPAs and MPA networks (Young et al. 2018) but has yet to be incorporated in conservation prioritization tools. The ability to map and subsequently quantify spatial pattern metrics in benthic environments has great potential for extending the field that links seascape connectivity and conservation prioritization and better support biodiversity conservation in these environments.

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Chapter 4 – Conclusions

4.1 General summary

Spatial data are often a critical component of area-based conservation initiatives in both marine and terrestrial systems (e.g. Falucci et al. 2007; Sass et al. 2012; Haggarty and Yamanaka 2018). High-resolution, full-coverage landscape maps have been available to terrestrial conservation planners for decades (Melesse et al. 2007), and have become integral to the design, monitoring and management of terrestrial protected areas (e.g. Townsend et al. 2009; Soverel et al. 2010). In contrast, the availability of comparable maps of the seafloor is relatively limited and patchy in distribution. Despite this limited availability, cases where seafloor maps have informed MPA design, monitoring and management continue to be on the rise, demonstrating their value throughout the MPA implementation and management process (Young and Carr 2015; Novaczek et al. 2017; Lacharité and Brown *in press*).

This thesis demonstrates the utility of seafloor maps in conservation planning by answering the following three research questions: (1) What is the composition and spatial configuration of the Newman Sound benthoscape?, (2) How can landscape ecology metrics be applied to benthoscape maps to describe and measure seafloor structural and species-specific potential connectivity?, (3) How can landscape ecology metrics be used by conservation prioritization tools so that MPA design considers benthoscape composition and configuration. The first question was addressed in

Chapter 2, where fine-scale seafloor mapping strategies were used to characterize the benthoscape in Newman Sound. Chapter 2 also involved the identification of regional scale benthoscape patterns in Newman Sound, an assessment of the extent of habitat representativity within the Eastport MPA, and provided insight into how we can support the transition from single-species to biodiversity conservation in Newman Sound.

Questions 2 and 3 were addressed in Chapter 3, where patch size and connectivity analyses were used to quantify structural and potential connectivity and create a Marxan cost layer where large, well connected patches are assigned lower costs. The "benthoscape cost" approach presented in Chapter 3 allowed the Marxan procedure to preferentially select large, patches within the home-range of a given species, which can be important for reducing fragmentation in conservation solutions to better support species and ecological processes. Chapter 3 also discusses how the flexibility of the benthoscape cost approach lends itself to applications in other geographic contexts.

As area-based conservation planning continues to shift towards protecting not only species and habitats, but also ecological processes such as connectivity (Carr et al. 2017; Pittman et al. 2017), the value of seafloor spatial data to conservation planning is becoming increasingly clear. As demonstrated in this thesis, fine-scale seafloor maps provide the foundation for assessing the composition and configuration of benthic habitat patches, and for exploring how seascape connectivity and fragmentation analyses can be integrated with widely used conservation prioritization tools. Seafloor spatial data also enables the exploration of more advanced questions related to how spatial pattern metrics such as patch size, shape and proximity influence the distribution and abundance of benthic species.

4.2 Limitations and future research directions

4.2.1 Availability of seafloor spatial data

Fine-scale spatial data representing conservation features are valuable for supporting MPA and MPA network design (Ferrari et al. 2018; Hogg et al. 2018), as well as for assessing whether MPA and MPA networks meet representativity and replication targets (Young and Carr 2015). In terms of MPA and MPA network design, methods such as those presented in Chapter 3 are limited by the quality of the input data, meaning that the outputs are only useful if the input data are accurate and meaningful. As such, spatial data used in any Marxan analyses should be carefully scrutinized in terms of how they may influence outputs (Ardron et al. 2010).

Additionally, fine-scale seafloor spatial data can be used to locate and identify features associated with high biodiversity that can be recommended as conservation features in MPA management plans. These conservation features are typically areas of high biodiversity and productivity and can include coarse-scale features such as seamounts (Morato et al. 2010) and submarine canyons (De Leo et al. 2010) as well as fine scale features such as regions of high habitat heterogeneity and rugosity (Elliot et al. 2017; McArthur et al. 2010; Gratwick and Speight 2005) and biogenic habitats (Dunham et al. 2018; Francis et al. 2014). Currently, only a fraction of the seafloor is mapped at scales appropriate for benthic habitat mapping and conservation planning (GEBCO 2018). There is a recognized need for a unified global bathymetric dataset, not only for supporting marine conservation and fisheries management needs, but also for understanding ocean circulation, under-water geohazards, tsunami forecasting and myriad other social, economic and conservation needs (Mayer et al. 2018). Recently, an international project (GEBCO 2030) has been launched with the goal of mapping the global seafloor at resolutions that permit the types of geomorphometric analyses required to understand and map benthic habitats (GEBCO 2018). The GEBCO 2030 project, and the recent release of the Canadian Hydrographic Service's NONNA-100 (non-navigational 100m resolution; CHS 2018) dataset, provide useful information that supports conservation planning by providing the basis for applying more complex analyses at broader spatial scales. In Canada, as we move towards meeting our spatial conservation targets, full-coverage bathymetric data will be valuable for designing, monitoring and managing our existing and future MPAs and MPA networks. Additionally, full-coverage spatial data would provide an extensive foundation for applying and testing the method presented in this thesis at broader spatial scales and coarser resolutions.

4.2.2 Future research in Newman Sound

Newman Sound is an ecologically diverse and well-studied region in Newfoundland and Labrador. However, the Eastport MPA located in Newman Sound protects very little of this diversity (Novaczek et al. 2017) despite our knowledge of the importance of conserving representative biodiversity and unique habitats. Local community and stakeholder interest in expanding the Eastport MPA as well as an interest in increasing the protection of the diverse and ecologically unique regions in Newman Sound (CPAWS-NL 2018) demonstrates some of the timely and socially relevant aspects of this research. The boundary expansion scenarios presented in Chapter 2 can inform potential adaptive management of the Eastport MPA - something rarely done for Canadian MPAs. In Canada, changing MPA boundaries and regulations is a complex and lengthy process (Morris and Green 2014). However, the ability to adaptively manage MPAs is important, as adaptive management allows regulations and boundaries to be improved as new data and knowledge accumulates and as ecological systems change through time (Wilhere 2002). In the case of the Eastport MPA, protecting regional biodiversity was not one of the MPA objectives. Our knowledge of the importance of ecosystem based management and ecological connectivity has advanced, and could perhaps be considered as reasons to support the adaptive management of the MPA and increasing the amount of protected area coverage in Newman Sound. Enhancing habitat representativity and connectivity within existing small MPAs such as Eastport can also contribute to broader conservation goals associated with local, national and international conservation commitments.

4.3 Contribution to conservation planning in Canada

Canada has committed to protecting ecosystems, species and biodiversity through the establishment of an ecologically representative and well-connected MPA network that covers 10% of its coastal and marine areas (DFO 2009). Fisheries and Oceans Canada (DFO) is advancing MPA network establishment in 5 priority bioregions (DFO 2018a) with representativity, replication and connectivity being identified as key design features (DFO 2009). Seafloor maps can help identify habitats derived from biophysical classifications that are represented in existing MPAs and MPA networks (Copeland et al. 2013; Novaczek et al. 2017) and in design scenarios at earlier stages of implementation. Seafloor maps can also help determine whether habitats are replicated within the reserve or across the reserve network, providing a safeguard against potential disturbances (Young et al. 2017). However, quantifying the relationship between benthoscape composition and configuration and seascape connectivity is perhaps less straightforward, despite evidence of movement and migration patterns of benthic and demersal species that occupy these environments (Comeau et al. 1998; Comeau and Savoie 2002; Shank et al 2010). Seascape connectivity continues to be a primary feature of effective MPA and MPA network planning, and will continue to guide the establishment of protected areas in a variety of regions and ecosystems, including deeper benthic environments that require acoustic techniques to map and identify connectivity patterns.

In the Canadian MPA network planning process, four out of the five priority bioregions are using Marxan to guide the design and establishment of their respective MPA networks (Canada-BC MPA Network Strategy 2014; DFO 2017a; DFO 2017b; DFO 2018b). While connectivity has been identified as a key design feature (DFO 2009, CBD-UNEP 2010), it appears that connectivity will be largely assessed in post-hoc evaluations of design scenarios or considered in adaptive management rather than integrated into early stages of network design to inform MPA placement (DFO 2017a; DFO 2017b; DFO 2018b). Methods such as those presented in this thesis, as well as recent applications and methods that also advocate for integrating connectivity analyses at earlier stages of MPA network design (D'Aloia et al. 2017; Weeks et al. 2017; Daigle et al. 2018) could potentially be considered in the remaining eight bioregions where MPA networks will be established. However, the limited availability of appropriate spatial data representing conservation features and species specific movement data may hinder the ability to apply these types of connectivity analyses in bioregional MPA network planning. An additional challenge may be the large spatial extent of the bioregional planning areas. However, the benthoscape cost approach is flexible and could potentially be applied to identify and prioritize large, structurally connected conservation features that are currently mapped at coarse resolutions. The regions identified using this approach could potentially become the focus of subsequent mapping and conservation efforts. For example, coarse resolution maps of geomorphic features such as seamounts, canyons and ridges have been produced over large areas (Rubidge et al. 2016), and while these maps may not be appropriate for investigating species-specific connectivity questions, they can be useful for identifying and prioritizing areas for conservation and for determining where fine-scale mapping efforts could potentially be focused. For countries like Canada with large Exclusive Economic Zones (EEZs) and limited-fine scale mapping data, coarse resolution maps can be valuable for setting the foundation for further fine-scale mapping, connectivity and conservation prioritization analyses.

Incorporating connectivity processes into MPA and MPA network planning is complex. It is also important for designing effective and resilient MPAs and MPA networks that support species and ecosystems, particularly as intense anthropogenic impacts continue to threaten marine biodiversity and habitats (WWF 2018). For deeper benthic ecosystems, this requires close integration of seafloor acoustic mapping, the use of connectivity and spatial pattern metric analyses and conservation prioritization tools (i.e., Marxan). As advances in this field continue to provide methods and operational tools for integrating connectivity and conservation prioritization, ensuring that conservation planners and agencies responsible for MPA design and establishment are aware of and have access to these methods is paramount. This is particularly important as meeting deadlines, such as Canada’s goal of meeting the 10% protection target by 2020, means that conservation planners require flexible and adaptable methods that are useful and applicable when the availability of fine-scale spatial data and species movement information is variable.

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Appendix

Appendix 1: Imagery data used to classify benthoscape in Newman Sound. Datum: NAD 83. GMS = Gravelly Muddy Sand.

Station	Image ID	Source	Latitude (decimal degrees)	Longitude (decimal degrees)	Benthoscape Class	Depth (m)	Slope (deg)	EM1002 Backscatter (decibels)	EM3000 Backscatter (decibels)
AC01	AC01_0	2016 surveys	48.58864	-53.81604	Mud	-136.00	10.8	-12.05	n/a
AC01	AC01_15	2016 surveys	48.58871	-53.81586	Mud	-137.50	10.9	-9.85	n/a
AC01	AC01_23	2016 surveys	48.58875	-53.81577	Mud	-137.10	12.4	-11.11	n/a
AC01	AC01_42	2016 surveys	48.58887	-53.81558	Mud	-140.90	11.7	-7.96	n/a
AC01	AC01_53	2016 surveys	48.58895	-53.81549	Mud	-142.50	11.3	-11.74	n/a
AC01	AC01_63	2016 surveys	48.58902	-53.81540	Mud	-142.50	11.3	-11.74	n/a
AC02	AC02_0	2016 surveys	48.58947	-53.82683	Mud	-142.10	8.1	-13.63	n/a
AC02	AC02_40	2016 surveys	48.58961	-53.82664	Mud	-143.70	7.9	-17.72	n/a
AC02	AC02_68	2016 surveys	48.58970	-53.82650	Mud	-146.40	6.4	-19.30	n/a
AC02	AC02_95	2016 surveys	48.58981	-53.82639	Mud	-147.60	5.3	-20.24	n/a
AC03	AC03_19	2016 surveys	48.58514	-53.82705	Mud	-119.70	8.0	-17.72	n/a
AC03	AC03_26	2016 surveys	48.58521	-53.82700	Mud	-119.70	8.0	-13.94	n/a

Station	Image ID	Source	Latitude (decimal degrees)	Longitude (decimal degrees)	Benthoscape Class	Depth (m)	Slope (deg)	EM1002 Backscatter (decibels)	EM3000 Backscatter (decibels)
AC03	AC03_33	2016 surveys	48.58528	-53.82695	Mud	-120.10	8.1	-13.94	n/a
AC03	AC03_48	2016 surveys	48.58541	-53.82680	Mud	-121.80	7.2	-11.42	n/a
AC03	AC03_5	2016 surveys	48.58501	-53.82717	Mud	-117.90	11.4	-22.45	n/a
AC03	AC03_64	2016 surveys	48.58556	-53.82666	Mud	-122.50	6.4	-13.94	n/a
AC03	AC03_73	2016 surveys	48.58564	-53.82657	Mud	-123.50	5.0	-12.37	n/a
AC03	AC03_84	2016 surveys	48.58572	-53.82644	Mud	-124.50	4.0	-13.00	n/a
AC03	AC03_96	2016 surveys	48.58583	-53.82632	Mud	-125.50	2.2	-16.46	n/a
AC05	AC05_21	2016 surveys	48.58192	-53.85234	Shallow Pebble/Cobble	-71.50	11.9	-12.05	n/a
AC05	AC05_44	2016 surveys	48.58180	-53.85273	Shallow Pebble/Cobble	-76.70	7.5	-12.05	n/a
AC07b	AC07b_0	2016 surveys	48.58114	-53.82967	Mud	-58.10	5.2	-21.82	n/a
AC07b	AC07b_84	2016 surveys	48.58196	-53.82914	Mud	-61.10	9.2	-13.94	n/a
AC07b	AC07b_92	2016 surveys	48.58203	-53.82907	Mud	-59.60	7.6	-15.52	n/a
AC08	AC08_0	2016 surveys	48.58209	-53.87006	GMS	-43.90	9.6	-13.31	n/a
AC08	AC08_100	2016 surveys	48.58294	-53.86934	GMS	-57.90	12.9	-14.26	n/a
AC08	AC08_108	2016 surveys	48.58302	-53.86930	GMS	-57.80	14.4	-16.46	n/a

Station	Image ID	Source	Latitude (decimal degrees)	Longitude (decimal degrees)	Benthoscape Class	Depth (m)	Slope (deg)	EM1002 Backscatter (decibels)	EM3000 Backscatter (decibels)
AC08	AC08_16	2016 surveys	48.58220	-53.86988	GMS	-47.70	13.1	-12.37	n/a
AC08	AC08_28	2016 surveys	48.58230	-53.86978	GMS	-49.80	12.7	-14.57	n/a
AC08	AC08_36	2016 surveys	48.58237	-53.86974	GMS	-52.90	12.4	-13.94	n/a
AC08	AC08_64	2016 surveys	48.58262	-53.86957	GMS	-58.10	10.9	-8.27	n/a
AC08	AC08_72	2016 surveys	48.58269	-53.86950	GMS	-58.20	10.6	-11.42	n/a
AC08	AC08_84	2016 surveys	48.58279	-53.86940	GMS	-59.30	11.4	-12.68	n/a
AC08	AC08_92	2016 surveys	48.58286	-53.86937	GMS	-59.30	11.4	-14.26	n/a
AC08	AC08_48	2016 surveys	48.58249	-53.86969	Shallow Pebble/Cobble	-54.30	11.7	-13.63	n/a
AC09	AC09_0	2016 surveys	48.57794	-53.85963	GMS	-53.30	1.4	-15.20	n/a
AC09	AC09_80	2016 surveys	48.57850	-53.85902	GMS	-58.00	7.6	-12.68	n/a
AC09	AC09_92	2016 surveys	48.57858	-53.85893	GMS	-60.00	8.1	-14.57	n/a
AC09	AC09_24	2016 surveys	48.57809	-53.85941	Shallow Pebble/Cobble	-53.90	2.3	-12.68	n/a
AC09	AC09_36	2016 surveys	48.57817	-53.85933	Shallow Pebble/Cobble	-54.20	3.0	-12.68	n/a
AC09	AC09_44	2016 surveys	48.57823	-53.85926	Shallow Pebble/Cobble	-55.10	4.0	-12.05	n/a
AC09	AC09_52	2016 surveys	48.57829	-53.85921	Shallow Pebble/Cobble	-55.70	4.5	-12.68	n/a

Station	Image ID	Source	Latitude (decimal degrees)	Longitude (decimal degrees)	Benthoscape Class	Depth (m)	Slope (deg)	EM1002 Backscatter (decibels)	EM3000 Backscatter (decibels)
AC09	AC09_72	2016 surveys	48.57843	-53.85907	Shallow Pebble/Cobble	-57.00	6.2	-13.31	n/a
AC09	AC09_8	2016 surveys	48.57798	-53.85955	Shallow Pebble/Cobble	-53.40	1.7	-13.63	n/a
NS01	NS01_106	2016 surveys	48.62315	-53.76288	Deep Pebble/Cobble	-160.30	14.4	-18.35	n/a
NS01	NS01_146	2016 surveys	48.62311	-53.76266	Deep Pebble/Cobble	-159.10	14.2	-19.93	n/a
NS01	NS01_18	2016 surveys	48.62326	-53.76334	Deep Pebble/Cobble	-156.60	18.0	-13.00	n/a
NS01	NS01_38	2016 surveys	48.62323	-53.76324	Deep Pebble/Cobble	-157.60	16.8	-14.57	n/a
NS01	NS01_68	2016 surveys	48.62320	-53.76308	Deep Pebble/Cobble	-160.70	14.4	-17.72	n/a
NS01	NS01_160	2016 surveys	48.62311	-53.76258	Mud	-159.10	15.1	-18.35	n/a
NS05b	NS05b_14	2016 surveys	48.60960	-53.80628	Bedrock	-79.60	18.0	-9.53	n/a
NS05b	NS05b_0	2016 surveys	48.60965	-53.80606	Deep Pebble/Cobble	-83.60	22.9	-6.38	n/a
NS05b	NS05b_24	2016 surveys	48.60957	-53.80644	Deep Pebble/Cobble	-77.30	16.1	-10.16	n/a
NS05b	NS05b_38	2016 surveys	48.60954	-53.80666	Deep Pebble/Cobble	-73.70	12.1	-10.48	n/a
NS09	NS09_16	2016 surveys	48.60280	-53.77046	Bedrock	-70.10	26.1	-14.57	n/a
NS09	NS09_32	2016 surveys	48.60290	-53.77034	Bedrock	-72.00	13.6	-15.20	n/a
NS09	NS09_44	2016 surveys	48.60296	-53.77025	Bedrock	-70.20	16.5	-15.52	n/a

Station	Image ID	Source	Latitude (decimal degrees)	Longitude (decimal degrees)	Benthoscape Class	Depth (m)	Slope (deg)	EM1002 Backscatter (decibels)	EM3000 Backscatter (decibels)
NS09	NS09_108	2016 surveys	48.60328	-53.76968	Deep Pebble/Cobble	-75.30	18.1	-8.59	n/a
NS09	NS09_52	2016 surveys	48.60300	-53.77017	Deep Pebble/Cobble	-71.80	13.5	-13.94	n/a
NS09	NS09_68	2016 surveys	48.60307	-53.77002	Deep Pebble/Cobble	-72.80	16.5	-10.16	n/a
NS09	NS09_84	2016 surveys	48.60316	-53.76990	Deep Pebble/Cobble	-74.70	18.7	-8.59	n/a
NS09	NS09_92	2016 surveys	48.60321	-53.76983	Deep Pebble/Cobble	-73.60	18.2	-10.16	n/a
NS11	NS11_32	2016 surveys	48.60258	-53.80856	Bedrock	-156.90	19.1	-9.22	n/a
NS11	NS11_40	2016 surveys	48.60264	-53.80850	Bedrock	-161.40	17.7	-8.59	n/a
NS11	NS11_52	2016 surveys	48.60273	-53.80841	Bedrock	-162.80	17.0	-8.59	n/a
NS11	NS11_16	2016 surveys	48.60246	-53.80869	Mud	-151.90	20.6	-8.90	n/a
NS11	NS11_8	2016 surveys	48.60241	-53.80875	Mud	-150.00	21.3	-9.22	n/a
NS13	NS13_84	2016 surveys	48.59931	-53.77804	Bedrock	-34.50	6.1	-12.68	n/a
NS13	NS13_92	2016 surveys	48.59936	-53.77801	Bedrock	-35.40	7.5	-11.74	n/a
NS13	NS13_20	2016 surveys	48.59895	-53.77836	Shallow Pebble/Cobble	-33.50	3.2	-11.11	n/a
NS13	NS13_36	2016 surveys	48.59904	-53.77826	Shallow Pebble/Cobble	-33.90	5.1	-11.42	n/a
NS13	NS13_4	2016 surveys	48.59886	-53.77845	Shallow Pebble/Cobble	-33.30	3.0	-11.74	n/a

Station	Image ID	Source	Latitude (decimal degrees)	Longitude (decimal degrees)	Benthoscape Class	Depth (m)	Slope (deg)	EM1002 Backscatter (decibels)	EM3000 Backscatter (decibels)
NS13	NS13_52	2016 surveys	48.59912	-53.77817	Shallow Pebble/Cobble	-33.80	5.1	-12.37	n/a
NS13	NS13_68	2016 surveys	48.59922	-53.77812	Shallow Pebble/Cobble	-34.40	6.6	-12.37	n/a
NS14	NS14_108	2016 surveys	48.59938	-53.82077	GMS	-49.40	1.1	-13.00	n/a
NS14	NS14_20	2016 surveys	48.59974	-53.81961	GMS	-48.90	1.7	-11.11	n/a
NS14	NS14_36	2016 surveys	48.59966	-53.81981	GMS	-49.20	1.9	-11.11	n/a
NS14	NS14_44	2016 surveys	48.59963	-53.81992	GMS	-49.30	1.4	-11.11	n/a
NS14	NS14_56	2016 surveys	48.59960	-53.82009	GMS	-49.20	1.0	-11.11	n/a
NS14	NS14_64	2016 surveys	48.59956	-53.82019	GMS	-49.40	0.8	-11.42	n/a
NS14	NS14_72	2016 surveys	48.59951	-53.82028	GMS	-49.40	1.0	-11.42	n/a
NS14	NS14_8	2016 surveys	48.59982	-53.81948	GMS	-48.70	1.8	-11.42	n/a
NS14	NS14_96	2016 surveys	48.59941	-53.82060	GMS	-49.50	1.2	-12.68	n/a
NS14	NS14_28	2016 surveys	48.59969	-53.81970	Shallow Pebble/Cobble	-48.90	1.7	-11.11	n/a
NS15	NS15_104	2016 surveys	48.59848	-53.78426	Mud	-82.50	5.5	-18.04	n/a
NS15	NS15_16	2016 surveys	48.59801	-53.78474	Mud	-81.00	5.5	-14.26	n/a
NS15	NS15_36	2016 surveys	48.59811	-53.78463	Mud	-80.90	4.9	-14.89	n/a

Station	Image ID	Source	Latitude (decimal degrees)	Longitude (decimal degrees)	Benthoscape Class	Depth (m)	Slope (deg)	EM1002 Backscatter (decibels)	EM3000 Backscatter (decibels)
NS15	NS15_52	2016 surveys	48.59821	-53.78456	Mud	-81.40	4.3	-18.98	n/a
NS15	NS15_76	2016 surveys	48.59833	-53.78442	Mud	-81.50	4.4	-17.41	n/a
NS15	NS15_8	2016 surveys	48.59796	-53.78478	Mud	-80.40	5.7	-15.52	n/a
NS15	NS15_92	2016 surveys	48.59841	-53.78433	Mud	-81.70	5.0	-16.46	n/a
NS17	NS17_73	2016 surveys	48.59481	-53.80072	Bedrock	-103.20	27.9	-16.78	n/a
NS17	NS17_86	2016 surveys	48.59489	-53.80059	Bedrock	-101.30	27.2	-17.09	n/a
NS17	NS17_0	2016 surveys	48.59426	-53.80126	Deep Pebble/Cobble	-96.40	22.1	-17.09	n/a
NS17	NS17_100	2016 surveys	48.59498	-53.80046	Mud	-100.00	24.5	-17.09	n/a
NS17	NS17_112	2016 surveys	48.59507	-53.80037	Mud	-102.90	23.4	-16.78	n/a
NS17	NS17_41	2016 surveys	48.59455	-53.80092	Mud	-100.60	27.3	-16.78	n/a
NS17	NS17_54	2016 surveys	48.59466	-53.80085	Mud	-104.40	26.7	-17.41	n/a
NS17	NS17_64	2016 surveys	48.59474	-53.80079	Mud	-104.40	26.7	-15.83	n/a
NS18	NS18_103	2016 surveys	48.59420	-53.79236	Mud	-97.70	3.8	-24.65	n/a
NS18	NS18_43	2016 surveys	48.59401	-53.79274	Mud	-96.20	3.8	-19.61	n/a
NS18	NS18_67	2016 surveys	48.59407	-53.79258	Mud	-97.00	3.7	-22.13	n/a

Station	Image ID	Source	Latitude (decimal degrees)	Longitude (decimal degrees)	Benthoscape Class	Depth (m)	Slope (deg)	EM1002 Backscatter (decibels)	EM3000 Backscatter (decibels)
NS18	NS18_7	2016 surveys	48.59394	-53.79300	Mud	-95.00	3.6	-20.56	n/a
NS18	NS18_91	2016 surveys	48.59416	-53.79244	Mud	-97.70	3.4	-23.08	n/a
NS19	NS19_0	2016 surveys	48.59458	-53.82205	Mud	-143.30	8.3	-10.16	n/a
NS19	NS19_101	2016 surveys	48.59488	-53.82153	Mud	-142.60	1.9	-9.85	n/a
NS19	NS19_113	2016 surveys	48.59490	-53.82145	Mud	-143.10	4.9	-10.16	n/a
NS19	NS19_29	2016 surveys	48.59469	-53.82193	Mud	-143.40	6.2	-9.85	n/a
NS19	NS19_53	2016 surveys	48.59476	-53.82180	Mud	-142.70	4.1	-9.85	n/a
NS19	NS19_73	2016 surveys	48.59482	-53.82169	Mud	-142.60	1.7	-14.26	n/a
NS20	NS20_106	2016 surveys	48.59529	-53.82715	Deep Pebble/Cobble	-103.20	23.0	-13.00	n/a
NS20	NS20_92	2016 surveys	48.59520	-53.82725	Deep Pebble/Cobble	-106.20	21.1	-13.00	n/a
NS20	NS20_44	2016 surveys	48.59496	-53.82765	Mud	-110.00	18.2	-13.31	n/a
NS20	NS20_64	2016 surveys	48.59502	-53.82744	Mud	-112.00	18.3	-13.00	n/a
NS20	NS20_78	2016 surveys	48.59511	-53.82735	Mud	-108.70	19.6	-13.00	n/a
NS20	NS20_8	2016 surveys	48.59481	-53.82799	Mud	-115.50	18.4	-12.68	n/a
NS21b	NS21b_2	2016 surveys	48.59369	-53.83630	Bedrock	-48.30	26.7	-8.59	n/a

Station	Image ID	Source	Latitude (decimal degrees)	Longitude (decimal degrees)	Benthoscape Class	Depth (m)	Slope (deg)	EM1002 Backscatter (decibels)	EM3000 Backscatter (decibels)
NS21b	NS21b_20	2016 surveys	48.59367	-53.83657	Mixed Boulder	-40.00	16.1	-8.59	n/a
NS21b	NS21b_14	2016 surveys	48.59367	-53.83648	Shallow Pebble/Cobble	-43.70	23.2	-8.59	n/a
NS24	NS24_0	2016 surveys	48.59238	-53.83115	Mud	-132.70	11.6	-16.78	n/a
NS24	NS24_34	2016 surveys	48.59250	-53.83094	Mud	-130.70	11.0	-16.46	n/a
NS24	NS24_54	2016 surveys	48.59256	-53.83082	Mud	-128.60	8.6	-17.09	n/a
NS24	NS24_74	2016 surveys	48.59261	-53.83068	Mud	-127.20	4.9	-15.83	n/a
NS24	NS24_94	2016 surveys	48.59267	-53.83055	Mud	-127.00	3.4	-15.20	n/a
NS25	NS25_16	2016 surveys	48.59110	-53.80561	GMS	-47.20	4.8	-18.67	n/a
NS25	NS25_24	2016 surveys	48.59118	-53.80559	GMS	-47.20	4.0	-18.67	n/a
NS25	NS25_32	2016 surveys	48.59126	-53.80557	GMS	-47.60	3.0	-18.98	n/a
NS25	NS25_40	2016 surveys	48.59134	-53.80554	GMS	-47.80	2.1	-21.82	n/a
NS25	NS25_48	2016 surveys	48.59142	-53.80551	GMS	-48.00	1.0	-17.09	n/a
NS25	NS25_56	2016 surveys	48.59149	-53.80545	GMS	-47.70	3.4	-15.52	n/a
NS25	NS25_72	2016 surveys	48.59165	-53.80539	GMS	-45.70	10.0	-16.15	n/a
NS25	NS25_80	2016 surveys	48.59173	-53.80537	GMS	-44.00	8.2	-13.31	n/a

Station	Image ID	Source	Latitude (decimal degrees)	Longitude (decimal degrees)	Benthoscape Class	Depth (m)	Slope (deg)	EM1002 Backscatter (decibels)	EM3000 Backscatter (decibels)
NS25	NS25_88	2016 surveys	48.59181	-53.80536	GMS	-44.00	8.2	-11.74	n/a
NS25	NS25_96	2016 surveys	48.59189	-53.80532	GMS	-43.50	5.1	-11.74	n/a
NS25	NS25_0	2016 surveys	48.59095	-53.80571	Mud	-46.60	6.0	-18.35	n/a
NS27	NS27_0	2016 surveys	48.59128	-53.80917	Bedrock	-119.40	43.8	-16.15	n/a
NS27	NS27_104	2016 surveys	48.59167	-53.80819	Bedrock	-106.10	52.3	-20.24	n/a
NS27	NS27_132	2016 surveys	48.59175	-53.80791	Bedrock	-96.90	51.8	-23.71	n/a
NS27	NS27_152	2016 surveys	48.59180	-53.80771	Bedrock	-90.20	51.9	-21.19	n/a
NS27	NS27_26	2016 surveys	48.59137	-53.80891	Bedrock	-113.90	41.5	-14.89	n/a
NS27	NS27_34	2016 surveys	48.59140	-53.80884	Bedrock	-121.20	43.6	-16.46	n/a
NS27	NS27_44	2016 surveys	48.59143	-53.80874	Bedrock	-115.70	42.9	-16.46	n/a
NS28	NS28_0	2016 surveys	48.59111	-53.83474	Mud	-121.00	7.1	-12.05	n/a
NS28	NS28_116	2016 surveys	48.59177	-53.83393	Mud	-120.40	10.3	-18.04	n/a
NS28	NS28_40	2016 surveys	48.59132	-53.83443	Mud	-122.20	9.0	-15.20	n/a
NS28	NS28_76	2016 surveys	48.59154	-53.83421	Mud	-120.60	10.0	-18.67	n/a
NS28	NS28_8	2016 surveys	48.59115	-53.83467	Mud	-121.00	7.1	-12.05	n/a

Station	Image ID	Source	Latitude (decimal degrees)	Longitude (decimal degrees)	Benthoscape Class	Depth (m)	Slope (deg)	EM1002 Backscatter (decibels)	EM3000 Backscatter (decibels)
NS28	NS28_84	2016 surveys	48.59159	-53.83416	Mud	-122.00	10.3	-18.67	n/a
NS29	NS29_20	2016 surveys	48.58942	-53.84238	GMS	-62.80	11.9	-16.46	n/a
NS29	NS29_28	2016 surveys	48.58938	-53.84255	GMS	-62.00	12.3	-15.20	n/a
NS29	NS29_38	2016 surveys	48.58933	-53.84278	GMS	-59.30	13.8	-12.05	n/a
NS29	NS29_4	2016 surveys	48.58947	-53.84201	GMS	-67.90	10.0	-15.83	n/a
NS29	NS29_100	2016 surveys	48.58886	-53.84402	Shallow Pebble/Cobble	-50.50	18.2	-12.05	n/a
NS29	NS29_104	2016 surveys	48.58884	-53.84411	Shallow Pebble/Cobble	-50.50	18.2	-12.05	n/a
NS29	NS29_112	2016 surveys	48.58884	-53.84429	Shallow Pebble/Cobble	-49.10	15.1	-13.63	n/a
NS29	NS29_48	2016 surveys	48.58928	-53.84300	Shallow Pebble/Cobble	-57.20	14.3	-11.74	n/a
NS29	NS29_56	2016 surveys	48.58922	-53.84316	Shallow Pebble/Cobble	-54.10	14.8	-11.74	n/a
NS29	NS29_64	2016 surveys	48.58916	-53.84332	Shallow Pebble/Cobble	-53.30	15.4	-10.48	n/a
NS29	NS29_72	2016 surveys	48.58909	-53.84347	Shallow Pebble/Cobble	-51.00	15.4	-10.48	n/a
NS29	NS29_84	2016 surveys	48.58900	-53.84372	Shallow Pebble/Cobble	-51.10	18.2	-10.79	n/a
NS29	NS29_94	2016 surveys	48.58891	-53.84391	Shallow Pebble/Cobble	-52.20	19.3	-12.37	n/a
NS31	NS31_36	2016 surveys	48.58530	-53.83344	Mud	-128.10	0.2	-14.89	n/a

Station	Image ID	Source	Latitude (decimal degrees)	Longitude (decimal degrees)	Benthoscape Class	Depth (m)	Slope (deg)	EM1002 Backscatter (decibels)	EM3000 Backscatter (decibels)
NS31	NS31_66	2016 surveys	48.58532	-53.83330	Mud	-128.30	0.9	-15.52	n/a
NS31	NS31_8	2016 surveys	48.58530	-53.83358	Mud	-128.20	0.3	-14.26	n/a
NS31	NS31_94	2016 surveys	48.58533	-53.83316	Mud	-128.20	1.2	-19.93	n/a
NS37	NS37_12	2016 surveys	48.58236	-53.83327	Bedrock	-61.90	37.5	-15.83	n/a
NS37	NS37_22	2016 surveys	48.58241	-53.83311	Bedrock	-58.60	36.0	-13.94	n/a
NS37	NS37_26	2016 surveys	48.58244	-53.83305	Bedrock	-64.70	37.6	-13.94	n/a
NS37	NS37_40	2016 surveys	48.58252	-53.83284	Bedrock	-58.40	35.7	-13.94	n/a
NS37	NS37_50	2016 surveys	48.58257	-53.83268	Bedrock	-53.90	31.1	-15.52	n/a
NS37	NS37_6	2016 surveys	48.58233	-53.83336	Bedrock	-65.60	39.7	-16.46	n/a
NS39	NS39_0	2016 surveys	48.58056	-53.84021	Mud	-113.10	3.3	-16.78	n/a
NS39	NS39_30	2016 surveys	48.58062	-53.83992	Mud	-113.60	2.9	-13.31	n/a
NS39	NS39_40	2016 surveys	48.58065	-53.83982	Mud	-114.30	3.0	-13.63	n/a
NS39	NS39_48	2016 surveys	48.58066	-53.83974	Mud	-114.40	2.8	-12.68	n/a
NS39	NS39_68	2016 surveys	48.58071	-53.83955	Mud	-114.90	2.2	-13.94	n/a
NS39	NS39_76	2016 surveys	48.58073	-53.83948	Mud	-114.80	2.9	-10.48	n/a

Station	Image ID	Source	Latitude (decimal degrees)	Longitude (decimal degrees)	Benthoscape Class	Depth (m)	Slope (deg)	EM1002 Backscatter (decibels)	EM3000 Backscatter (decibels)
NS39	NS39_92	2016 surveys	48.58074	-53.83931	Mud	-114.30	4.6	-9.53	n/a
NS43	NS43_0	2016 surveys	48.57737	-53.84508	Mud	-94.70	2.9	-17.72	n/a
NS43	NS43_102	2016 surveys	48.57796	-53.84422	Mud	-99.90	3.3	-24.02	n/a
NS43	NS43_114	2016 surveys	48.57803	-53.84411	Mud	-100.00	3.1	-24.34	n/a
NS43	NS43_16	2016 surveys	48.57748	-53.84496	Mud	-95.40	3.4	-19.30	n/a
NS43	NS43_38	2016 surveys	48.57757	-53.84475	Mud	-96.20	3.8	-19.61	n/a
NS43	NS43_68	2016 surveys	48.57776	-53.84450	Mud	-97.90	3.2	-22.45	n/a
NS43	NS43_75	2016 surveys	48.57780	-53.84445	Mud	-98.40	3.2	-22.13	n/a
NS43	NS43_84	2016 surveys	48.57786	-53.84438	Mud	-98.60	3.1	-21.82	n/a
Buckley1_e	Buckley1_e	Copeland 2006	48.58441	-53.90362	Mixed Boulder	-7.50	3.1	n/a	-7.01
Buckley1_s	Buckley1_s	Copeland 2006	48.58398	-53.90428	Mixed Boulder	-13.30	15.4	n/a	-6.38
Buckley2_e	Buckley2_e	Copeland 2006	48.58452	-53.90241	Mixed Boulder	-8.40	6.9	n/a	-6.70
Buckley2_s	Buckley2_s	Copeland 2006	48.58447	-53.90319	Mixed Boulder	-7.40	2.4	n/a	-7.01
Buckley3_s	Buckley3_s	Copeland 2006	48.58491	-53.90355	Sand	-6.80	0.6	n/a	-5.12
Cliff	Cliff	Copeland 2006	48.57725	-53.91677	Bedrock	-14.50	40.4	n/a	-6.70

Station	Image ID	Source	Latitude (decimal degrees)	Longitude (decimal degrees)	Benthoscape Class	Depth (m)	Slope (deg)	EM1002 Backscatter (decibels)	EM3000 Backscatter (decibels)
D10_e	D10_e	Copeland 2006	48.58030	-53.90958	Rhodolith	-17.60	1.5	-16.46	-5.75
D10_s	D10_s	Copeland 2006	48.58150	-53.91080	Mixed Boulder	-14.20	4.4	n/a	-5.75
D11_e	D11_e	Copeland 2006	48.57978	-53.90744	Rhodolith	-20.40	4.6	-21.82	-6.07
D11_s	D11_s	Copeland 2006	48.58096	-53.91156	Rhodolith	-16.20	0.8	-18.04	-6.07
D12_e	D12_e	Copeland 2006	48.57936	-53.91036	Rhodolith	-13.70	6.5	n/a	-6.07
D12_s	D12_s	Copeland 2006	48.58010	-53.91198	Rhodolith	-13.50	2.3	-17.72	-5.75
D126_e	D126_e	Copeland 2006	48.57541	-53.90871	Sand	-21.60	6.9	n/a	-7.64
D126_s	D126_s	Copeland 2006	48.57628	-53.90926	Sand	-15.50	6.2	-46.38	-8.27
D13_e	D13_e	Copeland 2006	48.57776	-53.90933	Mixed Boulder	-12.80	5.7	n/a	-5.75
D13_s	D13_s	Copeland 2006	48.57801	-53.90968	GMS	-11.10	2.5	n/a	-7.64
D15	D15	Copeland 2006	48.58201	-53.90545	GMS	-30.20	3.1	-10.48	-6.38
D20	D20	Copeland 2006	48.58218	-53.89248	Mixed Boulder	-33.90	0.8	-12.05	n/a
D22_e	D22_e	Copeland 2006	48.58071	-53.88056	Mixed Boulder	-41.30	3.8	-13.63	n/a
D22_s	D22_s	Copeland 2006	48.58139	-53.88072	Mixed Boulder	-38.20	1.8	-12.68	n/a
D25_e	D25_e	Copeland 2006	48.58361	-53.87265	Mixed Boulder	-41.50	4.0	-11.11	n/a

Station	Image ID	Source	Latitude (decimal degrees)	Longitude (decimal degrees)	Benthoscape Class	Depth (m)	Slope (deg)	EM1002 Backscatter (decibels)	EM3000 Backscatter (decibels)
D25_s	D25_s	Copeland 2006	48.58351	-53.87318	Mixed Boulder	-42.10	4.6	-12.37	n/a
D26_e	D26_e	Copeland 2006	48.58311	-53.86462	Mixed Boulder	-15.70	2.0	-15.20	n/a
D26_s	D26_s	Copeland 2006	48.58323	-53.86527	Mixed Boulder	-14.60	0.6	-14.26	n/a
D4_e	D4_e	Copeland 2006	48.57772	-53.92943	Mud	-50.50	0.3	-22.13	n/a
D4_s	D4_s	Copeland 2006	48.57746	-53.92198	Mud	-50.10	0.5	-25.60	-12.05
D7_e	D7_e	Copeland 2006	48.58458	-53.91215	Sand	-7.50	0.4	n/a	-6.38
D7_s	D7_s	Copeland 2006	48.58581	-53.91320	Sand	-7.50	2.3	n/a	-7.96
D8_e	D8_e	Copeland 2006	48.58280	-53.91304	Mixed Boulder	-22.70	15.3	-9.53	-4.49
D8_s	D8_s	Copeland 2006	48.58349	-53.91410	GMS	-34.50	13.3	-10.48	-6.38
D9_e	D9_e	Copeland 2006	48.58118	-53.91291	Rhodolith	-16.30	1.2	-16.78	-6.70
D9_s	D9_s	Copeland 2006	48.58186	-53.91345	Mixed Boulder	-19.10	16.5	n/a	-7.33
Dive3_e	Dive3_e	Copeland 2006	48.57704	-53.91073	Sand	-9.30	0.6	n/a	-8.27
Dive3_s	Dive3_s	Copeland 2006	48.57669	-53.91032	Sand	-9.90	1.0	-8.90	-8.90
Dive4_e	Dive4_e	Copeland 2006	48.57669	-53.90995	Sand	-10.50	1.7	n/a	-8.59
Dive4_s	Dive4_s	Copeland 2006	48.57648	-53.90995	Sand	-11.20	3.0	-9.22	-9.22

Station	Image ID	Source	Latitude (decimal degrees)	Longitude (decimal degrees)	Benthoscape Class	Depth (m)	Slope (deg)	EM1002 Backscatter (decibels)	EM3000 Backscatter (decibels)
Dive5_e	Dive5_e	Copeland 2006	48.57676	-53.91343	Mixed Boulder	-7.20	1.2	n/a	-4.81
Dive5_s	Dive5_s	Copeland 2006	48.57638	-53.91371	GMS	-6.50	2.9	n/a	-4.49
Dive6_e	Dive6_e	Copeland 2006	48.57629	-53.91371	GMS	-6.60	2.6	n/a	-7.01
Dive6_s	Dive6_s	Copeland 2006	48.57591	-53.91343	GMS	-7.20	0.7	n/a	-7.33
Prac_e	Prac_e	Copeland 2006	48.57740	-53.91570	Mixed Boulder	-11.60	18.6	n/a	-6.70
Prac_s	Prac_s	Copeland 2006	48.57700	-53.91630	GMS	-10.10	9.1	n/a	-7.33
ROVA_e	ROVA_e	Copeland 2006	48.58050	-53.90850	Rhodolith	-18.40	0.9	-8.90	-5.75
ROVA_s	ROVA_s	Copeland 2006	48.57980	-53.90840	Rhodolith	-17.70	1.9	-21.19	-6.07
ROVB_e	ROVB_e	Copeland 2006	48.57950	-53.87930	GMS	-57.00	11.5	-9.85	n/a
ROVB_s	ROVB_s	Copeland 2006	48.57950	-53.87950	Sand	-55.90	10.7	-11.11	n/a
ROVD_e	ROVD_e	Copeland 2006	48.62050	-53.69660	Bedrock	-83.50	59.2	-10.16	n/a
ROVD_s	ROVD_s	Copeland 2006	48.62030	-53.69750	Bedrock	-118.30	65.3	-10.79	n/a
ROVE_e	ROVE_e	Copeland 2006	48.66260	-53.59390	Bedrock	-32.30	7.2	-13.31	n/a
ROVE_s	ROVE_s	Copeland 2006	48.66270	-53.59410	Bedrock	-33.30	8.2	-12.05	n/a
Stamford_e	Stamford_e	Copeland 2006	48.57110	-53.90920	Mixed Boulder	-6.90	2.7	n/a	-8.27
Stamford_s	Stamford_s	Copeland 2006	48.57040	-53.90960	Mixed Boulder	-7.90	0.6	n/a	-11.42