A MULTI-CRITERIA METHOD FOR MAKING TRADEOFFS AND HARD DECISIONS SPATIALLY EXPLICIT IN MARINE CONSERVATION PLANNING

by

©Melinda T. Agapito

A Thesis submitted to the

School of Graduate Studies

In partial fulfillment of the requirements for the degree of

Doctor of Philosophy

Department of Geography

Memorial University of Newfoundland

April 2016

St. John's, Newfoundland and Labrador

THE WAYS OF MAN DO NOT BELONG TO HIM [JEREMIAH 10:23]

ABSTRACT

Identifying new marine protected areas (MPAs) typically requires considering competing priorities from a large range of stakeholders. While balancing socioeconomic losses with biodiversity gains is challenging, it is central to the planning process and will influence the effectiveness of the MPAs to be created.

This paper presents a new decision-support method named *Spatial Tier Framework-Ordered Weighted Averaging (STF-OWA)* that allows stakeholders to share their values and explore alternative planning scenarios, by varying levels of losses and gains, in a collaborative setting. Unlike methods that aim at finding one optimal solution (e.g. Marxan), the STF-OWA provides stakeholders with alternative planning options based on weights reflecting their priorities among and between biodiversity interests (e.g. corals vs. birds) and socioeconomic interests (e.g. fishing employment vs. fishing dollars). The approach was tested in the Newfoundland and Labrador shelf bioregion, Atlantic Canada (~1.2x106 km^2), using scientific survey data on groundfish, seabirds, and habitat-forming invertebrates, commercial fishing logbooks, data on marine transportation, and oil and gas activities.

Results show that the STF-OWA can identify easy-to-implement conservation sites (i.e. high biodiversity with low socioeconomic activities), although they represent only <5% of the analyzed area. Subsequently, the STF-OWA demonstrated that identifying >5% of the study area as an MPA often involves hard decision areas

(i.e. sites with both high socioeconomic impacts and high biodiversity gains). On making tradeoffs and hard decisions spatially explicit, the STF-OWA: (1) offers various options such as cheap, cost-effective, and expensive scenarios, making the toughest conservation decisions spatially explicit -- namely, tough decisions for and against biodiversity protection and tough decisions for and against socioeconomic protection; (2) allows visualizing multiple competing interests in a solution set that provides empirical evidence that a win-win option is rare; and (3) permits delineating regions of interest (ROIs) and percent area targets within a conservation scenario that makes balancing loss and gain more spatially explicit at a finer scale.

With these features available in the STF-OWA decision-support method, it is possible to identify not only the areas that minimize potential conflicts, but also areas of high importance for biological protection, and to do so without masking the tough political decisions needed in advancing conservation goals.

ACKNOWLEDGEMENTS

This project would not have been possible without the many people who have given me their time, expertise, financial, and moral support.

My supervisors, Rodolphe Devillers and Evan Edinger, have provided guidance, financial support, expertise, and 'push' from the very onset of this work.

I am also pleased to have had dedicated members of the supervising committee, such as Ratana Chuenpagdee who has provided a generous amount of her time on this work, as well as some financial support. Also, I am thankful for the support of Mariano Koen-Alonso, particularly in acquiring and understanding the data and review of this work.

I am also grateful to the reviewers who provided excellent feedback on this work, namely: Dr Natalie Ban (University of Victoria), Dr Peter Keller (University of Victoria), and Dr Paul Snelgrove (Memorial University of Newfoundland).

Undoubtedly, this research would not have been possible without those who spent their time to make the data available for this work. Some of them include: Carina Gjerdrum and Karel Allard (Canadian Wildlife Service), Sandra and Anne-Marie Russell (DFO Policy and Economics Branch), Debbie Power, Neil Ollerhead, and Vonda Wareham (DFO-St. John's), Andrew Szeto and John Mahdi (Canadian Coast Guard-Maritime Security), Alfonzo Perez Rodriguez (Institute of Marine Research, Norway), and Information Department of Canada-Newfoundland and Labrador Offshore Petroleum Board.

Many others have extended their expertise to answer data-related questions, such as Don Power, Vonda Wareham, Mardi Gullage, Tony Bowdring, Dave Orr, Eugene Murphy, and Sharmane Allen of DFO-St. John's, Keith Sullivan of Fish, Food & Allied Workers, Andrew Szeto of CCG-MS, and Barbara Best of DFO-Ottawa. Still many others spent their precious time at a one-day workshop for the test-run of the STF-OWA.

I also deeply appreciate the community and financial support from the Canadian Healthy Oceans Network (CHONE). It was such a great opportunity to be part of a network of marine researchers that enriched my research perspective, particularly in making the connection between conservation science and policy.

I am glad and thankful as well for being part of the Marine Geomatics Lab where respectful and mature labmates made this long journey very pleasant. It has been made even more pleasant by administrative and technical support from the dedicated staff of the Memorial University, Department of Geography.

I am grateful to friends who have shared their time to review and proofread my thesis sections or chapters: Adams Amantana, Randal Greene, Neha Gupta, Laura Jackson, Lara Miles, Wilfredo Diegor, Sharmane Allen, Andrew Song, Jaime Teran, Philippe Ferrand, and James Rosario. Also to many other friends who in one way or another made my university/Newfoundland life a lot easier and happier.

My special thanks to the Diegors (Elizabeth and Wilfredo) who have been very generous in sharing their time that made my graduate life a lot more inspiring.

Finally my deepest gratitude to my family (Mama Carmen, brothers, sisters, nephews, nieces, sister in law, and brother-in-law Willy), who for their undying and loving support, made my load lighter.

TABLE OF CONTENTS

List of Tables

List of Figures

List of Abbreviations

ALO	At Least One
CCG	Canadian Coast Guard
C-NLOPB	Canada-Newfoundland and Labrador Oil and Petroleum Board
CBD	Convention on Biological Diversity
CWS	Canadian Wildlife Service
COSEWIC	Committee on the Status of Endangered Wildlife in Canada
DFO	Department of Fisheries and Oceans Canada
ECSAS	Eastern Canada Seabirds at Sea
ENGOs	Environmental Non-Government Organizations
FAO	Food and Agriculture Organization
FFAW	Fish, Food and Allied Workers
FGs	Fisher Groups
GEBCO	General Bathymetric Chart of the Oceans
GIS	Geographic Information System
HDs	Hard Decisions
LRIT	Long Range Identification and Tracking
MPA	Marine Protected Area
MCDA	Multi-Criteria Decision Analysis
MSS	Multispecies Survey
NAFO	Northwest Atlantic Fisheries Organization
OWA	Ordered Weighted Averaging
O & G	Oil and Gas

PA	Protected Area
PIROP	Programme Intégré de Recherches sur les Oiseaux Pélagiques
PU	Planning Unit
SCP	Spatial Conservation Planning
SOLAS	Safety of Life at Sea
STF	Spatial Tier Framework
UN	United Nations
VTS	Vessel, Track, Set
WLC	Weighted Linear Combinations

List of Appendices

Co-Authorship Statement

This thesis is written in manuscript style with a goal of publishing three of its core sections. The candidate led the writing of all the chapters with improvements based on feedback from the supervisory committee.

The candidate developed and wrote the topic for chapter 2 in close coordination with committee member Dr Ratana Chuenpagdee. Dr Rodolphe Devillers assisted with the initial structuring of the chapter, and Dr Chuenpagdee helped with tuning and organizing the sections. Dr Chuenpagdee, Dr. Devillers, and Dr. Evan Edinger provided feedbacks on several versions of this chapter.

The candidate presented Chapter 3 with the goal of submitting it as a DFO technical report. Data analyses presented in this chapter were conducted by the candidate alone. The candidate wrote the first full draft of the chapter and revisions were done based on the two co-supervisors' feedback. Dr Mariano Koen-Alonso provided feedback on the final draft and helped facilitate the acquisition of some of the data used in this study.

Chapter 4 was written for submission to a journal after completion of the thesis (possibly in a shorter form). The general research focus and multi-criteria decision analysis (MCDA) approaches were first suggested by co-supervisor Dr Devillers.

The candidate through literature grounding identified the specific MCDA method and developed the decision-support method presented in this chapter. Data analysis was conducted exclusively by the candidate. Both co-supervisors provided suggestions and comments on all drafts of Chapter 4. All of the committee members participated to varying degrees, in providing feedback on the work done in this chapter. Everyone helped with preparing the workshop for the test-run of the proposed method (STF-OWA). Dr Chuenpagdee provided close guidance in preparing the workshop materials and in programming and facilitating the workshop itself.

CHAPTER 1 INTRODUCTION

Human impacts on ecosystems can be traced for thousands of years, the significance of which now defines what is commonly referred to as "Anthropocene" era (Ellis et al. 2013). In the marine realm, anthropogenic impacts are diverse in nature (e.g. Halpern et al. 2008), with overfishing being among the major concerns (Crowder & Norse 2005, Preikshot & Pauly 2005). The global decline in fisheries and the increasing threats to marine biodiversity have attracted attention from various perspectives such as research, development, and policy (Beverton & Holt 1957, Platteau 1989). Pauly (1995) suggested that the decline of fish stocks occurred many decades ago, even with the use of traditional fishing tools and methods. During the 20th century alone, it was estimated that more than 80% of high trophic level biomass had declined (Christensen 2000). The issue of overfishing had been documented prior to the 19th century such as the overexploitation of the local stocks in Europe during the middle ages (Roberts 2007), and the fishing collapse in coastal areas of Newfoundland in the early 1700s (Rose 2007). Overexploitation continued to receive international attention to put in place fishery regulatory measures in an attempt to curb the decline of fish stocks (ICO 1946).

The increasing demand for fish and seafood from a world population that increased about fourfold in the last century alone, combined with the availability of modern fishing methods and tools, resulted in a rapid decline of predatory fish (Myers & Worm 2003) and many other species. Several fish stocks eventually collapsed, resulting in closures of wild fishery activities, with one prominent example being the moratorium of the Newfoundland and Labrador cod fishery in 1992 (Hutchings & Myers 1994, Myers et al. 1996). Statistics from the Food and Agriculture Organization (FAO) of the United Nations estimate that more than 80% of global fish stocks have either already collapsed, or are over or fully exploited (Pauly et al. 2013).

The loss of biodiversity was not given much attention until the 1990s, almost two decades after species extinctions became apparent in terrestrial regions (Norse & Crowder 2005) and five decades after overexploitation of fisheries was noticed. Not surprisingly, the documentation of marine biodiversity decline and extinctions started with species that are mostly fished, caught as by-catch, and those indirectly impacted ecologically and biologically by fishing (Roberts & Hawkins 1999, Myers & Ottensmeyer 2005).

Beverton & Holt (1957) were the first authors to research scientifically the utility of marine reserves, as a way to address the situation, although they did not explicitly proposed to use them as a fisheries management tool. Instead, they suggested using fishery regulatory measures that could control fishing intensity, such as a reduction of motor power of fishing fleets, control of fishing activity and closures. However, these regulatory measures (e.g. quotas, closures), in general, have not had a significant effect in controlling excessive resource exploitation and protecting marine ecosystems (FAO 2011). This led to the growing interest in marine reserves or marine protected areas (MPAs), to limit or prohibit human exploitation, as a means to help address the impact of overexploitation of resources (Allison et al. 1998, Murray et al. 1999, Lubchenco et al. 2003). An MPA is defined by the IUCN (1994) as "any area of intertidal or subtidal terrain, together with its overlying water and associated flora, fauna, historical and cultural features, which has been reserved by law or other effective means to protect part or all of the enclosed environment".

Since its inception, the concept of MPAs has faced many challenges. One of them is the critique of its effectiveness where rigorous empirical evidence was lacking (Willis 2003), although a growing literature indicates that MPAs can work in some contexts (Edgar et al. 2014). This challenge is partly due to the difficulty of designing appropriate studies that can compare changes in ecosystems within and outside reserves (Roberts & Polunin 1993, Pauly 1995). Other challenges linked to MPA effectiveness include the difficulty of implementing, managing, and enforcing MPAs. Some of these relate to lack of political will and legal support (Eisma et al. 2005) and the socioeconomic challenges such as lack of compliance and uncertain outcomes associated with MPAs (Rudd et al. 2001, James et al. 2001, Macintosh et al. 2010).

Challenges also relate to methods used to identify suitable protected areas. Over the last four decades, methods for determining spatial priorities for conservation have been explored (Justus & Sarkar 2002, Williams et al. 2004, Moilanen et al. 2009). Most of these methods are computer-based and aim at balancing conservation tradeoffs that is space gained for biodiversity protection against space foregone for socioeconomic activities. These planning methods are often placed into two categories: optimization and non-optimization methods. The first type attempts to deliver optimal or near-optimal solutions based on sets of spatial priorities for conservation. Some of these are implemented in conservation tools such as Marxan and C-Plan (Moilanen et al. 2009). The second category, non-optimization methods, does not strictly search for an optimal solution but rather emphasizes stakeholder participation (Brown et al. 2001), including for instance MIRADI and SeaSketch (Salafsky 2011, Mitsova et al. 2013)

The following sections of this chapter present the research gap in developing conservation planning methods, research questions, goals and objectives, introduction to the conservation decision-support method proposed in this study, research stages, and the chapter outline of this thesis.

1.1 RESEARCH GAP IN DEVELOPING CONSERVATION PLANNING METHODS

Over the last four decades, a systematic form of conservation planning method started as simple scoring techniques (Margules & Usher 1981), followed by near optimal and optimal models (Pressey & Nicholls 1989, Justus & Sarkar 2002). The implementation of these methods as spatial decision support tools has become common in conservation planning research (Ball & Possingham 2000, Carwardine 2007).

To date, most of these planning methods and tools are framed around the integration of biological data, making them more robust in setting biological priorities. In contrast, social priorities are often loosely incorporated (Ban & Klein 2009, Adams et al. 2010). Some reasons are that social data are mostly qualitative and thus are hard to incorporate in quantitative models, too localized or too sparse and therefore difficult to integrate, or are simply not available or expensive to generate. Also, optimizing biological priorities for several social priorities is difficult mathematically using a single algorithm (von Neumann & Morgenstern 1947, p 11).

While the primacy of biological criteria was traditionally desired for conservation planning, the multiple socioeconomic criteria present practical constraints, particularly in places where economic and political challenges proved to be difficult issues (Oracion et al. 2005, Silva 2006, Pulgar-Vidal 2010, Roe & Walpole 2010). The importance of socioeconomics in conservation planning led some to suggest that finding a suitable solution should not be restricted to objective and scientifically rigorous principles, but should rather be context sensitive, communicative, open, and flexible (Hirsch et al. 2010, Knight et al. 2011).

A problem is that identifying a suitable solution does not, in practice, mean satisfying all interests. Instead, some interests are given greater weight at the expense of others. This mechanism refers to tradeoff, a process that involves loss and gain. There are issues associated with tradeoff. For example, not all losses and gains are tradable (Rittel & Webber 1972). The process of balancing loss and gain can be difficult when it involves answering questions like "who loses or who wins?" (Lackey 2006), and "whose values count?" (Schmid 2002, Yates 2003 p 141, Roe & Walpole 2010).

Even though tradeoff presents issues and challenges in planning, it is a mechanism that planners have to embrace in response to the limitations associated with the planning environment (Jentoft 2007). These limitations also bring in hard decisions (HDs) that require a planner or a policy maker to choose between incompatible or incommensurable, but perhaps equally important, options. Conflicts typically engender HDs, and some suggest that conflicts that are not addressed in planning tend to surface and plague management decisions (Lewis 2007, Muthiga 2009). One reason is that when tradeoffs are not made explicit, conflicts may be camouflaged and honest negotiation, especially involving hard, decisions suppressed (McShane et al. 2011). Another reason is that a less explicit tradeoff prevents reflective thought on the appropriate approach (Roe & Walpole 2010).

Consequently, it has been suggested that the acknowledgment of tradeoffs and HDs could propel genuine reflection, honest communication, and responsible action (Hirsch et al. 2010).

The concept of tradeoff in a multi-objective problem setting is not new in conservation planning methods. The term spatial prioritization itself connotes tradeoff, a common theme in all systematic conservation planning (SCP) approaches such as scoring, heuristics, mathematical programming, and multi-criteria methods. However, these methods handle tradeoffs in various ways.

For example, in heuristics and mathematical programming, biological targets are given precedence by explicitly targeting and achieving the competing biological priorities. In contrast, socioeconomic interests are typically reduced to a single parameter to constrain the biological targets (Moilanen et al. 2009). This type of tradeoff mechanism suggests that tradeoffs among socioeconomic interests are less important than biological criteria in systematic conservation planning. On the contrary, multi-criteria methods recognize that tradeoffs may exist, not only among biological priorities, but also among socioeconomic interests.

The multi-objective nature of planning acknowledges that multiple stakeholders may have varied interests. Subsequently, balancing these interests, through loss and gain, cannot be achieved without taking into account social values (Keeney 1992, Cowling & Pressey 2003). A classic conservation question such as "how much is enough" can be difficult to answer without considering the role of values in addition to scientific bases (Wilhere 2007). One reason is that the distribution of socioeconomic losses cannot be entirely objective (Rittel & Weber 1972, Connor 2002, Singleton 2009). In fact, the stakeholders' "buy-in" is often not based on objective scientific principles, but on subjective social values such as openness, empathy, and inclusiveness (Knight et al. 2011).

It is for these reasons that method development should also consider societal values (Theobald et al. 2000). To date, stakeholders' priorities, particularly for socioeconomic interests, are loosely integrated with planning methods (Ban & Klein 2009). It should however be noted that Marxan with Zones, a decision-support tool, allows a more complex integration of socio-economic criteria (Watts et al. 2009). While emerging tools are designed to make tradeoffs spatially explicit, there is, however, still a lot of work required in visualizing and understanding tradeoffs among competing socioeconomic groups (Adams et al. 2010).

1.2 RESEARCH QUESTIONS

This thesis attempts to answer the following questions to address the existing research gap on how data on competing interests can be integrated systematically along with biological data with a view that this can make conservation decisions more explicit.

1. What are the theoretical and practical bases for encouraging different types of methods in conservation planning and on what grounds do multi-criteria methods deserve exploration in spatial planning?

2. Is it possible to identify competing groups from available datasets and can detailed data on fishing activities help assess competing socioeconomic interests?
3. What are alternative methods for generating spatial socioeconomic criteria and biological data in spatial marine conservation planning?
4. How can tradeoffs among socioeconomic competing interests be integrated systematically and as explicitly as the biological competing interests? Will this systematic integration allow for:

    A. Making various levels of tradeoffs and HDs spatially explicit?
    B. Constraining and comparing regions of interest (ROIs) using a conservation scenario?
    C. Visualizing stakeholders' priorities concerning competing interests?
    D. Making competing socioeconomic groups aware of impacts?

1.3 GOALS AND OBJECTIVES

The study postulates that using a method that makes conservation losses and gains spatially explicit can help identify a range of conservation alternatives and the HDs required reaching conservation targets. Hence, the goal of this thesis is to develop a decision-support method for conservation planning that integrates and communicates conservation decisions such as those involving tradeoffs and HDs.

Subsequently, it leads to developing an evidence-based and stakeholder-driven tradeoff decision-support method whereby tradeoffs are made spatially explicit among and between competing biological criteria and socioeconomic interests. Achieving this goal is based on the following specific objectives:

1. To present conceptual illustration and insights into the challenges involved in making conservation decision such as those involving tradeoffs and HDs through a spatial perspective of conservation loss and gain.
2. To explore alternative GIS-based methods to capture biological and socioeconomic interests based on long term region-wide datasets.
3. To propose a new conservation decision-support method for making conservation tradeoffs spatially explicit that supports a systematic integration of social criteria in the conservation planning method.
4. To explore the utility of the proposed method and demonstrate evidence which makes tradeoffs and HDs spatially explicit among and between competing biological and socioeconomic interests.

1.4 STF-OWA: PROPOSED CONSERVATION DECISION-SUPPORT METHOD

This thesis proposes a new decision-support method named, *Spatial Tier Framework-Ordered Weighted Averaging (STF-OWA)* that is based on the concept of tradeoffs where losses and gains are weighed (or traded off) among and between them. As discussed in the remaining chapters, this study measures loss such as forgone fishing opportunities, as cost. In contrast, conservation benefits, such as biodiversity protection, are referred to as gains. The type of weighing scheme available in the STF-OWA has two potential advantages: (1) just like the balance scale, it offers several degrees of tilt (levels of tradeoffs) including extremes (that can mean lack of tradeoff) and (2) this type of weighing scheme can also be applied not only to differing biological interests but also to competing social interests.

This proposed conservation decision-support method is inspired by goal hierarchy and multi-criteria methods. Goal hierarchy allows structuring and unpacking higher-level objectives into lower-level objectives in a multi-objective decision-making environment (Keeney & Raiffa 1976). Higher-level objectives refer to broad overall objectives that may provide fewer details as to the specific actions that need to be taken. For this reason, a lower-level objective (i.e. a more detailed objective) needs to be identified.

A simple illustration of such hierarchy is provided by Keeney & Raiffa (1976, p 32). They suggested, for example, that improving the well-being of the city residents threatened by pollution can be thought about as a higher-level objective while the reduction of pollutants emissions from sources within the city can be considered as a lower-level objective. This lower-level objective can be further broken down into the reduction of nitrogen oxide emissions. It should be noted that nitrogen oxide emissions can be measured by using an attribute (e.g. nitrogen dioxide emitted per annum) that can contribute to the higher-level objective.

Similarly, a spatial tier framework (STF) is built around hierarchies of competing objectives. The difference that the STF offers is that objectives and attributes are captured using GIS-based maps, thus being spatially explicit. Also, the STF attempts to structure conservation objectives based on two fundamental competing categories, namely, loss and gains associated with conservation assessment. Therefore, the STF allows the competing objectives that might take the form of either a loss or a gain to be considered and captured in a spatially explicit manner.

In contrast with the STF that attempts to unpack the high-level objectives into low-level objectives and attributes, a multi-criteria method called ordered weighted averaging (OWA) quantitatively aggregates low-level objectives to achieve high-level objectives. OWA combines three types of decision strategy, namely, compensatory, very strict, and very liberal (Jiang & Eastman 2000). The compensatory decision is implemented by a multi-criteria procedure called weighted linear combination (WLC) that allows compensation or substitutability among losses and gains. Very strict and very liberal decision strategies are implemented respectively using the Boolean logical operations AND and OR. In GIS, these are respectively known as the intersection and union operations.

A very strict type of decision strategy requires all objectives to be satisfied in the solution set. Conversely, a very liberal type of decision requires that at least one of the objectives is satisfied in the solution set. For example, to choose a business location, three objectives might be considered: close proximity to an urban center, large parking site availability, and high accessibility to commuters. A very strict decision strategy makes sure that these three objectives are satisfactorily met in the chosen option while a liberal type of decision strategy might consider an option with just one of the objectives satisfied.

OWA combines these Boolean and WLC operations, resulting in the possibility of considering decision alternatives along this spectrum of very strict and very liberal decisions. In the context of decision-making, these three types of decisions can also correspond to decision-makers' attitudes. These include: risk-averse, risk-taking, and neutral attitudes to respectively complement the very strict decision, very liberal type of decision, and a compensatory decision.

To illustrate this, consider two competing sets of objectives represented by circles where bigger circles represent bigger objectives (Figure 1.1). The green shades represent the outcomes of decisions where a very strict decision generated a minimum outcome in contrast with the maximum outcome generated by a liberal type of decision, and an outcome in between these extremes results from a compensatory decision. We should note that varying levels of decisions and attitudes can exist between the two extreme types of decisions of very strict and liberal, thus offering a continuum of decisions, a mechanism that is captured in OWA (Figure 1.1).

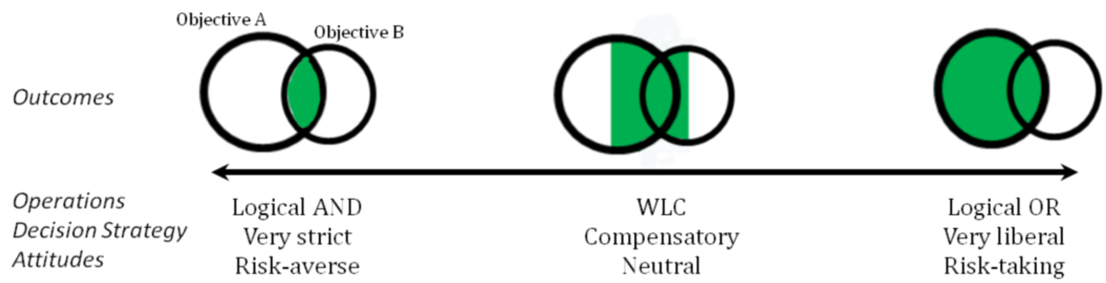


Figure 1.1Three types of decisions and attitudes that can be captured by the three operations combined in OWA. Circles represent competing objectives (e.g. biodiversity and socioeconomic objectives) with the bigger circle representing a bigger objective. Green shades represent decision outcomes where Logical AND tends to generate a minimum outcome in contrast with the maximum outcome from Logical OR, while the WLC generates an outcome in between these two extremes.

In the context of conservation planning, this study explores and understands how a combination of STF and OWA can serve as a conservation decision-support method. Specifically, this study investigates how various conservation decisions, attitudes, and outcomes can be made spatially explicit in identifying conservation areas. Also, it would be interesting to examine what these attitudes, decisions, and outcomes would indicate in identifying areas for conservation such as the MPAs.

1.5 RESEARCH STAGES

This project had five major stages (Figure 1.2). First, a comprehensive literature review allowed identification of research gaps, in terms of method development in conservation planning. It also allowed this study to see conservation planning from an image of a "funnel" where broad planning theories are filtered down to its narrow methodological and technical aspects of conservation assessment methods. Subsequently, it facilitated an understanding of the complexities around which the identified research gaps exist and simultaneously pointed this study to pursue an alternative model and approach.

Second, a conservation planning model STF-OWA was conceptualized through a combination of OWA, a multi-criteria decision analysis (MCDA) technique, and a tier based approach to unpack spatial data (i.e. STF).

Third, various datasets for a study area in Eastern Canada were gathered from public and private agencies. Fourth, datasets were processed and analyzed using GIS-based techniques. Some new alternative GIS-based techniques, tailored to the available data, were introduced in this study. Then, the data were organized into three broad categories of GIS-based maps: biological, socioeconomic, and other marine uses.

Finally, the utility of the STF-OWA as a decision-support method for making tradeoffs spatially explicit was explored and demonstrated through a set of priorities obtained from a workshop composed of participants from academic institutions, government agencies, and environmental non-government organizations (ENGOs).

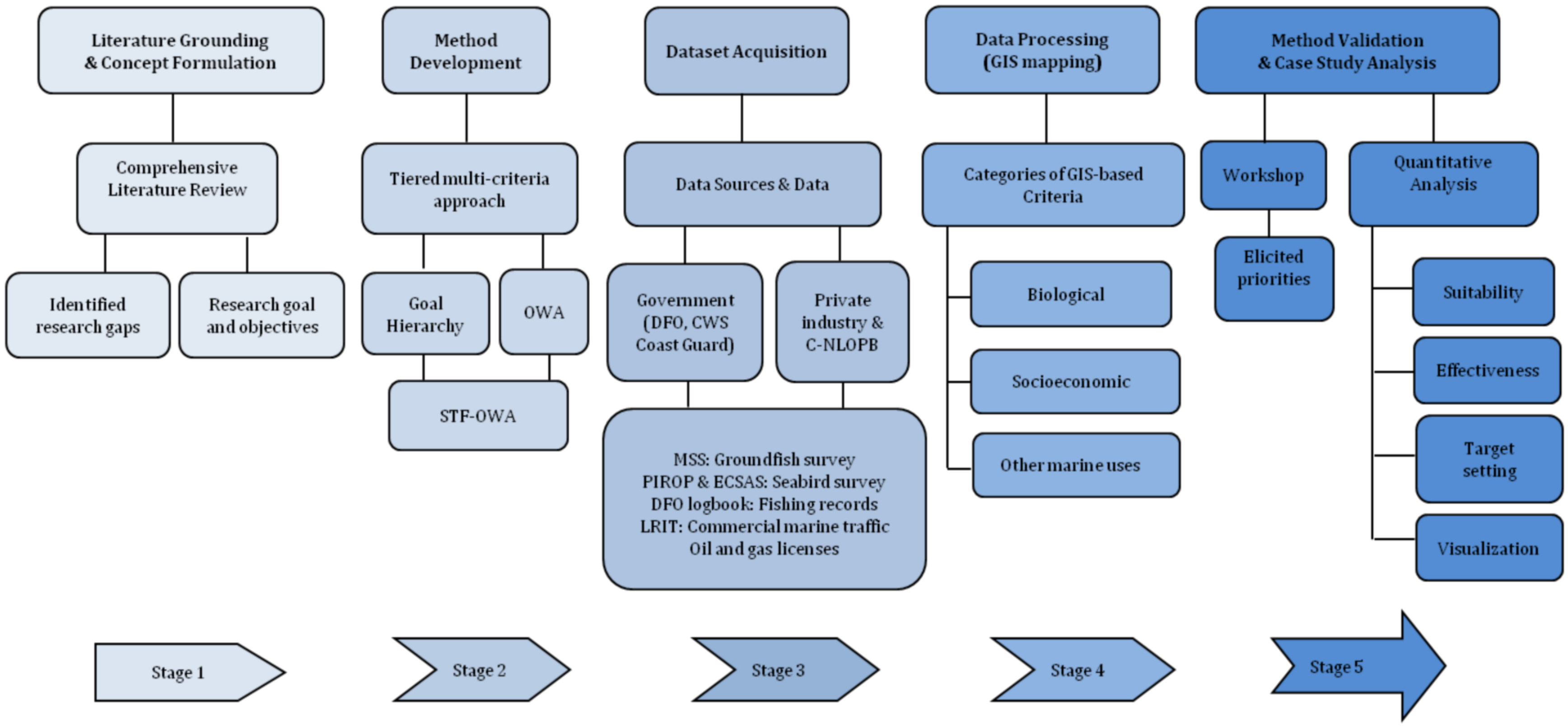


Figure 1.2 Phases of project development started from understanding the relevant theories and approaches in conservation planning and ended in developing an alternative conservation decision-support method. Results from these stages were addressed in the following chapters: Stage 1 in Chapter 1; Stages 3 and 4 in Chapter 3; and Stages 2 and 5 in Chapter 4.

1.6 THESIS OUTLINE

Chapter 2 presents a review of related literature in conservation. The presentation of this review is not as a standalone publishable contribution focused on a single, specific subject. Rather it covers broad subject reflecting the candidate's exposure to literature that ultimately influences the direction of this study. Hence, it starts broad by discussing the challenges that planning theories present to planners. Next, while still a broad subject, it pays attention to conservation planning approaches, highlighting their strengths and weaknesses. Then, it focuses on specific challenges associated with MPA planning, that is, decisions that involve tradeoffs (and lack thereof). Subsequently, this thesis proposes the importance of making conservation tradeoffs (and HDs) explicit especially from a spatial planning perspective. The candidate intends to develop this last section of the chapter into a standalone publication.

Chapter 3 presents the data assembled for the Newfoundland and Labrador continental shelf and slope case study, their sources, and the methods for generating the multiple biological and socioeconomic GIS-based attribute maps. These maps represent the different biological and socioeconomic interests based on the data that were made available for the project. The study area was selected due to the relatively rich sets of information available for this region.

Chapter 4 presents the design and use of the STF-OWA as a decision-support method. This chapter details how goal hierarchy can be tailored for structuring marine conservation spatial datasets. It also discusses the integration of value-laden conservation priorities through the technique offered by OWA. This chapter then describes the details of the workshop by which a set of conservation priorities was obtained. Finally, this chapter demonstrates the utility of the STF-OWA in making tradeoffs and HDs spatially explicit.

Chapter 5 concludes the thesis. It presents the summary of the findings, highlights and contributions of the study, how conservation tradeoffs and HDs are viewed in the STF-OWA, and recommendations for MPA planning. It also discusses the limitations associated with our conceptual assertions as well as the limitations of datasets that can be investigated in future work. In addition, the conclusions chapter mentions potential directions of the STF-OWA, particularly its further development and implementation as a decision support tool.

1.7 LITERATURE CITED

Adams, W. 2010. Path dependence in conservation. Pages 292-310 in N. Leader-Williams, W. Adams, and R. Smith, editors. Tradeoffs in conservation: Deciding what to save. Wiley-Blackwell, Oxford, UK.

Adams, V., B. Pressey, and R. Naidoo. 2010. Opportunity costs: Who really pays for conservation? Biological Conservation **143**:439–448.

Allison, G., J. Lubchenco, and M. Carr. 1998. Marine reserves are necessary but not sufficient for marine conservation. Ecological Applications **8**:79–92.

Ball, I. and H. Possingham. 2000. MARXAN (V1.8.2): Marine reserve design using spatially explicit annealing, a manual prepared for the Great Barrier Reef Marine Park Authority. Available from http://www.uq.edu.au/marxan/docs/marxan_manual_1_8_2.pdf (accessed January 2015).

Ban, N. and C. Klein. 2009. Spatial socioeconomic data as a cost in systematic marine conservation planning. Conservation Letters **2**:206-215.

Beverton, R. and S. Holt. 1957. On the dynamics of exploited fish populations. Fishery Investigations Series II Volume XIX, Ministry of Agriculture, Fisheries and Food. [Reprinted 1993. Chapman and Hall, London, UK].

Brown, K., W. Adger, E. Tompkins, P. Bacon, D. Shim, and K. Young. 2001. Trade-off analysis for marine protected area management. Ecological Economics **37**:417–434.

Carwardine, J., W. Rochester, K. Richardson, K. Williams, R. Pressey, and H. Possingham. 2007. Conservation planning with irreplaceability: Does the method matter? Biodiversity and Conservation **16**:245-258.

Christensen, V. 2000. Biomass distributions for high trophic-level fishes in the North Atlantic, 1900-2000. Available from http://www.seaaroundus.org/flash/NorthAtlanticTrends.htm (accessed January 2015).

Connor, M. 2002. Social cost and sustainability. Pages 181-202 in D. Bromley and J. Paavola, editors. Economics, ethics, and environmental policy: Contested choices. Blackwell Publishing, Oxford, UK.

Cowling, R. and R. Pressey. 2003. Introduction to systematic conservation planning in the Cape Floristic Region. Biological Conservation **112**:1–14.

Crowder, L. and E. Norse. 2005. The greatest threat: Fisheries. Pages 183-184 in A. Elliot and L. Crowder, editors. Marine conservation biology: The science of maintaining the seas biodiversity. Island Press, Washington.

Edgar, G., et al. 2014. Global conservation outcomes depend on marine protected areas with five key features. Nature **506**: 216-220.

Eisma, R., P. Christie, and M. Hershman. 2005. Legal issues affecting sustainability of integrated coastal management in the Philippines. Ocean & Coastal Management **48**:336-359.

Ellis, E., D. Fuller, J.Kaplan, and W Lutters. 2013. Dating the Anthropocene: Towards an empirical global history of human transformation of the terrestrial biosphere. Elementa: Science of the Anthropocene **1**: 000018.

FAO 2011. Review of the state of world marine fishery resources. FAO Fisheries and Aquaculture Technical Paper No. 569. FAO, Rome, 334 pp.

Halpern, B., K. McLeod, A. Rosenberg, and L. Crowder. 2008. Managing for cumulative impacts in ecosystem-based management through ocean zoning. Ocean and Coastal Management **51**:203-211.

Hirsch, P., W. Adams, P. Brosius, A. Zia, N. Bariola, and J. Dammert. 2011. Acknowledging conservation trade-offs and embracing complexity. Conservation Biology **25**:259-264.

Hutchings, J. and R. Myers. 1994**.** What can be learned from the Collapse of a renewable resource? Atlantic Cod, *Gadus morhua*, of Newfoundland and Labrador. Canadian Journal of Fisheries and Aquatic Resources **51**:2126-2146.

ICO (International Conference on Overfishing). 1946. International Conference on the Problem of Overfishing. Nature **157**:403-403.

IUCN. 1994. Guidelines for Protected Area Management Categories. IUCN, Cambridge, UK and Gland, Switzerland.

James, A., K. Gaston, and A. Balmford. 2001. Can we afford to conserve biodiversity? BioScience **51**:43-52.

Jentoft, S. 2007. Limits of governability: Institutional implications for fisheries and coastal governance. Marine Policy **31**:360-370.

Jiang, H. and J. Eastman. 2000. Application of fuzzy measures in multi-criteria evaluation in GIS. International Journal of Geographical Information Science **14**:173-184.

Justus, J. and S. Sarkar. 2002. The principle of complementarity in the design of reserve networks to conserve biodiversity: A preliminary history. Bioscience **27**:421-435.

Kahneman, D., and A. Tversky. 1979. Prospect theory: An analysis of decision under risk. Econometrica: Journal of the Econometric Society **47**:263-291.

Keeney, R. 1992. Value Focused Thinking. Harvard University Press, Massachusetts.

Keeney, R. and H. Raiffa. 1976. Decisions with multiple objectives: Preferences and value tradeoffs. Wiley, New York.

Knight, A., R. Cowling, A. Boshoff, S. Wilson, and S. Pierce. 2011. Walking in STEP: Lessons for linking spatial prioritizations to implementation strategies. Biological Conservation **144**:202-211.

Lackey, R. 2006. Axioms of ecological policy. Fisheries **31**:286-290. [Reprinted Renewable Resources Journal **24**:13-16].

Lewis, D. 2007. Opportunities and constraints for protected area management through increased connectivity to local livelihood needs in surrounding border areas: Lessons from Luangwa Valley, Zambia. Pages 38-49 in K. Redford, and E. Fearn, editors. Protected areas and human livelihoods. Working Paper No. 32. Wildlife Conservation Society, New York.

Lubchenco, J., S. Palumbi, S. Gaines, and S. Andelman. 2003. Plugging a hole in the ocean: Emerging science of marine reserves. Ecological Applications **13**:S3-S7.

Macintosh, A., T. Bonyhady, and D. Wilkinson. 2010. Dealing with interests displaced by marine protected areas: A case study on the Great Barrier Reef Marine Park Structural Adjustment Package. Ocean and Coastal Management **53**:581-588.

Margules C. and M. Usher. 1981. Criteria used in assessing wildlife conservation potential: A review. Biological Conservation **21**:79-109.

Mccall, M. and C. Dunn. 2012. Geo-information tools for participatory spatial planning: Fulfilling the criteria for 'good' governance? Geoforum **43**:81-94.

McShane, T., et al. 2011. Hard choices: Making trade-offs between biodiversity conservation and human well-being. Biological Conservation **144**:966-972.

Meir, E., S. Andelman, and H. Possingham. 2004. Does conservation planning matter in a dynamic and uncertain world? Ecology Letters **7**:615-622.

Myers, R., N. Barrowman, M. Hoening, and Z. Qu. 1996. The collapse of cod in eastern Canada: The evidence from tagging data. ICES Journal of Marine Science **53**:629-640.

Mitsova, D., F. Wissinger, A. Esnard, R. Shankar, and P. Gies. 2013. A collaborative geospatial shoreline inventory tool to guide coastal development and habitat conservation. ISPRS International Journal of Geo-Information **2**:385-404.

Moilanen A., K. Wilson, and H. Possingham. 2009. Spatial Conservation Prioritization: Quantitative methods and computational tools. Oxford University Press, New York.

Murray, S., et al. 1999. No take reserve networks: Sustaining fishery populations and marine ecosystems. Fisheries **24**:11–25.

Muthiga, N .2007. The evolution of management and impacts on communities adjacent to the Mombasa marine protected area, Kenya. Pages 27-37 in K. Redford and E. Fearn, editors. Protected Areas and human livelihoods, Working Paper No. 32. Wildlife Conservation Society, New York.

Muthiga, N. 2009. Evaluating the effectiveness of management of the Malindi-Watamu marine protected area complex in Kenya. Ocean and Coastal Management **52**:417-423.

Myers, R., and B. Worm. 2003. Rapid worldwide depletion of predatory fish communities. Nature **423**:280-283.

Norse, E., and L. Crowder. 2005. Why marine conservation biology? Pages 1-30 in A. Elliot and L. Crowder, editors. Marine conservation biology: The science of maintaining the seas biodiversity. Island Press, Washington.

Oracion, E., M. Miller, and P. Christie. 2005. Marine protected areas for whom? Fisheries, tourism, and solidarity in a Philippine community. Ocean and Coastal Management **48**:393-410.

Pauly, D. 1995. Anecdotes and the shifting baseline syndrome of Fisheries. Trends in Ecology and Evolution **10**:430.

Pauly, D., R. Hilborn, and T. Branch. 2013. Fisheries: Does catch reflect abundance? Nature **494**:303-306.

Pressey R. and A. Nicholls. 1989. Efficiency in conservation evaluation: Scoring versus iterative approaches. Biological Conservation **50**:199-218.

Preikshot D. and D. Pauly. 2005. Global fisheries and marine conservation: Is co-existence possible. Pages 185-197 in N. Leader-Williams, W. Adams, and R. Smith, editors. Tradeoffs in conservation: Deciding what to save. Wiley-Blackwell, Oxford.

Pulgar-Vidal, M., B. Monteferri, and J. Dammert. 2010. Tradeoffs between conservation and extractive industries. Pages 223-252 in N. Leader-Williams, W. Adams and R. Smith, editors. Trade-offs in conservation: Deciding what to save. Wiley-Blackwell, Oxford, UK.

Rittel, H. and M. Webber. 1973. Dilemmas in a general theory of planning. Policy Sciences **4:**155-169.

Roberts, C. 2007. The unnatural history of the sea. Island Press, Washington, D.C., USA.

Roberts, C. and J. Hawkins. 1999. Extinction Risk in the sea. Trends in Ecology and Evolution **14**:241-246.

Roberts, C. and N. Polunin. 1993. Marine Reserves: Simple solutions to managing complex fisheries? AMBIO: A Journal of the Human Environment **22**:363-368.

Roe, D. and M. Walpole. 2010. Whose Value Counts? Tradeoffs between biodiversity conservation and poverty reduction. Pages 157-174 in N. Leader-Williams, W. Adams and R. Smith, editors. Trade-offs in conservation: Deciding what to save. Wiley-Blackwell, Oxford, UK.

Rose, G. 2007. Cod: The ecological history of the North Atlantic Fisheries. Breakwater Books Ltd., St. John's, NL, Canada.

Rudd, M., A. Danylchuk, S. Gore, and M. Tupper. 2001. Are marine protected areas in the Turks and Caicos Islands ecologically or economically valuable? Fisheries Center Research Reports **9**:198–211.

Salafsky, N. 2011. Integrating development with conservation: A means to a conservation end, or a mean end to conservation? Biological Conservation **144:**973-978.

Silva, P. 2006. Exploring the linkages between poverty, marine protected area management, and the use of destructive fishing gear in Tanzania. World Bank

Policy Research Working Paper No. 3831. Available from https://openknowledge.worldbank.org/handle/10986/8779 (accessed January 2015).

Schmid, A. 2002. All environmental policy instruments require a moral choice as to whose interests count. Pages 133-147 in D. Bromley and J. Paavola, editors. Economics, ethics, and environmental policy: Contested choices. Blackwell Publishing, Oxford, UK.

Singleton, S. 2009. Native people and planning for marine protected areas: How "stakeholder" processes fail to address conflicts in complex, real-world environment. Coastal Management **37**:421-440.

Theobald, D., N. Hobbs, T. Bearly, J. Zack, T. Shenk, and W. Riebsame. 2000. Incorporating biological information in local land-use decision making: Designing a system for conservation planning. Landscape Ecology **15**:35-45.

von Neumann, J. and O. Morgenstern. 1947. Theory of Games and Economic Behavior Princeton University Press. Princeton, New Jersey.

Watts, M., I. Ball, R. Stewart, C. Klein, K. Wilson, C. Steinback, R. Lourival, L. Kircher, and H. Possingham. 2009. Marxan with Zones: software for optimal conservation based land-and sea-use zoning. Environmental Modelling & Software, **24**: 1513-1521.

Wilhere, G. 2007. The how-much-is-enough myth. Conservation Biology **22**:514-517.

Williams, J., C. ReVelle, and S. Levin. 2004. Using mathematical optimization models to design nature reserves. Frontiers in Ecology and the Environment **2**:98-105.

Willis, T. 2003. Comment: Burdens of evidence and the benefits of marine reserves: Putting Descartes before des horse? Environmental Conservation **30**:97-103.

Yager, R. 1996. Quantifier guided aggregation using OWA operators. International Journal of Intelligent Systems **11**:49-73.

Yates, F. 2003. Decision management: How to assure better decisions in your company. 1st edition. John Wiley & Sons, Inc., California.

CHAPTER 2 PLANNING FOR CONSERVATION: CURRENT PRACTICES AND NEW DIRECTIONS

2.1 INTRODUCTION

It is estimated that 41% of the ocean ecosystems are significantly impacted by human activities (Halpern et al. 2008). These human activities include waste disposal, mineral extraction, and fishing (Roberts 2002, Glover & Smith 2003, Davies et al. 2007). In adding to the direct impact on ecosystems, this growing human footprint is also expected to alter the biogeochemistry of the oceans and subsequently the marine biotic community (Doney 2010).

On a global level evidence suggests a decline of predatory fish by about two thirds over the last 100 years (Christensen et al. 2014). In the Pacific, Atlantic, and Indian oceans, pelagic predatory fish have declined approximately 90% due to industrial fishing (Myers & Worm 2003). Species extinctions on land (Ehrlich & Ehrlich 1981, Myers 1984) resulted in the development of conservation biology as a "crisis discipline" (Soulé 1985). Approximately a decade later Roberts & Hawkins (1999) documented extinctions and declines of marine species. This issue spurred similar interest in the development of marine conservation biology, a discipline dedicated to protecting the biodiversity of the oceans (Norse & Crowder 2005). Like terrestrial protected areas, marine protected areas (MPAs) and other spatial management tools are suggested as potential conservation tools to help protect marine biodiversity (Crowder & Norse 2005).

Soulé (1985) described conservation biology as a discipline concerned with applying science to conservation problems and providing principles and tools for biodiversity preservation. He mentioned that aside from scientific principles, ethical norms are a genuine aspect of conservation science. In time, the relevance of ethical norms became more apparent in marine conservation planning.

Recent reviews of conservation planning research and practices for over three decades show that protected areas (PAs) are multidimensional in nature (Gillman et al. 2011, Thorpe et al. 2011). As such, conservation planning tends to use divergent approaches. Science-based approaches focus on framing conservation problems based on scientific, quantitative, repeatable, and objective means of satisfying biodiversity goals (Williams et al. 2004, Leslie et al. 2003). In contrast, social-based approaches frame conservation problems based on subjective, value-based, and qualitative approaches to satisfy goals such as social acceptability and effectiveness of PA implementation (Brown et al. 2001, Knight & Cowling 2007). Additionally, some approaches attempt to frame conservation problems based on both conceptual bases of scientific and social-based approaches (Moffett & Sarkar 2006, Wood & Dragicevic 2007).

The presentation of this review is not as a standalone publishable contribution focused on a single, specific subject. Rather it covers broad subject reflecting the candidate's exposure to literature that ultimately influences the direction of this study. It begins, in Section 2.2, with the review of relevant planning theories, suggesting that the combination of "wicked" problems and the inability of decision-makers to find all possible alternatives results in planning dilemmas. This is particularly true under a planning framework that is based on goal similar to the type of framework explored in this research.

Section 2.3 discusses the similarities as well as the differences between planning and decision-making models. Section 2.4 presents the multiple dimensions of PAs that tend to compete with one another at the planning stage. This section presents the forms and examples of measurements that help express the PA dimensions in operational terms. This dissertation uses some of these measurements to generate the attributes presented in Chapter 3, and shows, in Chapter 4, how these attributes represent the definition and achievement of conservation objectives (i.e. biodiversity and socioeconomic objectives).

Section 2.5 discusses and categorizes various methods in conservation planning and proposes that these various approaches need to be encouraged as a way of dealing with "wicked" problems in conservation planning (Rittel & Webber 1973). It presents multi-criteria decision analysis (MCDA) as one of the categories of conservation planning methods that this thesis explores in Chapter 4.

Section 2.6 presents MPA planning dilemmas in the context of a goal-based planning framework that is fundamental to this chapter's conceptual illustration of tradeoffs and hard decisions (HDs) in the context of spatial conflict of conservation loss and gain. Finally, in Section 2.7 of this chapter discusses the challenge to making tradeoffs and HDs explicit, a core motivation for this study in designing a new decision-support method in MPA planning.

2.2 BOUNDED RATIONALITY AND WICKED PROBLEMS CREATE PLANNING DILEMMAS

The concept of bounded rationality (Simon 1991) suggests that humans lack the cognitive capacity to identify and evaluate all possible choices; hence, humans are bounded decision-makers. This causes problems, especially in the context of decision-making, where setting goals is an essential component of the process (Adler 2008). A goal-based planning framework often starts from identifying the problem, typically an undesired or less desired "initial state" or situation that calls for change into something more desirable in the future (Simon & Newell 1972, Huber 1986, p 110). The particular desirable future eventually becomes the basis for goals (Grunig & Khun 2006, p 17). Narrowing the gap between the undesired and desired future typically requires a procedure or method that can help evaluate choices. But, as previously noted, bounded decision-makers have difficulty in identifying and evaluating all possible choices. Rittel & Webber (1973) explained this problem further, and referred to it as a planning dilemma, under the notion of "wicked" problems.

Framed around goals, public or social-based planning, according to Rittel & Webber (1973), faces dilemmas when confronted with "wicked" problems. They argue that, as opposed to tame or benign problems such as those often addressed by scientific formulation, "wicked" problems are malignant or vicious. Rittel & Webber (1973) offer several propositions of "wicked" problems that make planning difficult. Key to their propositions is the notion that it is nearly impossible to pinpoint "the problem" or agree on what it is and what causes it. This suggests two planning dilemmas: (1) identification of the solution; and (2) and the challenge of determining "the definitive procedure" that narrows down the gap between the undesired and desired future. This chapter discusses the challenges that planning dilemmas present to bounded conservation planners in terms of designing conservation methods and in making conservation decisions.

2.3 PLANNING IS MOSTLY ABOUT DECISION-MAKING

In general decisions are a choice between options while the act of decision-making is a procedure followed for selecting particular options over others (Grunig

& Khun 2009). The process of planning is about how and what to do to achieve the desired goal. An important component of this process is decision-making. As a result, planning follows a decision-making structure.

Table 2.1 summarizes the process steps typically taken in a prescriptive type of decision-making model (e.g. structured decision-making), comparing it to systematic conservation planning steps. In effect, planning steps provide more details as to how decision-making steps can become operational. Due to the similarities between decision-making and planning, this thesis uses these terms interchangeably.

Table 2.1 Comparison of process steps between structured decision-making (Wintle et al. 2011) and systematic conservation planning (Pressey & Botrill 2008)

Structured Decision-making Steps	Systematic Conservation Planning Steps
Problem formulation	• Scoping and costing the planning process • Identifying and involving stakeholders • Identifying the context for conservation areas
Setting goals	• Identifying conservation goals • Setting conservation targets • Determining measurable objectives
Identification of management strategies	• Identifying management alternatives
Decision Modeling (e.g. Linear programming)	• Collecting data on biodiversity and other natural features • Reviewing target achievement in existing conservation areas • Selecting additional conservation areas • Tradeoffs, allocate resources
Outcome evaluation	• Maintaining and monitoring established conservation areas

Models in decision-making and planning exhibit similarities. The prescriptive type of decision-making model and the systematic type of planning model both emphasize how decisions *should* be made. The former typically prescribes how options ought to be selected (von Neumann & Morgenstern 1944), and the latter imparts how planning is or ought to be (Innes 1995). Also, a descriptive type of decision-making model and a communicative action type of planning model exhibit similarities. The former seeks to provide observations on how people *actually* make decisions (Kahneman & Tversky 1979), and the latter model outlines "what planning is by finding out what planners do" (Innes 1995, p 184).

It should be noted that while these planning and decision-making models show similarities, they are inconsistent on two accounts. First, the prescriptive and systematic model emphasizes the value of outcome (or utility) such as the final wealth while the descriptive and communicative model focuses on the value of losses and gains (i.e. deviations from current wealth). In the latter, both judgment and biases play important roles. Second, the prescriptive model views outcome to exhibit linear probabilities while the descriptive model views the probabilities associated with judgment and biases as non-linear. Therefore, in contrast with prescriptive and systematic models that consider technical and objective type of decision-making processes, the descriptive and communicative action model suggests that planners need to pay "attention to the messy part of planning that does not fit into a systematic framework" (Innes 1995, p 184).

Consequently, the communicative action model includes the use of qualitative and interpretive inquiry including: social learning (Friedmann 1987, Knight et al. 2006), negotiation and consensus building (Ury & Fisher 1981, Susskind & Cruikshank 1987), intergovernmental relations (Christensen 1985, Jentoft et al. 2007), and institutional analysis (Healey et al. 1988, Fanning et al. 2007, Grilo 2011). In contrast, the systematic models pay attention to quantitative, rule-based, repeatable, and objective methods such as mathematical models (Grunig & Khun 2009).

As later discussed, these contrasting models reflect the diverging design of conservation planning approaches, a manifestation of ways bounded planners deal with "wicked" planning problems.

2.4 CONSERVATION PLANNING FACES COMPETING DIMENSIONS

The wicked nature of conservation planning problems in the context of protected areas (PAs) is related to three main dimensions discussed in Thorpe et al (2011): biological/ecological, socioeconomic, and governance/management. Thorpe et al. (2011) discuss the intrinsic and extrinsic benefits of protected areas under the biological-ecological considerations. Intrinsic benefits refer to the positive effects of protected areas (e.g. reduced mortality rates of species, and reduced habitat damage) to the population and community dynamics, and habitats within the protected area, such as within MPA boundary (Bohnsack 1998, Boersma & Parrish 1999). Extrinsic benefits refer to benefits outside protected areas boundaries (e.g. spill-over, recruitment subsidy) such as those cases of emigration of fish and export of pelagic eggs and larvae outside marine reserves as documented in Gell & Roberts (2003).

The socioeconomic dimension of protected areas includes benefits and costs (Thorpe et al. 2011). Examples of benefits are enhanced local economies, improvement of quality of life, and increased tourism related jobs (Leeworthy & Wiley 2003, Alcala 2004, Hind et al. 2010). In terms of costs, MPAs may cause loss of economic opportunities, displacement of fishers from their traditional fishing grounds (Sanchirico 2002) or loss of fisheries jobs (Oracion et al. 2005).

The governance-management dimension has two facets according to Thorpe et al. (2011): the internal governance structure of protected areas and its nesting in broader governance context. The former relates to the actual governing actions directed at the protected areas, for instance, monitoring of conservation sites and setting of acceptable resource-use rules (Ostrom 1990, IUCN-WCPA 2008). The latter refers to the fact that protected area governance cannot be separated from a broader political system where protected areas are embedded with other regulatory domains and various jurisdictional frameworks (White 2002, Crowder et al. 2006, Fanning et al. 2007, Jentoft et al. 2007).

In conservation planning, these various PA dimensions can easily develop conflicting situations. A priority to set aside space for biodiversity conservation, such as the no-take zones, may not be compatible with the priority on keeping fishers' jobs and other related fishing business. The priority of setting a budget for the operational expenses of PA management may conflict with the priority of increasing government revenue from extracting natural resources. A priority for provisioning compensation to disadvantaged stakeholders may compete directly with the government's priority of minimizing government expenditures. Conflicts such as these usually create tension, dissatisfaction (Badalamenti 2000, Lewis 2007, Ledee & Sutton 2011), lack of political will, and rejection of PAs (Fiske 1992, James et al. 1999).

Such conflicts are typically viewed under the notion of conservation tradeoffs and have become of great interest in protected area planning. Some have defined conservation tradeoffs as "getting the balance right" (Jones et al. 2011) or "a balancing of factors all of which are not attainable at the same time" (Webster 2010, also adopted in Leader-Williams et al. (2010). Viewing tradeoffs as a way to balance competing factors provides little insight into which alternative methods may be best pursued. However, it offers a springboard for specifying tradeoffs in more operational terms. These tradeoffs will be discussed in the following paragraphs.

The concept of tradeoff cuts across different fields. In multi-attribute utility theory, tradeoffs refer to ratios of relative contributions of factors in measuring decision alternatives (Keeney & Raiffa 1976, Lai & Hopkins 1989). In medical care, tradeoff goes under the term of triage which is a way to rank the urgency of clinical risks in the emergency department. Triage is necessary due to insufficient resources available to deal with all clinical risks at once (Mackaway-Jones et al. 2006). However, triage has undesirable consequences such as delay in providing medical attention, failure to provide the needed medical care, and even death of some patients (Aacharya et al. 2011).

The above definitions of tradeoffs resonate in conservation planning for two reasons. First, planners view the relative contributions of species as one way to prioritize conservation features (Marris 2007). Second, conservation actions are carried out in the face of an emergency situation where all relevant goals may not be given attention all at once (Soulé 1985, Botrill et al. 2008). In conservation practice, this tradeoff could mean conservation gain and socioeconomic loss at the same time (Lackey 2006, McShane et al. 2011).

Viewing conservation tradeoffs based on loss and gain, as defined in Kahneman & Tversky (1979), provides four advantages: (1) loss and gain offer specific details as to what to balance and can help gauge the reference point where balancing needs to be done, (2) loss and gain may encourage explicit treatment of desires that can/cannot be attained especially when they compete with one another, (3) loss and gain factors can be identified and, to some extent, measured and (4) loss and gain can integrate the role of values in making tradeoffs as explained in Section 2.6.

In mainstream research, tradeoffs between competing objectives are often carried out after expressing the PA dimensions in operational terms through some forms of measurements. Measurements can come in two forms: subjective and objective. Subjective measurements, which can be qualitative, quantitative or both, are applied to socioeconomic and governance/management dimensions. Subjective measurements can involve methods such as Delphi, consensus, and analytic hierarchy process (AHP). Objective measurements, typically quantitative, are extensively explored in academic research as particularly applied to biological dimension. Tradeoffs are explicitly or implicitly carried out when objectives are being defined, prioritized, or achieved in the solution set. Depending upon the type of measurement used, tradeoffs can have varying outcomes.

For measuring biodiversity, a number of measurements have been suggested. Three of the most common measurements are: species richness, species abundance, and species evenness. Combining measures of richness and evenness has also been suggested (Good 1953, Hurlbert 1971) for creating diversity indices. Two of the most popular measures are (1) the Simpson's diversity index that excludes any assumptions of species abundance (Simpson 1949), and (2) the Shannon-Wiener diversity index which considers the degree of evenness in species abundance (Pielou 1969). The concepts of rarity, endemism, and species endangerment have also been suggested to guide biodiversity conservation amidst threats, depletion, and rapid extinction of species, (Tubbs & Blackwood 1971, Gehlbach 1975, Ratcliffe 1977, Wright 1977, Salm et al. 2000, Langhammer et al. 2007, Edgar et al. 2008ab).

Though many biodiversity measures exist, understanding and identifying the most meaningful biodiversity measurement for conservation purposes remains a challenge (Purvis & Hector 2000, Di Minin & Moilanen 2012). In most cases, data availability determines which biodiversity measurement to use. For example, analyses of biodiversity in megadiverse countries or centers of biodiversity in terrestrial (McNeely et al. 1990, Mittermeier 1990, Myers et al. 2000), and marine (Roberts et al. 2002) regions are mostly based on species richness.

Nevertheless, the expanding breadth of conservation, which considers geographic space, results in numerous other conservation concepts that come with their own sets of measurements. These concepts include adequacy, comprehensiveness, representativeness, representation, complementarity, efficiency, effectiveness, and flexibility. Most of these concepts have developed or evolved along with specific conservation planning methods, as discussed in the section to follow. The reader is referred to Kukkala & Moilanen (2012) for a comprehensive review of these concepts.

2.5 CONSERVATION PLANNING APPROACHES: AN OVERVIEW

Many of the formal conservation assessment techniques are included under the umbrella of systematic conservation planning (SCP) (Margules & Pressey 2000, Margules & Sarkar 2007). SCP is a growing field of inquiry in conservation biology (Moilanen 2008, Kukkala & Moilanen 2012). In general, SCP is concerned with the process of prioritizing sites for conservation with two major components: setting goals using systematic and quantitative methods and the identification or delivery of actions to meet the goals, see Table 2.1 (Margules & Pressey 2000, Knight et al. 2006a, Pressey et al. 2007).

The first component typically relies on computer-based techniques and scientific concepts to identify quantifiable biodiversity targets and process spatially explicit data (Ando et al. 1998, Williams et al. 2004, Stewart & Possingham 2005, Pressey et al. 2007). Knight et al. (2006) calls this "systematic conservation assessment". The second component relates to conservation actions toward implementing the goals. Many of these conservation actions are generally considered as being socio-political in nature (Knight et al. 2006b, 2010, 2011a, Margules & Sarkar 2007). This second component often relates to the first component especially in setting or negotiating goals. Some cases on-the-ground show that a rigorous, quantitative means of setting of biodiversity targets are less favored in real-world planning making SCP approaches less relevant (Knight et al. 2011a, Game et al. 2011). As we shall see in Section 2.5.5, the PA dimensions, as summarized by Thorpe et al. (2011), and the SCP components compete in the design of conservation planning approaches.

These two components of SCP reflect the multiple facets of protected areas summarized by Thorpe et al. (2011). As we shall see in Section 2.5.5, the PA dimensions and the SCP components compete in the design of conservation planning approaches.

2.5.1 (INFORMED) OPPORTUNISM AND EXPERT-DRIVEN APPROACHES

SCP is typically contrasted with *ad hoc*/opportunistic or expert-driven planning. These two approaches are generally less systematic in the way they integrate biological criteria or quantitative biodiversity targets when prioritizing conservation areas (Margules & Pressey 2000, Roberts et al. 2003). Some have argued that these non-systematic approaches can lead to negative outcomes such as creating residual conservation areas that fail to reach larger conservation goals (Pressey & Tully 1994, Pressey 1994, Devillers et al. 2014)

2.5.1.1 (INFORMED) OPPORTUNISM

Ad hoc or opportunistic approaches are generally regarded as random political and organizational opportunism (Pressey 1994). *Ad hoc* approaches include two downsides (Pressey 1994, Pressey & Tully 1994). First, it can fail to protect species and ecosystems that urgently need the most protection. This is particularly true when *ad hoc* designations of conservation areas resulted from lack of communication among concerned departments, government support, and user conflicts (McNeill 1994). Second, it can be less efficient in protecting the overall regional biodiversity, as shown by empirical evidence provided by Pressey & Tully (1994) and Stewart et al. (2007), showing that priority sites identified through *ad hoc* approaches protect less biodiversity relative to area coverage.

It is worth noting, however, that an *informed* type of opportunism may offer additional value over simple opportunistic approaches (Roberts 2000, Noss et al. 2002, Knight & Cowling 2007). An informed opportunism can involve "mapping of conservation opportunities that assist in decision making that pertains to not only where conservation action is required, but also when and how to implement actions when opportunities appear" (Knight & Cowling 2007, p 1125). It places importance on the value of getting areas implemented and actually achieving some conservation goals and obtaining local support (Roberts 2000, Shears & Babcock 2003). One of the reasons for the push for informed opportunism is that in certain areas even high quality data can fail to identify local hotspots. The available data quality and quantity may also not always be sufficient for a rigorous quantitative modeling (Cowling et al. 2009). Another reason is that when the human and social variables dictate success of implementation, failure to get their support compromises conservation efforts altogether, something informed opportunism seeks to avoid (Game et al. 2011).

2.5.1.2 EXPERT-DRIVEN

Expert-driven planning processes convene experts with various perspectives on the issues at hand in a workshop or virtual conference. It can be a multi-stage process that seeks to secure a group consensus (Linstone and Turoff 1975), or a process to generate a discussion and creative thinking about the problem in question (Garrod et al. 2003).

The fundamental basis of expert-driven planning (using for instance the Delphi method) is to obtain expert judgment on issues and problems that are complex and subjective in nature (Linstone & Turoff 2002). The process can offer a transparent, formal, and agreed upon alternative (MacMillan & Marshall 2005). The approach requires fewer technical requirements and less time than conducting quantitative analyses (Lourie et al. 2004). MacMillan and Marshall (2005) cited three positive aspects of using an expert-driven approach. First, it is a good alternative when objective models and scientific data are not available. Second, the process itself can encourage discussions around controversial issues that can help achieve an agreement. Finally, some expert knowledge on complex ecological questions, for which an empirical model is not as helpful, can be accommodated and discussed.

Examples of its application in marine conservation planning include: the priority-setting for the Baja California to Bering Sea marine conservation initiative (Lourie et al. 2004), identification of marine special areas in Newfoundland and Labrador (Rao et al. 2009), the setting of conservation priorities by Conservation International in the Philippines (Ong et al. 2002), and the zoning of the Bunaken MPA in Indonesia (Salm & Clark 2000). Expert-driven planning can use criteria, although typically of qualitative nature, to quickly derive the skeleton of a network of conservation areas, but was considered to rarely result in an efficient design for such a network (M. Beck, cited in Lourie et al. 2004).

2.5.2 RANKING AND SCORING

In conservation planning, the ranking and scoring approach marks the spatial prioritization of sites (Justus & Sarkar 2002). In this approach, conservation sites are identified based on explicit criteria rather than on the intuitive judgment used in an *ad hoc* approach. Justus & Sarkar consider this approach as the beginning of a technical view of protected area design. Ratcliffe (1977) was the first to propose the use of explicit criteria in selecting conservation sites considering that not all biologically interesting areas can be conserved. Hence, Ratcliffe introduces the relative importance of various criteria (e.g. rarity is more important than the site's intrinsic value) to assist in conservation planning.

The ranking and scoring approach uses a procedure whereby a combined score of various criteria orders a conservation site. Early work on ranking and scoring focused on biological criteria although socioeconomic consideration such as the human impact was deemed important (Tans 1974, Gehlbach 1975). Examples of biological criteria include: richness and rarity (Tubbs & Blackwood 1971), naturalness (Tans 1974, Wright 1977), and representativeness (Gehlbach 1975, Wright 1977).

Scoring techniques had difficulty integrating the social, economic, and political criteria along with biological criteria (Margules & Usher 1981, Margules et al. 1982, Smith & Theberge 1987). As a result, it has been suggested that the social/economic/political be separated from natural scientific criteria (Margules & Usher 1981). Eventually, this led to the primacy of natural scientific criteria where socio-political considerations often play a role in the latter part of the process such as the final selection of a conservation site (Justus & Sarkar 2004). Since the usage of this approach started in land-based planning, few applications to marine environments can be found.

2.5.3 MATHEMATICAL OPTIMIZATION (HEURISTIC AND OPTIMAL MODELS)

With the increasing focus on identifying biologically or ecologically suitable conservation areas, a representation of various types of environment and species has become important in setting conservation goals. For this matter, the ranking and scoring approach was considered as being less effective than an optimization model (Pressey & Nicholls 1989). Kirkpatrick (1983) introduced the idea of an iterative procedure to represent biodiversity in priority areas. He implied that a non-iterative procedure (or formula such as the species richness to represent different species) can lead to the duplication of species (or other biodiversity features) in priority sites.

In response, Kirkpatrick (1983) proposed an iterative procedure where sites are considered one at a time with the first best site chosen relative to the weightings of criteria. In assessing the second best site, those species considered in the first site will have less weighting. This iterative process continues until all the species are represented, avoiding duplication of species representation. This procedure gave way to the concept of complementarity coined in Vane-Wright et al. (1991). It is referred to as “a measure of the extent to which an area, or set of areas, contributes unrepresented features to an existing area or set of areas” or “most simply, it can be thought of as the number of unrepresented species (or other biodiversity features) that a new area adds” (Margules & Pressey 2000, p 249).

The concept of complementarity led to the use of heuristic and optimal models. Heuristic models are iterative procedures that use different local search methods such as stepwise iterative heuristics (see Moilanen et al. 2009 for details). Heuristic methods achieve biological objectives (e.g. representing all species in the solution set), but only guarantee near-optimal solutions. Optimal methods, also called exact optimization methods, use linear integer programming models that guarantee a single optimal solution. This procedure is a class of mathematical optimization models and their application emerged during World War II in an attempt to maximize the allocation of scarce resources during military operations (Winston 1994). Rodrigues & Gaston (2002) provide a list of studies that use integer programming models in designing reserves.

Optimization is also related to the concept of efficiency. In identifying priority sites, efficiency (E) is defined, according to Pressey & Nicholls (1989), as E = 1 – (X/T), where X is the number or extent of highest ranking sites needed to contain all attributes a given number of times, and T is the total number of area of sites. In conservation planning, efficiency is high if biodiversity targets are achieved with a relatively small number of sites. This concept is based on the notion that choosing the smallest possible amount of area can reduce the cost of reserves, which eventually could afford greater chance of getting reserves implemented (Stewart & Possingham 2005). Optimization algorithms that provide optimal or sub-optimal solutions have become commonly used through Marxan and Zonation software applications (Ball & Possingham 2000, Moilanen 2007). One drawback of these optimization models is that they typically lump all socioeconomic data into a single "cost" term, with the risk of obscuring different social and economic factors that contribute to the overall cost (Ban & Klein 2009).

Heuristic models were found to be more efficient (i.e. tend to select less conservation areas) than scoring and ranking systems (Pressey & Nicholls 1989) but less efficient than optimal models (Pressey et al. 1997). Nevertheless, it has been argued that heuristic models are adequate for identifying priority sites (Pressey et al. 1996, 1997). Heuristic models also present significant gains in the processing time compared to optimal models. Nevertheless, the sub-optimality of heuristic models led some authors to argue that research about optimal models should be pursued to achieve maximum efficiency (Underhill 1994, Rodrigues & Gaston 2002, Fischer & Church 2005).

There are three common types of conservation prioritization problems based on how biological objectives and socioeconomic objectives (in the form of constraints) are defined in heuristics and optimal models (Moilanen et al. 2009 p31). First is the minimum set coverage or the problem of identifying a minimum reserve set (Underhill 1994). This procedure is designed to obtain a solution that reaches biological targets at the least possible socioeconomic cost. There are two potential means of expressing costs: cost can be calculated (a) only for individual sites or (b) for individual sites but also for the allocation of conservation across the landscape. Leslie et al. (2003) applied a minimum set coverage approach when designing a network of MPA reserves using data from the Florida Keys, but most applications of this method were in terrestrial environments (Pressey & Tully 1994, Pressey et al. 1997, McDonnel et al. 2002).

The second type of conservation prioritization problem is the maximal coverage, or the problem of identifying a maximum biological benefit based on a given socioeconomic cost (Camm et al. 1996, Church et al. 1996). A cost, in this case, could involve setting aside specific amounts of area for conservation or committing resources, such as a monetary budget to implement a protected area. An example of this type of problem prioritization includes maximizing the number of species represented to a pre-determined amount of reserve area (Polasky et al. 2000).

Similarly with minimum set coverage, application of this is done mostly using terrestrial data (Camm et al. 1996, Csuti et al. 1997, Snyder et al. 1999).

Finally, the last type is the maximum utility or maximization problem. It is very similar to maximal coverage but calculates the biodiversity objective based on a varying target (or benefit function) (Arponen et al. 2005). For example, a gain for a biodiversity feature is "an increasing function of the level of that feature and the total value is an additive sum across features" (Moilanen et al. 2009, p 32). Examples of work that used this approach mostly used terrestrial data (e.g. Davis et al. 2006, Wilson et al. 2006, Moilanen & Cabeza 2007).

It is important to note that in heuristic and optimal models, biological objectives (i.e. biodiversity targets) can be defined by several criteria (e.g. sets of species and habitat types). Defining a socioeconomic constraint (or cost), however, has been typically expressed as a single criterion (e.g. a specific amount of budget). This approach often provides more attention and primacy on biological objectives than socioeconomic ones (Williams et al. 2005).

2.5.4 MULTI-CRITERIA DECISION ANALYSIS (MCDA)

While optimization methods, implemented within tools like Marxan or Zonation, are popular in marine conservation planning research and practice, multi-criteria methods have also been proposed as an alternative (Sarkar et al. 2004a, Moffett & Sarkar 2006). Since biodiversity conservation is multidimensional in nature, (see Section 2.4), conservation planning should aim at capturing a number of different criteria (Gilman et al. 2011, Thorpe et al. 2011). Hence, another type of conservation prioritization problem is considering a conservation design as a multi-criteria decision problem (Moffett & Sarkar 2006, Seip & Wenstop 2006). Multi-criteria decision analysis (MCDA) is a common approach used in decision theory for reaching a decision based on a large number of criteria. These methods are also employed in decision analysis or management science (Dyer et al. 1992, Keeney & Raiffa 1993, Grunig & Khun 2009).

In conservation planning, MCDA methods offer techniques that consider different aspects of both the biodiversity targets and socio-political considerations (Brown et al. 2001, Villa et al. 2001, Moffet et al. 2005). A unique feature of MCDA techniques is their capacity to accommodate simultaneously several socio-political criteria and biodiversity targets (*sensu* Moffett et al. 2006, Sarkar et al. 2009). MCDA also allows qualitative and quantitative means of ordering the conservation sites, as opposed to generating a single set of solution areas (Moffett et al. 2005, Moffett & Sarkar 2006).

There are two tradeoff mechanisms to determine a solution set using MCDA. First is the use of the iterative stage protocol where each conservation site is evaluated by all criteria (Bojorquez-Tapia et al. 2004). Second is the terminal stage protocol composed of two phases: (a) initial selection of priority sites that satisfies a biodiversity representation; and (b) using the rest of the non-biological criteria are used to order or rank the initial priority sites (Sarkar et al. 2004b, Moffett & Sarkar

2006). In the iterative stage protocol, biodiversity targets can possibly be compromised by other criteria (e.g. socio-political). In contrast, in a terminal stage protocol, biodiversity is satisfied first and cannot be compromised.

MCDA methods include numerous techniques that can accommodate quantitative and/or qualitative criteria into a single planning process. Moffett & Sarkar (2006) divide the groupings of MCDA based on whether alternatives and criteria are ranked quantitatively or qualitatively. Some methods can be objective and as rigorous as optimal and near-optimal approaches, while other methods can be qualitative, similar to the expert-driven approach.

Another way of grouping MCDA approaches looks at whether or not objectives are explicitly or implicitly defined (Malczewski 1999). Implicit and explicit objectives are typically carried out respectively by multi-attribute decision-making attributes (MADM) and multi-objective decision-making (MODM) methods. One difference between these two groups of methods relates to how priorities and tradeoffs between PA dimensions, in the form of weights or functions, are obtained. MADM methods directly obtain priorities for attributes (or measurements) while MODM methods derive priorities from objectives (indirectly from measurements) (MacCrimon 1973). Other differences between these two groups of methods are discussed in Malczewski (1999). Some MADM methods express tradeoffs between PA dimensions through the compensatory and non-compensatory methods and outranking aggregation methods (Greene et al. 2011). Some MODM methods include the multi-attribute value and utility theories (Dyer 1979, 2005) as well as some heuristic procedures (Figueira et al. 2005).

Sarkar et al. (2009) present an application of MCDA approaches in terrestrial planning, using the ResNet and MultSync tool, a web-resource for conservation planning. Moffett & Sarkar (2006) discussed a comprehensive review of a wide variety of MCDA techniques that have potential utility in designing conservation areas, but few of them are used as planning methods. Thus far, MCDA techniques have been less explored or applied in marine planning than on land.

2.5.5 TYPOLOGY OF CONSERVATION PLANNING APPROACHES

When technical capacity and strong legal, political, and funding support exist, it may be possible to apply a single conservation approach to identify MPAs and then base the planning decisions mostly on scientific guidelines (Gleason et al. 2010). This is not often the case, however, as all PA dimensions and objectives can hardly be addressed in that manner. One reason is that social concepts relating to conservation (e.g. flexibility and effectiveness) may not be achievable along with the scientific principles that aim to achieve biological objectives such as complementarity and efficiency. The social concepts of flexibility and effectiveness regard the success of conservation implementation a priority which may not go well with achieving high biodiversity (Knight et al. 2010, Knight et al. 2011b). Knight et al. (2011a, p 207) note that “scientific rigor and sophistication are comparatively minor elements of a successful strategy development process, because stakeholder uptake and 'buy-in' are not dependent upon scientific principles." For PA decisions where prioritization of different objectives is context-specific, a classification of planning approaches based on the type of conservation principles, whether social, biological or a combination of both, is useful.

As discussed earlier in this chapter, conservation planning approaches range from a continuum of very formal (e.g. linear integer programming) to very informal (e.g. *ad hoc)*. Figure 2.1 illustrates the typology of conservation approaches using three general categories: Class A approaches largely give priority to biological criteria. They include mathematical optimization procedures like heuristic and optimal models that use mostly objective and rule-based means of measuring the attainment of objectives in the solution set. On the other end, Class C includes expert-driven and (informed) opportunism models that mostly use subjective and opportunistic means to achieve objectives in the solution set. They put a strong emphasis on socioeconomic criteria, while acknowledging the other criteria. In between the two is Class B, or an intermediate group of approaches like MCDA and scoring, which aims to better balance biological and socioeconomic criteria using a combination of subjective and objective methods to achieve conservation objectives in the solution set.

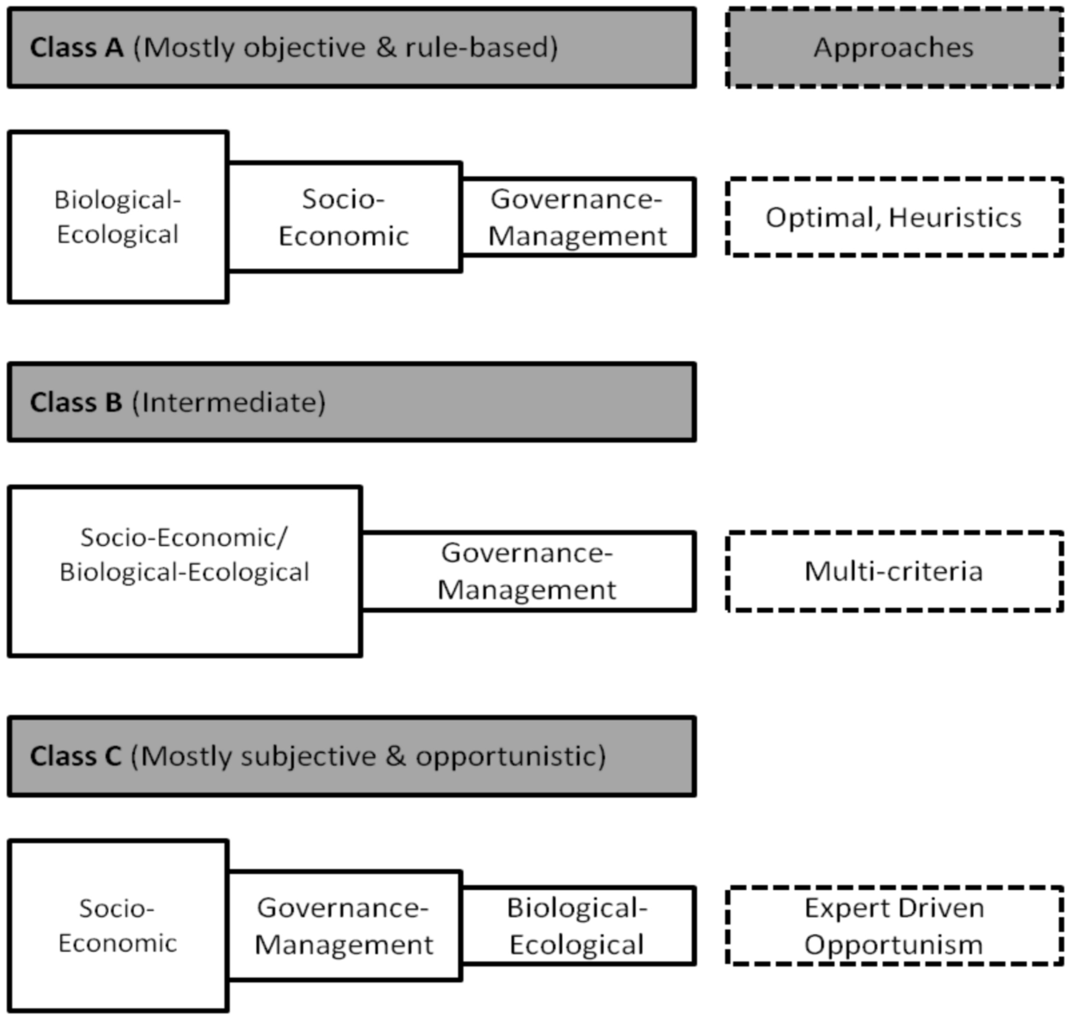


Figure 2.1 Three groupings of the existing conservation planning approaches.

When a method is designed to give biological gains priority, a solution tends to become sensitive to achieving higher biodiversity targets (Williams et al. 2004). In contrast, when a method is more concerned with socio-political aspects, taking into consideration the possibility and consequences of the implementation of priority areas, there is a tendency for a solution to become more sensitive to social constraints. Thus, it tends to lean toward compromising biodiversity to increase chances of social support and PA implementation (Knight et al. 2011a). As a result, socially sensitive approaches can easily fail the test of efficiency (Stewart et al. 2003); just as bio-ecologically focused approaches may sacrifice equity and social justice (Christie 2004, Singleton 2009). Methods designed to deal with both the multiple biological targets and the competing socioeconomic goals, like MCDA, hold promise of trying to reach acceptable tradeoffs between biological and social criteria. They have been, however, far less explored (Brown et al. 2001, Moffett & Sarkar 2006, Ban & Klein 2009).

This grouping of the approaches is useful for two reasons. First, the key principles addressed by each class of method differ among classes. Class A is grounded in natural science principles (e.g. complementarity, efficiency) based on quantitative, rule-based, repeatable, and objective procedures. Class C, on the other hand, uses qualitative social-based principles (e.g. effectiveness, flexibility) based on human, social, and financial capital as well as dynamic response to opportunities (informed opportunism). Class B attempts to combine these natural and social science principles.

Second, each method has strengths and weaknesses (Pressey & Nicholls 1989, Pressey 1994, Roberts 2000, Knight & Cowling 2007, 2008, Pressey & Botrill 2008, Cowling et al. 2009), which ultimately shape what can be achieved and/or the type of impacts to stakeholders (Calabresi 1991, Gurney et al. 2014). Considering the traditional focus of PAs on biodiversity conservation, the concepts of efficiency and complementarity are good measures to track the achievement of biodiversity targets through the biologically sensitive approaches. However, the capacity to achieve biodiversity protection may vary across socio-political contexts (Wilson et al. 2007, Jones et al. 2011).

In a situation where socio-political factors can easily compromise PA success, a socio-politically sensitive approach (e.g. Class C) should be explored (Game et al. 2011). Finally, when a situation can allow balancing between these two complex sets of criteria, a Class B approach could be more appropriate. In this regard, it is important to examine how methodological limits might affect expression of conservation priorities in selecting conservation areas.

Such typology of conservation planning approaches is important, not only because a careful choice needs to be made, but also because this choice is not universal, as limits vary across planning environments due to differences in historical, social, economic, and political situations. Context does matter, as what might apply in one may not apply in another, causing Smith et al. (2009) to argue that planning is to be led by those who understand the context, for instance the locals and not the distant institutions, who may have different sets of priorities. Therefore, planners, decision-makers, and stakeholders alike need to understand what methods can and cannot do to avoid misconceptions. Understanding different approaches to conservation planning may facilitate acceptance of the methodological limits, and hopefully dispel objections and frustrations associated with the use of conservation models (Addison et al. 2013).

2.6 DILEMMAS OF GOAL-BASED MPA PLANNING: TRADEOFFS AND HARD DECISIONS

Recognizing the multiple types of approaches enables an appreciation that conservation planning is riddled with dilemma; MPA planning dimensions compete with each other in the design of MPA approach. This acknowledgment cannot, in itself, solve the "wickedness" of the MPA problem. Below is an explanation of how pervasive the three propositions of wicked problems, articulated by Rittel & Webber (1973) and Jentoft and Chuenpagdee (2009), are in MPA planning.

First, an MPA problem has no definitive formulation. In other words, people perceive the problems associated with marine environment that give rise to MPAs differently depending on their own experience and values. To some, the problem might be the overexploitation of the natural resource that leads to biodiversity loss (Crowder & Norse 2005). While this may be true, it is necessary to ask what drives the overexploitation. Is it the perverse subsidies that promote overfishing (Sumaila & Pauly 2006)? Or is it the market policy that promotes cheap production through technological advancement that ruins fish habitats (Chuenpagdee et al. 2003, Lackey 2005)? What is the role of poverty in the overexploitation of natural resources (Williams et al. 2004)? What is the impact of other threats, such as pollution from land-based sources, on biodiversity loss (Thomas et al. 2012)? In the absence of a proper exploration of these questions, the MPA may not address the potential drivers of overexploitation or other potential explanations for the loss of marine biodiversity. It is for this reason that Boersma & Parrish (1999) described MPAs as a limited solution.

Following from the first proposition, MPA problems cannot be addressed by a definitive strategy. As suggested by Rittel & Webber (1973), a wicked problem “does not have an enumerable set of potential solutions or a well-described set of permissible operations that may be incorporated in the plan” (p. 164). They reason that this is the case because “there are no criteria which enable one and prove that all solutions to a wicked problem have been identified and considered” (p. 164). Therefore, if defining a problem *is* a problem, so is determining a definitive strategy and procedure. Again, consider the issue of biodiversity loss. To find "the criteria", one has to examine the causes of biodiversity loss. However, as noted previously, this can go beyond the scope of MPA solution. Even in cases where competing criteria may be known, selection of a procedure is typically contested or is not easy to apply (Lackey 2006, Fanning et al. 2007, Salafsky 2011).

Finally, Rittel & Webber (1973) stated that “wicked problems have no stopping rule” (p. 162). In MPA planning, this is apparent in the nature and the way goals develop. Jentoft et al. (2011) argued that goals are not straightforward, and their formation may go through several stages. For example, goals can get initially formed, displaced, and adjusted over time. Others suggest that goals may have a tendency to conflict with one another (Bailey & Jentoft 1990, Lackey 2005, Jones 2006, Seip & Wenstop 2006). These conflicting objectives reflect the fact that desirable goals may not be in harmony with one another or cannot be achieved all at once. In fact, after establishment and implementation of MPA, the MPA can simultaneously be a biological success and a social failure (Christie 2004). One reason is that conflicts and issues often arise from an attempt to simultaneously achieve biodiversity protection and human well-being (McShane et al. 2011). As a result, some document that existing conflicts and new types of conflicts can continue to cause problems in MPA management (Oracion et al. 2005, Macintosh et al. 2010, Ledee & Sutton 2011).

Mounting evidence shows that, in practice, not everyone can win from MPA decision-making outcomes (Schmid 2002, Chuenpagdee et al. 2005, Oracion et al. 2005, Hirsch et al. 2011, Macintosh et al. 2010, McShane et al. 2011). As a result, there is a growing interest in better understanding the notion of HDs where everyone cannot possibly win, as opposed to the view of "win-win" choice (Bailey & Jentoft 1990, Lackey 2006, McShane et al. 2011). Choices or decisions are driven by values (Keeney 1996) so are the conservation choices (Soulé 1985, Schmid 2002, Seip & Wenstop 2006, Wilhere 2007). Disregarding human values can easily motivate a lofty goal because there is less regard on the limits they can pose in making practical choice (Bailey & Jentoft 1990). One of these limits is associated with the (in)compatibility and (in)comparability of values involved in making decisions (Chang 1997, Kooiman & Jentoft 2005).

When values are compatible or comparable, exchange between values is possible; otherwise, it is difficult (Kooiman & Jentoft 2005). It is in this regard that tradeoffs and HDs differ. Tradeoff "refers to the idea that, in any choice between a range of options that we face may be compared as more, less, or equally to be preferred” [*sic*] (Holland 2002, p 17). Hence, in order for tradeoff to occur, compatibility and comparability between the values is essential (Chang 1997) so they can be substitutable (Jiang & Eastman 2000) or exchangeable (Holland 2002). In this case, choices that involve tradeoffs can be relatively easy or moderate (Kooiman & Jentoft 2005). However, when incomparability between values exists, it is beyond the scope of tradeoff and may require HDs where choices can involve "either-or" type of decisions (Schmid 2002, Kooiman & Jentoft 2005, p 294).

To explain HDs in conservation terms, consider socioeconomic losses that tend to be diverse and distributed unequally among stakeholders (Daw et al. 2011). As a result, conservation choices tend to make some stakeholders pay more than others (Lackey 2005, Adams et al. 2010). This often causes disagreement and/or conflicts among various resource users (Lackey 2006, Muthiga 2007, Hind et al. 2010, Macintosh et al. 2010). Choices like this are fundamentally value-based, it implicitly or explicitly respond to the planning question "conservation for whom?" which often is difficult to answer (Oracion et al. 2005) or poses HDs (Nutt 2002, Schmid 2002). HDs are hard as the values towards who is going to lose or win are incomparable as it would require foregoing even valid interests (Bailey & Jentoft 1990, Kooiman & Jentoft 2005).

Disregarding incomparable values often leads to conflicting objectives. Bailey & Jentoft (1990) illustrated a classic example where valid policy objectives that are typically supported by many development programs in developing nations are plainly antagonistic and irreconcilable. For example, increasing exports and domestic supply of fish are compelling objectives, yet they are hardly achieved simultaneously as export oriented markets lead to low supply of domestic fish and encourage over-exploitation of fish at the local level. From the point of view of spatial planning, tradeoffs and HDs can occur when two conservation decision-making environments are satisfied.

First, when conservation loss and gain overlaps in one site, tradeoffs and HDs can occur. In one instance, the overlap might involve biological gain and socioeconomic loss, while in another the overlap is about biological loss and socioeconomic gain (Figure 2.2). In particular, these losses and gains refer to deviations from current wealth (Kahneman & Tversky 1979). This means that gain and loss are linked respectively to an increase and a decrease from what is currently enjoyed. For example, in identifying sites for conservation, gain typically refers to biodiversity protection or gains in socioeconomic activities with reference to current situation. Loss refers to cost such as the negative impacts on socioeconomic activities often referred to as foregone benefits (Naidoo et al. 2006, Carwardine et al. 2009). Loss also refers to the negative impacts of socioeconomic activities on biodiversity (Ban & Klein 2009) especially when activities are to be continued from an undesired status quo. Losses (or costs) have many other categories in conservation planning (Naidoo et al. 2006). Nevertheless, the biodiversity loss and foregone socioeconomic benefits discussed in the literature are the two common costs that hinder the achievement of MPA goals.

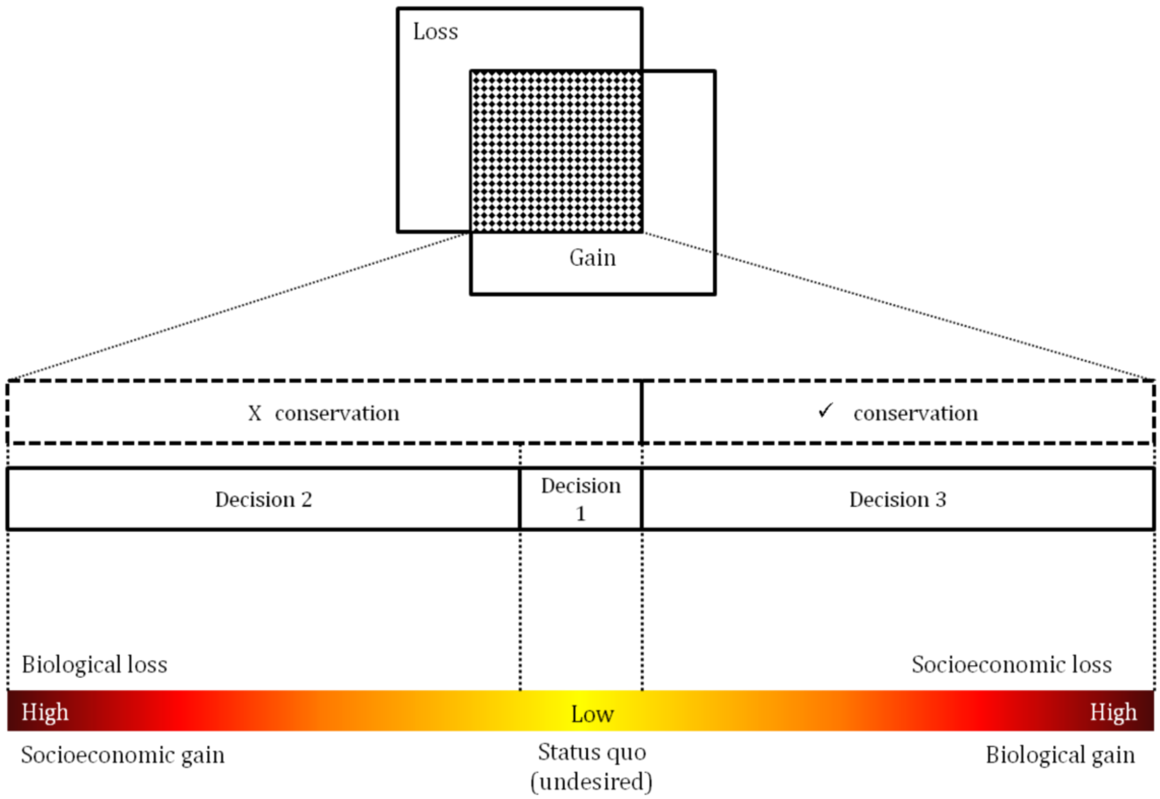


Figure 2.2 Three types of decisions can occur when spatial conflict (overlap) between loss and gain occurs along with an undesired status quo. The conservation decision can be hard as it has to choose between socioeconomic loss and biological gain.

Second, when the conservation decision environment deals with an undesired *status quo* (situated in the middle portion of the bar), see Figure 2.2. In a well-managed fisheries or when there is no exploitation, an MPA can be easily established or may be unnecessary. On the contrary, if the undesired *status quo* involves overexploitation or unsustainable use of resources, any decision toward achieving conservation gain can involve difficult choices (Jones 2006).

These two conditions create a tug-of-war between competing losses (or competing gains). At a glance, the term competing gains (i.e. socioeconomic gain vs.

biological gain) does not outrightly communicate its true nature, which is when more preference is given to one, it can actually mean loss of the other. For this reason the term "competing losses" is emphasized to explain the pervasive nature of loss in making conservation choices which could explain why the "win-win" choice is rare or may not be possible (Calabresi 1991).

In the spatial planning perspective, three types of decisions are possible within the tug-of-war of competing losses. *Decision 1* chooses the *status quo*, resisting change. This decision poses a relatively low risk against the existing socioeconomic activities but poses risks of losing biodiversity. *Decision 2* chooses the left side of the *status quo* (cf. Figure 2.2), resisting conservation. This decision aims for more protection and development of socioeconomic activities, but is possible only with losing more biodiversity. In contrast, *Decision 3* chooses the right side of the *status quo*, promoting biodiversity conservation. This decision aims for biological protection but is possible only in the exchange of socioeconomic losses.

Decisions 1 and 2 involve actions outside the focus of MPA and conservation which typically recommends banning or restricting exploitative activities. Hence, with conserving biodiversity as a goal, a planner is left with Decision 3 that expects some form of socioeconomic loss. The strength of loss involved in Decision 3 can be associated with the intensity of overlap between loss and gains that increases from the middle towards the end of the bar. This means that losses may have varying degrees that can result to tradeoffs and HDs.

2.7 THE CHALLENGE OF MAKING COMPETING LOSSES, TRADEOFFS, AND HARD DECISIONS EXPLICIT

A current challenge for the conservation planning community is to find explicit ways of identifying losses, tradeoffs, and HDs more explicitly (Smith et al. 2010, McShane et al. 2011). Often, tradeoffs and HDs are further complicated by a planning process that is neither transparent nor explicit, which is often the case (Adams 2010, Brosius 2010, Hirsch et al. 2010). In such instances, doubts about the legitimacy of the process and the conservation effort itself are raised leading ultimately to a sense of injustice among affected stakeholders (Muthiga 2007, Hind et al. 2010). Thus, when a conservation program falls short of expectation, it could be due to insufficient recognition and less explicit treatment of tradeoffs and HDs.

Over the last four decades of method development research in conservation planning, there is less emphasis on the integration of losses in planning (Ban & Klein 2009). One reason is that, so far, procedures mostly researched are of those of Class A approaches (Justus & Sarkar 2002). As noted in the previous section, these approaches tend to pay less attention to socioeconomic losses or exclude the tradeoffs or HDs involved among the competing socioeconomic losses.

Since competing losses and HDs are inseparable, it is worth investigating how explicit treatment of losses can make tradeoffs and HDs acceptable or legitimate. If HDs are associated with the spatial conflict between conservation loss and gains, one might ask if it is possible to know the extent of this spatial conflict as well as of the HDs themselves. Another reason for making losses explicit is related to the increasing evidence that conflicts among ocean users exist and can hamper success of MPAs.

In identifying areas for conservation, it would be interesting to see how these losses, tradeoffs, and HDs can be made spatially explicit following a goal-based MPA planning framework. For the purpose of this work, tradeoffs and HDs are used to elaborate the emerging notion of the lack of win-win and the increasing need to make difficult choices (see Chapter 4).

2.8 LITERATURE CITED

Aacharya, R., C. Gastmans, and Y. Denier. 2011. Emergency department triage: An ethical analysis. BMC emergency medicine **11**:16.

Adams, W. 2010. Path dependence in conservation. Pages 292-310 in N. Leader-Williams, W. Adams, R. Smith editors. Tradeoffs in conservation: Deciding what to save. Wiley-Blackwell, Oxford.

Adams, V., B. Pressey, and R. Naidoo. 2010. Opportunity costs: Who really pays for conservation? Biological Conservation **143**:439–448.

Adams, W., Aveling, R. Brockington, D. Dickson, B. Elliott, J. Hutton, D. Rose, B. Vira, and W. Wolmer. 2004. Biodiversity conservation and the eradication of poverty. Science **306**:1146-1149.

Adams, V., B. Pressey, and R. Naidoo. 2010. Opportunity costs: Who really pays for conservation? Biological Conservation **143**:439–448.

Addison, P., L. Rumpff, S. Bau, J. Carey, Y. Chee, F. Jarrad, M. McBride and M. Burgman. 2013. Practical solutions for making models indispensable in conservation decision-making. Diversity and Distributions **19**:490-502.

Adler, M. 2008. Bounded Rationality and Legal Scholarship. Research Paper No. 08-03. Institute for Law and Economics. University of Pennsylvania Law School, Pennsylvania.

Airame, S., J. Dugan, K. Lafferty, H. Leslie, D. McArdle, and R. Warner. 2003. Applying ecological criteria to marine reserve design: A case study from the California Channel Islands. Ecological Applications **13**:170-184.

Alcala, A. 2004. Marine reserves as tools for fishery management and biodiversity conservation: natural experiments in the Central Philippines, 1974–2000. Available from https://www.cbd.int/doc/nbsap/fisheries/ALCALA.pdf (accessed December 2014).

Altshuler, A. 1965. The City planning process. Cornell University Press. Ithaca, New York,

Ando, A., Camm, J. Polasky, S. and A. Solow. 1998. Species distributions, land values, and efficient conservation. Science **279**:2126–2128.

Arponen, A., R. Heikkinen, C. Thomas, and A. Moilanen. 2005. The value of biodiversity in reserve selection: Representation, species weighting, and benefit functions. Conservation Biology **19**:2009-2014.

Badalamenti, F., A. Ramos, E. Voultsiadou, J. Sánchez Lizaso, G. D'Anna, C. Pipitone, J. Mas, D. Ruiz Fernandez, D. Witmarsh, and S. Riggio. 2000. Cultural and socio-economic impacts of Mediterranean marine protected areas. Environmental Conservation **27**:10-125.

Bailey, C. and S. Jentoft. 1990. Hard choices in fisheries development. Marine Policy **14:** 333-344.

Ball, I. and H. Possingham. 2000. MARXAN (V1.8.2): Marine reserve design using spatially explicit annealing, a manual prepared for the Great Barrier Reef Marine Park Authority. Available from http://www.uq.edu.au/marxan/docs/marxan_manual_1_8_2.pdf (accessed December 2014).

Ban, N., and C. Klein. 2009. Spatial socioeconomic data as a cost in systematic marine conservation planning. Conservation Letters **2**:206-215.

Bavinck, M., and V. Vivekanandan. 2011. Conservation, conflict, and the governance of fisher wellbeing: Analysis of the establishment of the Gulf of Mannar National Park and Biosphere Reserve. Environmental Management **47**:593-602.

Boersma, P. and J. Parrish. 1999. Limiting abuse: Marine protected areas, a limited solution. Ecological Economics **31**:287-304.

Bohnsack, J. 1998. Application of marine reserves to reef fisheries. Australian Journal of Ecology **23**:298-304.

Bojórquez-Tapia, L., H. de la Cueva, S. Díaz, D. Melgarejo, G. Alcantar, M. José Solares, G. Grobet and G. Cruz-Bello. 2004. Environmental conflicts and nature reserves: Redesigning Sierra San Pedro Mártir National Park, Mexico. Biological Conservation **117**:111-126.

Bottrill, M., et al. 2008. Is conservation triage just smart decision making? Trends in Ecology and Evolution **23**:649-654.

Brosius, P. 2010. Conservation tradeoffs and the politics of knowledge. Pages 311-328 in N. Leader-Williams, W. Adams and R. Smith, editors. Tradeoffs in conservation: Deciding what to save. Wiley-Blackwell, Oxford, UK.

Brown, K., W. Adger, E. Tompkins, P. Bacon, D. Shim, and K. Young. 2001. Trade-off analysis for marine protected area management. Ecological Economics **37**:417–434.

Calabresi, G. 1991. The pointlessness of pareto: Carrying Coase further. The Yale Journal Company, Inc. **100**:1211-1237.

Camm, J., S. Polasky, A. Solow, and B. Csuti. 1996. A note on optimal algorithms for reserve site selection. Biological Conservation **78**:353-355.

Carwardine, J., C. Klein, K. Wilson, R. Pressey, and H. Possingham. 2009. Hitting the target and missing the point: Target-based conservation planning in context. Conservation Letters **2**:4-1.

Chang, R. 1997. Incommensurability, incomparability, and practical reason. Cambridge, Harvard University Press.

Christensen, K. 1985. Coping with uncertainty in planning. Journal of the American Planning Association **51**:63-73.

Christensen, V., M. Coll, C. Piroddi, J. Steenbeek, J. Buszowski, J. and D. Pauly. 2014. A century of fish biomass decline in the ocean. Marine Ecology Progress Series **512**:155-166.

Christie, P. 2004. Marine Protected Areas as Biological Successes and Social Failures in Southeast Asia. American Fisheries Society **42**:155–164.

Chuenpagdee, R., P. Degnbol, M. Bavinck, S. Jentoft, D. Johnson, R. Pullin, and S. Williams. 2005. Challenges and concerns in capture fisheries and aquaculture. Pages 11-24 in J. Kooiman, M. Bavinck, S. Jentoft, and R. Pullin, editors. Fish for life: Interactive governance for fisheries. Amsterdam University Press, Amsterdam.

Chuenpagdee, R., L. Morgan, S. Maxwell, E. Norse, and D. Pauly. 2003. Shifting gears: assessing collateral impacts of fishing methods in US waters. Frontiers in Ecology and the Environment **1**: 517-524.

Chuenpagdee, R., and S. Jentoft. 2009. Governability assessment for fisheries and coastal systems: A reality check. Human Ecology **37**:109-120.

Chuenpagdee, R., J. Pascual-Fernandez, E. Szelianszky, J. Alegret, J. Fraga, and S. Jentoft. 2013. Marine protected areas: Re-thinking their inception. Marine Policy **39**:234–240.

Cicin-Sain, B., and S. Belfiore. 2005. Linking marine protected areas to integrate coastal and ocean management: A review of theory and practice. Ocean and Coastal Management **48**:847-868.

Chuenpagdee, R., and S. Jentoft. 2007. Step zero for fisheries co-management: What precedes implementation? Marine Policy **31**:657-668.

Church, R., D. Stoms, D. and F. Davis. 1996. Reserve selection as a maximal covering location problem. Biological Conservation **76**:05-112.

Cowling, R., A. Knight, S. Privett, and G. Sharma. 2009. Invest in opportunity, not inventory of hotspots. Conservation Biology **24**:633-635.

Crowder, L., and E. Norse. 2005. The greatest threat: Fisheries. Pages 183-184 in A. Elliot and L. Crowder, editors. Marine conservation biology: The science of maintaining the seas biodiversity. Island Press, Washington.

Crowder L., et al. 2006. Resolving mismatches in U.S. Ocean Governance. Science **313**:617-618.

Csuti, B., S. Polasky, P. Williams, R. Pressey, J. Camm, M. Kershaw, A. Keister, B. Downs, R. Hamilton, M. Huso, and K. Sahr. 1997. A comparison of reserve selection algorithms using data on terrestrial vertebrates in Oregon. Biological Conservation **80**:83-97.

Davis, F., C. Costello, and D. Stoms. 2006. Efficient conservation in a utility-maximization framework. Ecology and Society **11**:33.

Davies, A., J. Roberts, and J. Hall-Spencer. 2007. Preserving deep-sea natural heritage: emerging issues in offshore conservation and management. Biological Conservation **138**:299-312.

Daw, T., K. Brown, S. Rosendo, and R. Pomeroy. 2011. Applying the ecosystem services concept to poverty alleviation: The need to disaggregate human well-being. Environmental Conservation **38**:370-379.

Degnbol, P., H. Gislason, S. Hanna, S. Jentoft, J. Nielsen, S. Sverdrup-Jensen and D. Clyde Wilson. 2006. Painting the floor with a hammer: Technical fixes in fisheries management. Marine Policy **30**:534-543.

Devillers, R., R. Pressey, A. Grech, J. Kittinger, G. Edgar, T. Ward, and R. Watson. 2014. Reinventing residual reserves in the sea: Are we favouring ease of establishment over need for protection? Aquatic Conservation: Marine and Freshwater ecosystems.

Di Minin, E., and A. Moilanen. 2012. Empirical evidence for reduced protection levels across biodiversity features from target-based conservation planning. Biological Conservation **153**:187-191.

Doney, S. 2010. The growing human footprint on coastal and open-ocean biogeochemistry. Science **328**:1512-1516.

Dyer, J. 2005. MAUT-multiattribute utility theory. Pages 265–295 in J. Figueira J, S. Greco, and M. Ehrgott, editors. Multiple Criteria Decision Analysis: State of the Art Surveys. Kluwer, Dordrecht, Netherlands.

Dyer, J., P. Fishburn, R. Steuer, J. Wallenius, and S. Zionts. 1992. Multiple criteria decision making, multiattribute utility theory: the next ten years. Management Science **38**:645-654.

Dyer, J. and R. Sarin. 1979. Measurable multiattribute value functions. Operations Research **27**:810–22.

Edgar, G., S. Banks, R. Bensted-Smith, M. Calvopiña, A. Chiriboga, L. Garske, S. Henderson, K. Miller, and S. Salazar, 2008a. Conservation of threatened species in the Galapagos Marine Reserve through identification and protection of marine key biodiversity areas. Aquatic Conservation: Marine and Freshwater Ecosystems **18**:955-968.

Edgar, G., et al. 2008b. Key biodiversity areas as globally significant target sites for the conservation of marine biological diversity. Aquatic Conservation: Marine and Freshwater Ecosystems **18**:969-983.

Ehrlich, P. and A. Ehrlich. 1981. Extinction. Random House, New York.

Fanning, L., et al. 2007. A large marine ecosystem governance framework. Marine Policy **31**:434-443.

Fernandes, L., et al. 2005. Establishing representative no-take areas in the Great Barrier Reef: Large-scale implementation of theory on marine protected areas. Conservation Biology **19**:1733–1744.

Figueira, J., S. Greco and M. Ehrgott. 2005. Multiple Criteria Decision Analysis: State of the Art Surveys. Kluwer, Dordrecht, Netherlands.

Fischer, D., and R. Church. 2005. The SITES reserve selection system: A critical review. Environmental Modeling and Assessment **10**:215-228.

Fisher, R., and R. Ury. 1981. Getting to yes: Negotiating agreement without giving in. Houghton Mifflin Company, Boston.

Fiske, S. 1992. Sociocultural aspects of establishing marine protected areas. Ocean & Coastal Management **18**:25-46.

Friedmann, J. 1987. Planning in the Public Domain: From Knowledge to Action. Princeton University Press, New Jersey.

Game, E., G. Lipsett-Moore, R. Hamilton, N. Peterson, J. Kereseka, W. Atu, M. Watts, and H. Possingham. 2011. Informed opportunism for conservation planning in the Solomon Islands. Conservation Letters **4**:38-46.

Garrod, B. 2003. Defining marine ecotourism: A Delphi study. Pages 17-36 in B. Garrod, and J. Wilson. Marine Ecotourism: Issues and experiences. Channel View Publications.

Gehlbach, F. R. 1975. Investigation, evaluation, and priority ranking of natural areas. Biological Conservation **8**:79-88.

Gell, F., and C. Roberts. 2003. Benefits beyond boundaries: The fishery effects of marine reserves. Trends in Ecology and Evolution **18**:448–56.

Gilman, E., D. Dunn, A. Read, K. Hyrenbach, and R. Warner. 2011. Designing criteria suites to identify discrete and networked sites of high value across manifestations of biodiversity. Biodiversity and Conservation **20**:3363-3383.

Gleason, M., S. McCreary, M. Miller-Henson, J. Ugoretz, E. Fox, M. Merrifield, W. McClintock, P. Serpa, and K. Hoffman. 2010. Science-based and stakeholder-driven marine protected area network planning: A successful case study from north central California. Ocean & Coastal Management **53**:52–68.

Glover, A. and C. Smith. 2003. The deep-sea floor ecosystem: Current status and prospects of anthropogenic change by the year 2025. Environmental Conservation **30**:219-241.

Good, I. 1953. The population frequencies of species and the estimation of population parameters. Biometrika **40**:237-264.

Greene, R., R. Devillers, J. Luther, and B. Eddy. 2011. GIS-based multiple-criteria decision analysis. Geography Compass **5**:412-432.

Grilo, C. 2011. Institutional interplay in networks of marine protected areas with community-based management. Coastal Management **39**:440-458.

Grunig, K., and R. Kuhn. 2009. Successful decision-making: A systematic approach to complex problems. 2nd edition. Springer, Berlin.

Gurney, G., R. Pressey, C. Ban, J. Álvarez-Romero, S. Jupiter, and V. Adams. 2015. Efficient and equitable design of marine protected areas in Fiji through inclusion of stakeholder-specific objectives in conservation planning. Conservation Biology.

Halpern, B., K. McLeod, A. Rosenberg, and L. Crowder. 2008. Managing for cumulative impacts in ecosystem-based management through ocean zoning. Ocean and Coastal Management **51**:203-211.

Healey, P. 1992. A planner's day: Knowledge and action in communicative practice. Journal of the American Planning Association **58**:9-20.

Hind, E., M. Hiponia, and T. Gray. 2010. From community-based to centralised national management - a wrong turning for the governance of the marine protected area in Apo Island, Philippines? Marine Policy **34**:54-62.

Hirsch, P., W. Adams, J. Brosius, A. Zia, N. Bariola and J. Dammert. 2011. Acknowledging conservation trade-offs and embracing complexity. Conservation Biology **25**:259-264.

Holland, A. 2002. Are Choices tradeoffs? Pages 17-34 in D. Bromley and J. Paavola, editors. Economics, ethics, and environmental policy: Contested choices. Blackwell Publishing, Oxford, UK.

Huber, O. 1986. Decision making as a problem solving process. Pages 109-138 in B. Brehmer, H. Jungermann, P. Lourens, and G. Sevon, editors. New directions in research on decision making. Elsevier Science Publishers B.V., The Netherlands.

Hurlbert, S. 1971. The non-concept of species diversity: A critique and alternative parameters. Ecology **52**:577-586.

Innes, J. 1995. Planning theory's emerging paradigm: Communicative action and interactive practice. Journal of Planning Education and Research **14**:183-189.

IUCN-World Commission on Protected Areas (IUCN-WCPA). 2008. Establishing resilient marine protected area networks-making it happen. IUCN-WCPA, National Oceanic and Atmospheric Administration and the Nature Conservancy, Washington, DC.

Jackson, J., et al. 2001. Historical overfishing and the recent collapse of coastal ecosystems. Science **293:**629-638.

James, A. 1999. Institutional constraints to protected area funding. Parks **9**:15-26.

Jameson, S., M. Tupper, and J. Ridler. 2002. The three screen doors: Can marine protected areas be effective. Marine Pollution Bulletin **44**:1177-1183.

Jiang, H. and J. Eastman. 2000. Application of fuzzy measures in multi-criteria evaluation in GIS. International Journal of Geographical Information Science **14**:173-184.

Jentoft, S., T. van Son, and M. Bjorkan. 2007. Marine protected areas: A governance system analysis. Human Ecology **35**:611-622.

Jentoft, S. and R. Chuenpagdee. 2009. Fisheries and coastal governance as a wicked problem. Marine Policy **33**:553-560.

Jentoft, S., R. Chuenpagdee, and J. Pascual-Fernandez. 2011. What are MPAs for: On goal formation and displacement. Ocean and Coastal Management **54**:75-83.

Jones, P. 2006. Collective action problems pose by no-take zones. Marine Policy **30**:143–156.

Jones, P., W. Qiu, and E. De Santo. 2011. Governing marine protected areas: Getting the balance right. United Nations Environment Program, Nairobi.

Justus, J., and S. Sarkar. 2002. The principle of complementarity in the design of reserve networks to conserve biodiversity: A preliminary history. Journal of Biosciences **27**:421-435.

Kahneman, D., and A. Tversky. 1979. Prospect theory: An analysis of decision under risk. Econometrica: Journal of the Econometric Society **47**:263-291.

Keeney, R. and H. Raiffa. 1976. Decisions with multiple objectives: Preferences and value tradeoffs. Wiley, New York.

Kelleher, G., C. Bleakley, and S. Wells. 1995. A global representative system of marine protected areas. IUCN, Gland, Switzerland.

Kirkpatrick, J. 1983. An iterative method for establishing priorities for the selection of nature reserves: An example from Tasmania. Biological Conservation **25**:127-134.

Kirlin, J., M. Caldwell, M. Gleason, M. Weber, J. Ugoretz, E. Fox, and M. Miller-Henson. 2013. California's Marine Life Protection Act Initiative: Supporting implementation of legislation establishing a statewide network of marine protected areas. Ocean and Coastal Management **74**:3-13.

Knight, A., R. Cowling, and B. Campbell. 2006a. An operational model for implementing conservation action. Conservation Biology **20**:408-419.

Knight, A., et al. 2006b. Designing systematic conservation assessments that promote effective implementation: Best practice from South Africa. Conservation biology **20**:739-750.

Knight, A. and R. Cowling. 2007. Embracing opportunism in the selection of priority conservation areas. Conservation Biology **21:**1124–1126.

Knight, A. and R. Cowling. 2008. Clearing the mud from the conservation opportunity debate: Reply to Pressey and Bottrill. Conservation Biology **22**:1346–1348.

Knight, A., R. Cowling, A. Boshoff, S. Wilson, and S. Pierce. 2011a. Walking in STEP: Lessons for linking spatial prioritizations to implementation strategies. Biological Conservation **144:**202-211.

Knight, A. T., H. Grantham, R. Smith, G. McGregor, H. Possingham, and R. Cowling. 2011b. Land managers' willingness-to-sell defines conservation opportunity for protected area expansion. Biological Conservation **144**:2623-2630.

Knight, A., R. Cowling, M. Difford, and B. Campbell. 2010. Mapping human and social dimensions of conservation opportunity for the scheduling of conservation action on private land. Conservation Biology **24**:1348–1358.

Kooiman, J. and S. Jentoft. 2005. Hard choices and values. Pages 285-299 in J. Kooiman, M. Bavinck, S. Jentoft, and R. Pullin, editors. Fish for life: Interactive governance for fisheries. Amsterdam University Press, Amsterdam.

Kreps, D. 1988. Notes on the Theory of Choice. Westview Press, Boulder.

Kukkala, A., and A. Moilanen. 2013. Core concepts of spatial prioritization in systematic conservation planning. Biological Reviews **88**:443-464.

Lai, S., and L. Hopkins. 1989. The meanings of trade-offs in multi-attribute evaluation methods: A comparison. Environment and Planning B-Planning and Design **16:**155-170.

Lackey, R. 2005. Economic growth and salmon recovery: An irreconcilable conflict? Fisheries **30**:30-32.

Lackey, R. 2006. Axioms of ecological policy. Fisheries **31**:286-290. [Reprinted Renewable Resources Journal **24**:13-16].

Langhammer P., et al. 2007. Identification and gap analysis of key biodiversity areas: Targets for comprehensive protected area systems. IUCN best practice protected areas. Guidelines Series No. 15. Gland, Switzerland.

Leader-Williams, N., W. Adams, and R. Smith. 2010. Trade-offs in conservation: Deciding what to save. Wiley-Blackwell, Oxford, UK.

Leslie, H., M. Ruckelshaus, I. Ball, S. Andelman, and H. Possingham. 2003. Using siting algorithms in the design of marine reserve networks. Ecological Applications **13**:185-198.

Linstone, H., and M. Turoff. 1975. The delphi method: Techniques and applications. Addison-Wesley Publishing Company, Reading, Massachusetts.

Linstone, H. and M. Turoff. 2002. The delphi method: Techniques and applications. Available from http://is.njit.edu/pubs/delphibook/index.html#toc (accessed December 2014).

Ledee, E., S. Sutton, R. Tobin, and D. De Freitas. 2012. Responses and adaptation strategies of commercial and charter fishers to zoning changes in the Great Barrier Reef Marine Park. Marine Policy **36**:226-234.

Leeworty V. and P. Wiley. 2003. Socioeconomic impact analysis of marine reserve alternatives for the Channel Islands National Marine Sanctuary. US Department of Commerce, National Oceanic and Atmospheric Administration, National Ocean Service (Special Project), Silver Spring, Maryland.

Lourie, S., and A. Vincent. 2004. Using biogeography to help set priorities in marine conservation. Conservation Biology **18**:1004-1020.

MacCrimmon, K. 1968. Descriptive and normative implications of the decision theory postulates. Pages 3-32 in K. Borch and J. Mossin, editors. Risk and uncertainty. Martin's Press, New York.

Mackway-Jones, K., Marsden, J. and J. Windle. 2006. Emergency triage. 2nd edition. Blackwell Publishing, Massachusetts.

Malczewski, J. 1999. GIS and Multicriteria Decision Analysis. John Wiley & Sons, Inc., New York.

Marris, E. 2007. Conservation priorities: What to let go. Nature **450**:152-155.

Meir, E., S. Andelman, and H. Possingham. 2004. Does conservation planning matter in a dynamic and uncertain world? Ecology Letters **7**:615-622.

Merrifield, M., W. McClintock, C. Burt, E. Fox, M. Gleason, P. Serpa, and C. Steinback. 2013. Marine map: A web-based platform for collaborative marine protected area planning. Ocean and Coastal Management **74**:67-76.

Macintosh, A., T. Bonyhady, and D. Wilkinson. 2010. Dealing with interests displaced by marine protected areas: A case study on the Great Barrier Reef Marine Park structural adjustment package. Ocean and Coastal Management **53**:581-588.

MacMillan, D., and K. Marshall. 2006. The delphi process-an expert-based approach to ecological modelling in data-poor environments. Animal Conservation **9**:11-19.

Margules, C., A. Higgs, and R. Rafe. 1982. Modern biogeographic theory: Are there any lessons for nature reserve design? Biological Conservation **24**:115-28.

Margules, C., and R. Pressey. 2000. Systematic conservation planning. Nature **405**:203-253.

Margules, C. and S. Sarkar. 2007. Systematic Conservation Planning. University Press, Cambridge.

Margules, C., and M. Usher. 1981. Criteria used in assessing wildlife conservation potential: A review. Biological Conservation **21**:79-109.

McDonnell, M., H. Possingham, I. Ball, and E. Cousins. 2002. Mathematical methods for spatially cohesive reserve design. Environmental Modeling and Assessment **7**:107-114.

McNeely J., K. Miller, W. Reid, R. Mittermeier, and T. Werner. 1990. Conserving the world's biological diversity. IUCN, Gland, Switzerland.

McNeill, S. E. 1994. The selection and design of marine protected areas: Australia as a case study. Biodiversity and Conservation **3**:586-605.

McShane, T., et al. 2011. Hard choices: Making trade-offs between biodiversity conservation and human well-being. Biological Conservation **144**:966-972.

Mittermeier, R., and T. Werner. 1990. Wealth of plants and animals unites “megadiversity” countries. Tropicus **4**:1-5.

Moffett, A., J. Dyer, and S. Sarkar. 2006. Integrating biodiversity representation with multiple criteria in North-Central Namibia using non-dominated alternatives and a modified analytic hierarchy process. Biological Conservation **129**:181-191.

Moffett, A., J. Garson, and S. Sarkar. 2005. MultCSync: A software package for incorporating multiple criteria in conservation planning. Environmental Modelling and Software **20**:1315-1322.

Moffett, A., and S. Sarkar. 2006. Incorporating multiple criteria into the design of conservation area networks: A minireview with recommendations. Diversity and Distributions **12**:125-137.

Moilanen, A. 2007. Landscape zonation, benefit functions and target-based planning: unifying reserve selection strategies. Biological Conservation **134**:571-579.

Moilanen, A. 2008. Generalized complementarity and mapping of the concepts of systematic conservation planning. Conservation Biology **22**:1655-1658.

Moilanen, A., and M. Cabeza. 2007. Accounting for habitat loss rates in sequential reserve selection: Simple methods for large problems. Biological Conservation **136**:470-482.

Moilanen A., K. Wilson, and H. Possingham. 2009. Spatial Conservation Prioritization: Quantitative methods and computational tools. Oxford University Press, New York.

Muthiga, N. 2007. The evolution of management and impacts on communities adjacent to the Mombasa marine protected area, Kenya. Pages 27-37 in K. Redford, and E. Fearn, editors. Protected Areas and human livelihoods, Working Paper No. 32. Wildlife Conservation Society, New York.

Myers, N. 1984. Genetic resources in jeopardy. Ambio: A Journal of the Human Environment **13**:171-174.

Myers, N., R. Mittermeier, C. Mittermeier, G. Da Fonseca, and J. Kent. 2000. Biodiversity hotspots for conservation priorities. Nature **403**:853-858.

Myers, R., and B. Worm. 2003. Rapid worldwide depletion of predatory fish communities. Nature **423**:280-283.

Naidoo, R., A. Balmford, P. Ferraro, S. Polasky, T. Ricketts, and M. Rouget. 2006. Integrating economic costs into conservation planning. Trends in Ecology and Evolution **2**:681-687.

Newell, A., and H. Simon. 1972. Human problem solving theory. Prentice Hall, Inc., New Jersey.

Norse, E., and L. Crowder. 2005. Why marine conservation biology? Pages 1-30 in A. Elliot and L. Crowder, editors. Marine conservation biology: The science of maintaining the seas biodiversity. Island Press, Washington.

Noss, R. F., Carroll, C., Vance-Borland, and G. Wuerthner. 2002. A multicriteria assessment of the irreplaceability and vulnerability of sites in the Greater Yellowstone ecosystem. Conservation Biology **16**:895-908.

Nutt, P. 2002. Why decisions fail: Avoiding the blunders and traps that lead to debacles. Berrett-Koehler Publishers, Inc., San Francisco.

Ong, P., L. Arfuang, and R. Rosell-Amball. 2002. Philippine biodiversity conservation priorities: a second iteration of the national biodiversity strategy plan. Department of Environment and Natural Resources, Conservation International Philippines, University of the Philippines Center for Integrative and Development Studies, and Foundation for the Philippines Environment, Quezon City, Philippines.

Oracion, E., M. Miller, and P. Christie. 2005. Marine protected areas for whom? Fisheries, tourism, and solidarity in Philippine community. Ocean and Coastal Management **48**:393-410.

Ostrom, E. 1990. Governing the commons: The evolution of institutions for collective action. Cambridge University Press, Cambridge.

Pielou, E. 1969. An introduction to mathematical ecology. Wiley, New York.

Polasky, S., J. Camm, A. Solow, B. Csuti, D. White, and R. Ding. 2000. Choosing reserve networks with incomplete species information. Biological Conservation **94**:1-10.

Pressey, R. 1994. *Ad hoc* reservations: forward or backward steps in developing representative reserve systems? Conservation Biology **8**:662-668.

Pressey, R., H. Possingham, and C. Margules. 1996. Optimality in reserve selection algorithms: when does it matter and how much? Biological Conservation **76**:259-267.

Pressey, R., et al. 2007. Conservation planning in a changing world. Trends in ecology and evolution **22**:583-592.

Pressey R., and M. Bottrill. 2008. Opportunism, threats, and the evolution of systematic conservation planning. Conservation Biology **22**:1340-1345.

Pressey, R., T. Hager, K. Ryan, J. Schwarz, S. Wall, S. Ferrier, S. and P. Creaser. 2000. Using abiotic data for conservation assessments over extensive regions: Quantitative methods applied across New South Wales, Australia. Biological Conservation **96**:55-82.

Pressey, R., and A. Nicholls. 1989. Efficiency in conservation: Scoring versus iterative approaches. Biological Conservation **50:**199-218.

Pressey, R., H. Possingham, and J. Day. 1997. Effectiveness of alternative heuristic algorithms for identifying indicative minimum requirements for conservation reserves. Biological Conservation **80**:207-219.

Pressey, R., and K. Taffs. 2001. Scheduling conservation action in production landscapes: Priority areas in western New South Wales defined by irreplaceability and vulnerability to vegetation loss. Biological Conservation **100**:355-376.

Pressey, R., and S. Tully. 1994. The cost of ad hoc reservation: a case study in western New South Wales. Australian Journal of Ecology **19**:375-384.

Purvis, A. and A. Hector. 2000. Getting the measure of biodiversity. Nature **405**:212-219.

Rao, A.,L. Outhouse, and D. Gregory. 2009. Areas of interest in our marine backyards. Canadian Parks and Wilderness Society (CPAWS), Newfoundland and Labrador Chapter, 181 pp.

Ratcliffe, D. 1977. The selection of biological sites of nation importance to nature conservation in Britain. Cambridge University Press, Cambridge.

Rittel, H., and M. Webber. 1973. Dilemmas in a general theory of planning. Policy Sciences **4:**155-169.

Roberts, C. 2000. Selecting marine reserve locations: optimality versus opportunism. Bulletin of marine science **66**:581-592.

Roberts, C., and J. Hawkins. 1999. Extinction risk in the sea. Trends in Ecology and Evolution **14**:241-246.

Roberts, C., et al. 2002. Marine biodiversity hotspots and conservation priorities for tropical reefs. Science **295**:1280-1284.

Roberts, C., et al. 2003. Ecological Criteria for Evaluating Candidate Sites for Marine Reserves: The Science of Marine Reserves. Ecological Applications **13**:199-214.

Rodrigues, A., and K. Gaston. 2002. Optimisation in reserve selection procedures-why not? Biological Conservation **107**:123-129.

Salafsky, N. 2011. Integrating development with conservation: a means to a conservation end, or a mean end to conservation? Biological Conservation **144**:973-978.

Salm, R., J. Clark, and E. Siirila. 2000. Marine and coastal protected areas: A guide for planners and managers. 3rd edition. IUCN, Gland Switzerland.

Sanchirico J., K. Cochran, and P. Emerson. 2002. Marine protected areas: Economic and social implications. Resources for the Future, Washington, DC.

Sarkar, S., A. Moffett, R. Sierra, T. Fuller, S. Cameron, and J. Garson. 2004a. Incorporating multiple criteria into the design of conservation area networks. Endangered Species Update **21**:100-107.

Sarkar, S., C. Pappas, J. Garson, A. Agarwal, and S. Cameron. 2004b. Place prioritization for biodiversity conservation using probabilistic surrogate distribution data. Diversity and Distributions **10**:125-133.

Sarkar, S., R. Pressey, D. Faith, C. Margules, T. Fuller, D. Stoms, A. Moffett, K. Wilson, K. Williams, P. Williams, and A. Andelman. 2006. Biodiversity conservation planning tools: Present status and challenges for the future. Annual Review of Environment and Resources **31**:123-159.

Sarkar, S., T. Fuller, A. Aggarwal, A. Moffett, and C. Kelly. 2009. The ConsNet software platform for systematic conservation planning. Pages 235-248 in A. Moilanen, K. Wilson, and H. Possingham, editors. Spatial conservation prioritization: Quantitative methods and computational tools. Oxford University Press, New York.

Seip, K., and F. Wenstop. 2006. A primer on environmental decision-making: An integrative quantitative approach. Springer, Dordrecht, The Netherlands.

SEP (Stanford Encyclopedia of Philosophy). Fall 2013 Edition. Available from http://plato.stanford.edu/archives/fall2013/entries/rationality-instrumental/ (accessed December 2014).

Schmid, A. 2002. All environmental policy instruments require a moral choice as to whose interests count. Pages 133-147 in D. Bromley and J. Paavola, editors. Economics, ethics, and environmental policy: Contested choices. Blackwell Publishing, Oxford, UK.

Seip, K. and F. Wenstop. 2006. A primer on environmental decision-making: An integrative quantitative approach. Springer, The Netherlands.

Shea, K. 1998. Management of populations in conservation, harvesting, and control. Trends in Ecology and Evolution **13**:371-375.

Shears, N. and R. Babcock. 2003. Continuing trophic cascade effects after 25 years of no-take marine reserve protection. Marine Ecology. Progress Series **246**:1-16.

Singleton, S. 2009. Native people and planning for marine protected areas: How "stakeholder" processes fail to address conflicts in complex, real-world environments. Coastal Management **37**:421-440.

Simon, H. 1991. Bounded rationality and organizational learning. Organization Science **2**:125-134.

Smith, R., W. Adams, N. Leader-Williams. 2010. Another entangled bank: Making conservation trade-offs more explicit. Pages 365-376 in N. Leader-Williams, W. Adams and R. Smith, editors. Tradeoffs in conservation: Deciding what to save. Wiley-Blackwell, Oxford, UK.

Smith, P., and J. Theberge. 1987. Evaluating natural areas using multiple criteria: Theory and practice. Environmental Management **11**:447-60.

Smith, R.J., D. Verissimo, N. Leader-Williams, R. Cowling, A. Knight. 2009. Let the locals lead. Nature **462**:280-281.

Snyder, S., J. Miller, A. Kibbe, and R. Haight. 2007. Habitat acquisition strategies for grassland birds in an urbanizing landscape. Environmental Management **40**:981-992.

Soulé, M.1985. What is conservation biology? BioScience **35**:727-734.

Stewart, R., T. Noyce, and H. Possingham. 2003. Opportunity cost of ad hoc marine reserve design decisions: An example from South Australia. Marine Ecology Progress Series **253**:25-38.

Stewart, R., and H. Possingham. 2005. Efficiency, costs and trade-offs in marine reserve system design. Environmental Modeling and Assessment **10:**203-213.

Sumaila, U., and D. Pauly. 2007. All fishing nations must unite to cut subsidies. Nature **450**:945-945.

Susskind, L., and J. Cruikshank. 1987. Breaking the Impasse Consensual Approaches to Resolving Public Disputes. Basic Books, New York.

Svancara, L., R. Brannon, J. Scott, C. Groves, R. Noss, and R. Pressey. 2005. Policy-driven versus evidence-based conservation: A review of political targets and biological needs. BioScience **55:**989-995.

Tans, W. 1974. Priority ranking of biotic natural areas. Michigan Botanist **13**:31-39.

Thomas, C., I. Gordon, S. Wooldridge, and P. Marshall. 2012. Balancing the tradeoffs between ecological and economic risks for the Great Barrier Reef: A pragmatic conceptual framework. Human and Ecological Risk Assessment: An International Journal **18**:69-91.

Thorpe, A., M. Bavinck, and S. Coulthard. 2011. Tracking the debate around marine protected areas: Key issues and the BEG framework. Environmental Management **47**:546-563.

Tubbs, C., and J. Blackwood. 1971. Ecological evaluation of land for planning purposes. Biological Conservation **3**:169-172.

Underhill, L. 1994. Optimal and suboptimal reserve selection algorithms. Biological Conservation **70**:85-87.

UNEP. 1995. Global biodiversity assessment. Cambridge University Press, Cambridge.

Vane-Wright, R., C. Humphries, and P. Williams. 1991. What to protect? Systematics and the agony of choice. Biological Conservation **55**:235-254.

Villa, F., L. Tunesi, and T. Agardy. 2001. Zoning marine protected areas through spatial multiple-criteria analysis: The case of the Asinara Island National marine Reserve of Italy. Conservation Biology **16**:515-526.

von Neumann, J., and O. Morgenstern. 1947. Theory of Games and Economic Behavior. Princeton University Press, New Jersey.

Weeks, R., G. Russ, A. Bucol, and A. Alcala. 2010. Incorporating local tenure in the systematic design of marine protected area networks. Conservation Letters **3:**445-453.

White A., and C. Courtney. 2002. Experience with marine protected area planning and management in the Philippines. Coastal Management **30**:1-26.

Wilhere, G. 2007. The how-much-is-enough myth. Conservation Biology **22:**514–517.

Williams, J., C. ReVelle, and S. Levin. 2004. Using mathematical optimization models to design nature reserves. Frontiers in ecology and the environment **2**:98-105.

Williams, J., C. ReVelle, and S. Levin. 2005. Spatial attributes and reserve design models: a review. Environmental Modeling and Assessment **10**:163-181.

Wilson, K., M. McBride, M. Bode,and H. Possingham. 2006. Prioritizing global conservation efforts. Nature **440**:337-340.

Wilson, K. A., et al. 2007. Conserving biodiversity efficiently: What to do, where, and when. Public Library of Science Biology **5:**1850-1861.

Winston, W. 1994. Operations research-applications and algorithms. 3rd edition. Duxbury Press, Belmont, CA.

Wood, L., and S. Dragicevic. 2007. GIS-based multi-criteria evaluation and fuzzy sets to identify priority sites for marine protection. Biodiversity Conservation **16**:2539-2558.

Worm, B., et al. 2006. Impacts of biodiversity loss on ocean ecosystem services. Science **314**:787-790.

Worm, B., M. Sandow, A. Oschlies, H. Lotze, and R. Myers. 2005. Global patterns of predator diversity in the open oceans. Science **309**:1365-1369.

World Resources Institute (WRI). 1992. Global biodiversity strategy. World Resources Institute, Washington, D.C.

Wright, D. 1977. A site evaluation scheme for use in the assessment of potential nature reserves. Biological Conservation **11**:293-305.

CHAPTER 3 DATASETS AND METHODS IN GENERATING BIOLOGICAL AND SOCIOECONOMIC ATTRIBUTES

3.1 INTRODUCTION

Biodiversity preservation is one of the key considerations when conserving natural ecosystems. Conservation biology is fundamentally anchored to the notion that natural ecosystems and their biological processes have intrinsic value and need protection (Soulé 1986). An additional reason for protecting biodiversity is that it contributes to the productivity of the entire ecosystem (Loreau et al. 2001, Hooper et al. 2005). Consequently, a productive ecosystem can help secure ecosystem services (e.g. food, water, clean air, ecotourism), ultimately benefiting society (Holmlund & Hammer 1999, Worm et al. 2006).

Some authors have suggested that conservation planning should help protect all aspects of biodiversity, including species, genetics, and ecosystems (WRI 1992, UNEP 1995). In practice, however, such levels of representation are often not possible due to an insufficient knowledge of ecological systems. As a result, different concepts were developed and used for defining and representing biodiversity over the last four decades. Moilanen (2008) and Kukkala and Moilanen (2012) presented comprehensive discussions of the various concepts used in conservation planning. Our study used species diversity (i.e. richness, evenness), abundance, and rarity to represent the spatial distributions of species.

Issues related to maintaining biodiversity and ecosystem services are hardly separable from humanity's innate relationship with nature that tends to create challenges in the context of MPA planning (Erlich & Wilson 1991, UNEP 1992). This relationship can make biodiversity harder to protect, as places hosting valuable biodiversity are often places of high human and economic interests (Myers et al. 2000, Roberts et al. 2002).

As a result, balancing the economic losses that result from allocating spaces for conservation purposes, such as marine protected areas (MPA), has been a challenge. This "balancing" process is typically referred to as dealing with tradeoffs between biodiversity and socioeconomic objectives (Margules & Pressey 2000, Stewart & Possingham 2005, and Leader-Williams 2010). While this theoretically requires representation of both biodiversity and socioeconomic interests, most systematic conservation planning (SCP) methods often consider biological objectives in more detail than socioeconomic objectives (Pressey & Nicholls 1989, Ando et al. 1998, Ban & Klein 2009).

To explore and test a novel multi-criteria approach to be discussed in Chapter 4, sets of data were gathered and processed to generate attributes that can represent biological and socioeconomic interests. In multi-criteria decision analysis (MCDA), attributes can be considered as indicators or fundamental sources of information to formulate and achieve desired outcomes such as biodiversity objectives or economic objectives (Starr & Zeleny 1977). In short, an attribute “becomes an objective when it is assigned a purpose, a direction of desirability or improvement” (Zeleny 1982, p 6).

Another important term in MCDA is "criteria," which refers to both concepts of attributes and objectives (Malczewski 1999); this similar definition of criteria was used in this study. In MCDA, objectives can be organized as a "hierarchy" in which the high-level objectives are unpacked into low-level objectives (Malczewski 1999). All these levels of objectives are called "criteria," as are the attributes.

This chapter presents the datasets and methods used for generating the 25 attributes in the form of maps, which will be introduced in Section 3.2. These attributes were derived from marine datasets for Newfoundland and Labrador and were grouped into three types: (1) biodiversity, (2) fishing activities, and (3) other marine uses. Biodiversity attributes are measures of abundance and diversity of marine taxa including groundfish, corals, sponges, and seabirds. The attributes for fishing activities include landed weights and monetary values of fish, the distribution of unique fishing vessels, based on vessel ID, (i.e. fishing business), the distribution of fishers (i.e. fishing employment), and the diversity measures of different groups of fishers, hereby called fisher groups (FGs). Attributes for other human uses include density calculations of marine commercial traffic, offshore oil and gas exploration, and production, gear impacts, and spatial overlap between gears.

The following sections present the datasets and their spatial and temporal coverage, followed by descriptions of the calculations used to generate the spatial attributes into GIS-based raster format. Then, the actual raster-based attributes per category are presented in the results section, followed by a discussion and summary chapter.

3.2 STUDY AREA, DATASETS, AND DATA SOURCES

This study was conducted for the Newfoundland and Labrador (NL) continental shelf and slope biogeographic unit, one of the 12 biogeographic units of Canada's marine areas (DFO 2009). It is situated in the Atlantic Ocean and the eastern portion of the province of Newfoundland and Labrador. Additional areas outside the boundary of the NL bioregion were included in the study area. These include the Flemish Pass, Flemish Cap, and Southwest tail of the Grand Banks (Figure 3.1).

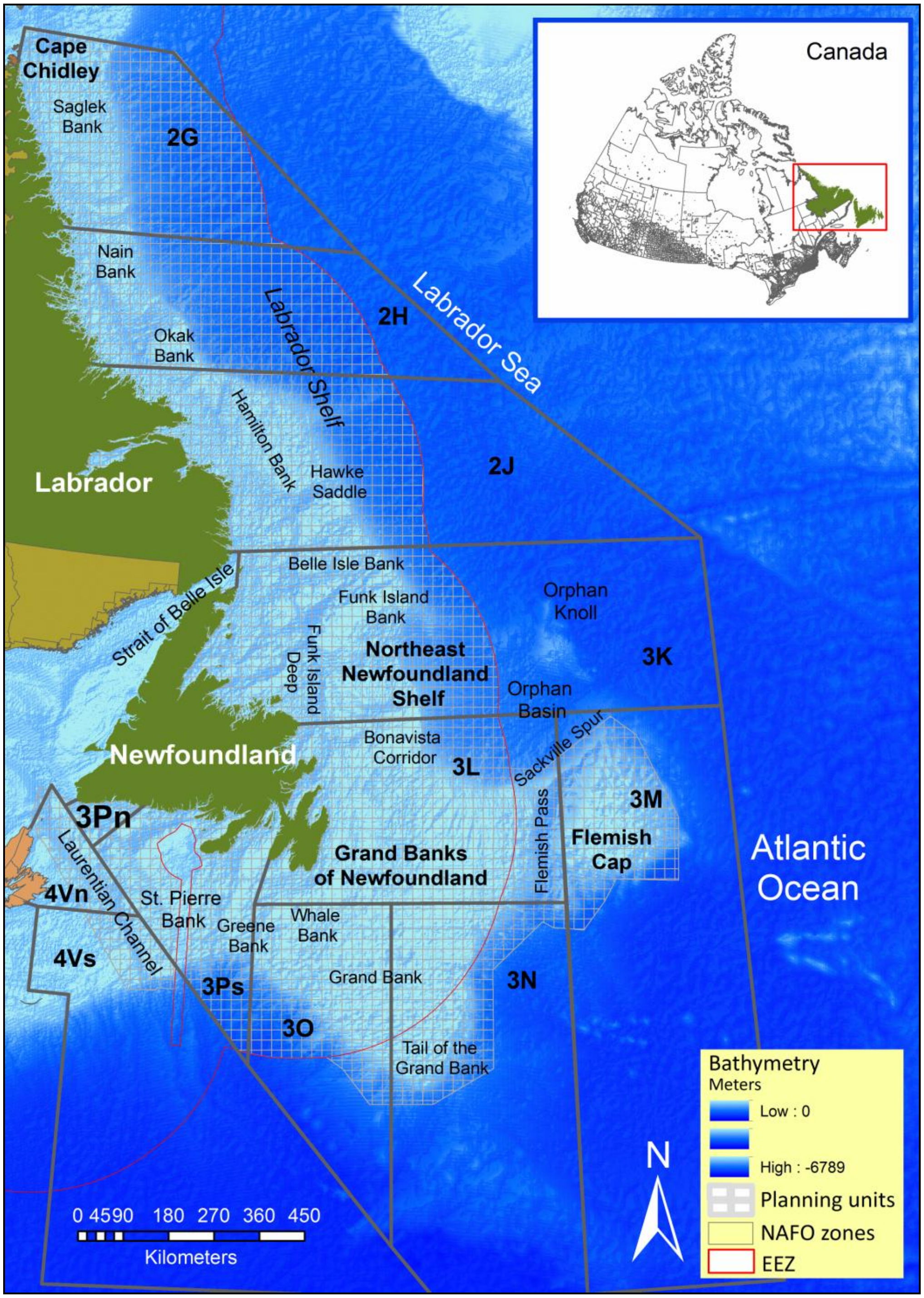


Figure 3.1 Study area, Newfoundland and Labrador (NL) bioregion, showing the Northwest Atlantic Fisheries Organization (NAFO) zones, the Canadian Exclusive Economic Zone (EEZ), and the study planning units (PUs). GEBCO bathymetry is shown in meters. The lightest shade of the study area outlines the NL shelves.

Most datasets, as mentioned in Table 3.1, cover the entire study area, except for many inshore areas and those areas beyond the mid-continental slope at 1500 m depth, due to the absence of data.

Historically, this region has been recognized as a rich ground for commercial cod fishing, which provided catches of about 100,000 tons annually from the late 1500s to early 1700s, and 1 million tons annually from the mid-1960s to the 1970s (Rose 2003). Catches however significantly declined in the 1980s and led to the collapse of the cod fishery in the early 1990s (Hutchings & Myers 1994, Rose 2003).

Due to the increasing need for science-based advice to manage fishery resources, the multi-species survey (MSS) program was implemented (McCallum & Walsh 1997). The MSS has made an annual collection of benthic marine species since 1971, but a more consistent and wider area coverage of these surveys started in 1995 (Brodie 2005). The MSS data from the Department of Fisheries and Oceans (DFO) are the primary biological datasets used in the study. These are supplemented by the seabird data obtained from the Canadian Wildlife Service (CWS).

Besides MSS, logbooks containing records of actual commercial fishing trips were also used in the study. Other human uses also exist in the study area, including marine transportation and oil and gas. Datasets for these activities were respectively obtained from the Canadian Coast Guard and the Canada-Newfoundland and Labrador Oil and Petroleum Board (C-NLOPB), respectively.

Table 3.1 GIS thematic layers used in the study and how they were generated

Dataset	Temporal Coverage	Source/Name	Description	Generated decision attribute
Groundfish	1995-2007	DFO-St. John's Multispecies survey	Datasets arranged based on per tow (coded by vessel, trip, set or VTS) for each species. VTS records correspond to per sample set standardized on a 15-minute tow	Regional rare and regional endemic species, species richness, species density, species evenness species status (e.g. endangered, threatened, special concern)
Corals	2000-2010	DFO-St. John's Multispecies survey	Datasets in VTS format for each species based on a standardized 15-min tow.	Sensitivity of coral groups, species richness, species density
Sponges	1995-2011	DFO-St. John's (DFO multispecies survey*)	Datasets received in raw format (coded entries) but were extracted in VTS for each species based on a standardized 15-min tow	Biomass
Exploited invertebrates	1995-2011	DFO-St. John's (Multispecies survey)	Shrimp and crab records in VTS format for each species based on a standardized 15-min tow	Biomass
Seabirds	1965-1992 Programme intégré de recherches sur les oiseaux pélagiques (PIROP). 2006-2011 Eastern Canada Seabirds at Sea (ECSAS)	CWS (PIROP and ECSAS surveys)	Datasets arranged for each species per WatchID (i.e. per sample set)	Richness, evenness, density
Logbook	2001-2010	DFO-Policy & Economics Branch, St. John's	Datasets arranged on a per trip basis	Gear conflict, gear impact, landed catch, fishing business, fishing employment, FG-Richness, FG-Evenness
Long Range Identification and Tracking (LRIT)	February 2010-February 2011	CCG-Maritime Security	Location of mobile offshore drilling units, high speed craft, passenger and cargo vessels (≥ 300 gross tonnage on international travel) recorded on a 6-hour interval	Density of marine commercial traffic
Oil and gas	1986-2010 (Exploration, production and significant discovery) and 1980-2010 for Wells with some data from 1966-1979	C-NLOPB	Point data for wells and polygon data for licenses including exploration, significant discovery and production areas	Density of oil and gas activities

3.3 METHODS: GENERAL GIS-BASED METHODS AND CALCULATIONS

The following subsections describe the three general GIS-based methods and calculations used in this study. Due to the nature of the datasets (e.g. large amount of data, positional errors, different formats in which the original datasets were provided), organizing and cleaning the data were necessary before any calculation could be performed. Python scripting, generally performed outside a GIS environment, was used for organizing and calculating the large amount of data.

Section 3.3.1 introduces how planning units (PUs) were generated, serving as the basic unit of analysis for which each attribute was calculated. Section 3.3.2 discusses the general concepts and procedures used when applying kernel density analyses to generate the spatial attributes. Section 3.3.3 presents the calculation used for generating the biological diversity (richness, evenness) and species abundance based on weight or count data.

3.3.1 GENERATING PLANNING UNITS

An empty polygonal vector grid of 20 x 20 km resolution, covering the entire study region, was created using the ArcGIS™ 10 *Fishnet* tool as a base for planning units (PUs). All raster-based attributes were calculated using this grid to make sure that all attributes followed the same grid size and alignment (i.e. coordinates of a corner of the grid and orientation of the grid). The grid resolution was selected based on the distribution and density of the data points for key datasets used in this study and was consistent with earlier studies using similar datasets for the region of interest (e.g. Edinger et al. 2007, Goulet et al. 2010). Finer spatial resolution is possible for regions where sample points are more densely distributed. Each cell of the polygonal grids held a unique identifier (PU ID).

The grid was extended by three grid cells beyond the boundary of the study area to avoid potential edge effects for calculating the raster-based attributes. This extended grid was subsequently converted to a raster grid using the *Polygon to Raster* tool in the ArcGIS™. Raster-based attributes were clipped back to the extent of the study area after density calculations.

3.3.2 ESTIMATING RASTER-BASED ATTRIBUTES

The density maps presented in the results section are the spatial attributes in raster format (also called raster-based attributes). They were estimated using the ArcGIS™ *Kernel Density* tool to calculate, for each raster cell, a density value. These raster-based attributes (e.g. species richness, species evenness) were derived from species distribution data, such as point sample data defined by latitude and longitude coordinates. Density analyses were used to generate continuous density maps of features (e.g. species abundance) from original sample data points. Not all PUs were surveyed. Those that were not surveyed (i.e. few km from the shoreline, outside the shelf edge) were excluded from the analysis. PUs with no record or observations were given a value of zero (e.g. crab layer). PUs with no record sandwiched between PUs with records received values through the kernel density analysis.

There are several existing methods for calculating density. This study used the kernel density method for estimating all raster-based attributes, with the exception of the marine commercial traffic attribute, which used the line density method. Kernel density is a commonly used non-parametric method for estimating the probability density function of a random variable (Silverman 1986). Estimating the function f from observed data provides inferences about the distribution of a random variable. A number of kernel functions exist (Silverman 1986, de Smith et al. 2007). This study used a quadratic or *Epanechnikov* kernel function (Equation 3.1) that ArcGIS™ implements. The quadratic function is considered the optimal function compared to other kernel density estimators (Wand & Jones 1995, Zucchini 2003).

Another important aspect of calculating the relative surface density is the search radius. It is a parameter that determines the smoothing of the density surface. A larger search radius tends to show generalized patterns, while a smaller radius can show more local variation (Mitchell 1999, p 80). While a search radius for calculating relative surface density can be established using rigorous quantitative procedures, some researchers suggest that identifying an appropriate search radius is more an art than a science (De Smith et al. 2007, p 140).

Surface density maps with different search radii were compared visually to assess appropriate radii for each attribute layer. Due to the different number of data points available for each dataset, different values for search radius were used for different datasets (see Table 3.2 for details). By using 10, 15, and 25 km as the search radius, regional patterns were more easily observed. Over-smoothing was observed in a search radius above 30 km, whereby generalized patterns sacrificed some local variation.

Table 3.2 GIS thematic layers used in the study and how they were generated

Data and Sources	GIS based Indicators	Description	Calculation Method
Seabird historical and recent surveys (PIROP and ECSAS), CWS	Species richness	Average species count per sample set	• Kernel density is based on seabird species count per sample set, then an average is estimated per planning unit • Search Radius: 10 km
	Species density	Average count of individuals per sample set	• Kernel density is based on the number of seabird individuals per sample set then an average is estimated per planning unit • Search Radius: 10 km
	Species evenness	• Equality or distribution of individuals among species per sample set • Evenness score ranges from 0-1, lowest to highest • Evenness is equal to 1 when all species in the planning unit are equally abundant	• Used Shannon-Wiener Index (H') as a measure of evenness. This index is affected by both number of species and evenness of their population. Diversity increases as both increases • Kernel density is based on the evenness score, per sample set, then an average is estimated per planning unit • Search Radius: 10 km
Groundfish – all (1995-2007), taken from multi-species surveys (MSS), DFO.	Species richness	Average of groundfish species count per sample set	• Kernel density is based on the count of groundfish species per sample set then an average is estimated per planning unit • Search Radius: 10 km
	Species biomass	Average of groundfish biomass per sample set	• Kernel density is calculated based on biomass of groundfish per sample set, then an average is estimated per planning unit • Search Radius: 10 km
	Species evenness	Same as the evenness for seabird	• Same as species evenness for seabird • Search Radius: 10 km
	Regional rarity	• Distribution of rare species based on presence. • Number of rare species per square kilometer	• Kernel density is based on presence of identified rare species • These are groundfish species with small number of individuals (≤ 8) and occupy small percentage of planning units (< 0.5 %) and can be found in more than

Data and Sources	GIS based Indicators	Description	Calculation Method
			two defined areas of the ecoregion for the study area. • Search Radius: 15 km
	Regional endemism	• Distribution of endemic species based on presence • Number of regional endemic species per square kilometer	• Kernel density is based on presence of identified rare species per planning unit • These are groundfish species with small to medium number of individuals (≤64) and occupy small percentage of planning units (up to 1 %) and can be found in one or two defined areas of the ecoregion. • Search Radius: 15 km
	Species status	• Density distribution of species with vulnerable status – that is, endangered, threatened and of special concern. Winter skate was included even though it is labeled data deficient in NL • Included 11 species listed by COSEWIC and by Devine et al. (2006) • Average of species status score per planning unit	• The various species, based on status, were aggregated, then weighted – that is, 0.48, 0.24, 0.16, 0.12 for endangered, threatened, special concern and data deficient respectively • The average of species status scores were calculated per sample set. Then, an average of the status score is obtained per planning unit • Search Radius: 10 km
Corals (1998-2010), by-catch in MSS, DFO.	Species Richness	Average of species count per sample set	• Kernel density is based on the count of groundfish species per sample set then an average is estimated per planning unit. • Search Radius: 25 km
	Coral biomass	Average biomass of coral species per sample set per PU	• Kernel density is based on the biomass of corals, per sample set, and then an average is estimated per planning unit. • Search Radius: 25 km
	Coral group sensitivity	• Distribution of corals by groups, based on their sensitivity It is the average of sensitivity score	• The coral species were divided into 7 groups that were given the following sensitivity scores: hard bottom gorgonian (5), hard bottom antipatharian (5), hard bottom cup coral (4), soft bottom gorgonian (4), seapen (3), soft bottom cup coral (2) and soft coral (1). The kernel density is based on the sensitivity score per sample set. Then, the sensitivity score is averaged per

Data and Sources	GIS based Indicators	Description	Calculation Method
			planning unit. • Search Radius: 25 km
Sponges (1998-2010), by-catch in MSS, DFO.	Sponge biomass	Average biomass of sponges per sample set	• Kernel density is based on biomass of sponges per sample set, then an average is estimated per planning unit • Search Radius: 25 km
Exploited invertebrates (1995-2011), MSS, DFO	Shrimp biomass	Average biomass of shrimp per sample set	• Kernel density is based on the biomass of shrimp per sample set then an average is estimated per planning unit. • Search Radius: 10 km
	Crab biomass	Average biomass of crab per sample set	• Kernel density is based on the biomass of crab, per sample set, and then an average is estimated per planning unit. • Search Radius: 10 km
Aggregated foregone benefits of fishing (2001-2010), DFO logbook.	Landed biomass(kg)	Average prorated biomass of fished species per trip	• Kernel density is based on the prorated biomass of fished species per trip, then an average is estimated per planning unit • Search Radius: 20 km
	Landed value (C$)	Average prorated revenue (in dollar value) of fished species per trip	• Kernel density is based on the prorated dollar value of fished species, then an average is estimated per planning unit • Search Radius: 20 km
Fisher groups (2001-2010), DFO logbook.	Richness – Fisher groups	Sum of fisher groups per PU. In short, each planning unit score indicates the number of fisher groups that uses it.	• Fisher groups were modeled using vessel length and 6 major fisheries in the province. • Biodiversity index for species richness was used in calculating this layer. In this calculation, fisher group is similar to a unique species in a biological dataset. • Kernel density is based on the count of fisher groups per planning unit. • Search Radius: 20 km
	Evenness – Fisher groups	Equality of fisher groups' distribution per PU	• Fisher groups were modeled using vessel length and 6 major fisheries in the province • Biodiversity index for species evenness was used in

Data and Sources	GIS based Indicators	Description	Calculation Method
			calculating this layer • Kernel density calculation is based on the proportion or equality of distribution of *fisher-groups* in each planning unit. • Search Radius: 20 km
Fishing business, Fishing Employment (2001-2010), DFO logbook.	Fishing business	• Number of unique vessel per square kilometer for each PU • Higher score means that there are more individual businesses interested in those planning units	• The dataset is represented with only one geographic point for each unique vessel per unique PU. Then, kernel density calculation was based on unique vessels recorded per PU. Search Radius: 15 km
	Fishing employment	• Sum of fishing vessel crew per km^2 for each planning unit • Higher score means that there is more fishing employment in those planning units	• The dataset is represented with only one geographic point for each unique vessel per unique PU. • Crew member multipliers were used for each vessel category. • Kernel density is calculated based on multipliers • Search Radius: 15 km
Oil and gas (Historical and recent records, C-NLOPB)	Density of oil and gas activities	Density of the oil and gas activities, per square kilometer, approximated from geographic locations of oil and gas wells and licenses	• Areas of oil and gas licenses – that is, production, significant discovery and exploration were converted to geographic point. • The above points were then merged with the geographic points representing oil and gas wells • Kernel density was calculated based on these merged datasets. • Search Radius: 20 km
Gear Conflicts (2001-2010), DFO logbook.	Effort overlap between fix and mobile gears (including shrimp and crab gears)	• Effort overlap between fixed and mobile gear • Higher PU score means that the effort overlap between fix and mobile is also higher.	Vessel length was used to determine and standardize effort. Effort overlap was calculated using the lowest density score for fix and mobile gears as threshold. MCE-FLOWA using "ALL" was used to determine this threshold
Gear impacts (2001-2010), DFO logbook.	Negative ecological impacts of fishing gears	• Sum of the severity scores for each planning unit • Higher score means that there is high ecological negative impact	• Gears were scored based on severity ranking. This severity ranking is based on the ecological negative impacts of gears in Fuller et al. (2007) • Kernel density was calculated based on the severity

Data and Sources	GIS based Indicators	Description	Calculation Method
			score of gear per planning unit • Search Radius: 10 km
Marine Traffic 2010-2011, LRIT, CCGS-Maritime Security	Line density of commercial vessels	• Density of commercial vessel tracks (per square kilometer in each planning unit) based on long range identification and tracking (LRIT) of ship data • A higher value means higher usage for commercial transportation	• The 6-hour geographic point for each trip of commercial vessel, based on LRIT data, was converted into line object using a script tool in ArcGIS™ 10 • Line density (commercial vessel tracks) was calculated for each planning unit using *Point Density* tool in ArcGIS™ 10 • Search Radius: 10 km

The kernel density value in planning unit i (D_i), at a distance d_{ig} from a data point g is determined as the sum of individual kernel surfaces generated for the data points. Given a normalized function, where the distances d_{ig} are divided by the search radius h, a quadratic kernel function was obtained through the following equation.

$$K_g = (3 * (1 - t^2))/4, |t| \leq 1, otherwise = 0, |t| > 1 \quad \text{Equation 3.1}$$

Where $t = d_{ig}/h$ and h is the search radius or smoothing parameter (De Smith et al. 2007). Note that each data point g of an attribute j (e.g. species richness, fishing employment) is associated with a value (X_g), a quantity to be smoothed out to create a continuous surface of an attribute j across PUs. This value specifies the volume under the kernel surface by multiplying the kernel (K_g) of a data point with its associated value (X_g). Hence, the relative kernel density output for PU i (or D_i) was obtained by summing the values of all kernel surfaces where they overlay the center of PU i (Equation 3.2). In cases where averages or correction of effort were necessary, an average of sums of kernel density surfaces was obtained (Equation 3.3).

$$D_i = \sum_{g=1}^{n} K_g * X_{ig}^{j} \quad \text{Equation 3.2}$$

$$D_i = \sum_{g=1}^{n} \frac{K_g * X_{ig}^{j}}{N_i} \quad \text{Equation 3.3}$$

Where N_i is the total number of data points (or sample sets) used in calculating the kernel density for PU i.

Put another way, kernel density estimation is similar to fitting a smoothly curved surface over each data point where the value of the surface is highest at the location of the point and decreases as it moves away from the point until it reaches zero at the boundary of search radius. We used ArcGIS™ in calculating the 25 attributes presented in this chapter. ArcGIS™ only supports circular neighborhoods as search radius.

DFO conducted MSS based on a random sampling design, stratified by depth, following DFO's stratification grid (Brodie 2005). We expected that the PUs created for this study would contain a varying number of sample sets. To avoid bias in density estimation, sampling effort was corrected for each planning unit (Equation 3.4). This correction was carried out by giving each sample point a value of 1 (X_j = 1), whereby a correction density layer, C_{ij}, was obtained for PU *i* and attribute *j*. This correction layer was then used to generate effort-corrected density surface for PU_i, denoted by EC_{ij}, using the ArcGIS™ *Raster Calculator* tool.

$$EC_{ij} = \frac{uD_{ij}}{\sum C_{ij}} \qquad \text{Equation 3.4}$$

Where uD_{ij} is the uncorrected density surface for attribute *j* and planning unit *i*.

3.3.3 GIS-BASED BIODIVERSITY MEASUREMENT

One of the important discussions around biodiversity conservation is how areas with high biodiversity can be identified. In response, measuring biodiversity has become an important field of inquiry, and through time the concept itself has become multifaceted (e.g. richness, evenness, beta diversity). It is difficult or impossible to reflect these many facets of biodiversity in a single index (Purvis & Hector 2000). This problem often presents issues related to how biodiversity should ultimately be represented in conservation assessment (Di Minin & Moilanen 2012). Another practical issue that arises when identifying biological objectives is information availability. In many cases, the available information cannot measure up to the urgency of protecting ecosystem function, a far-reaching goal of biodiversity protection (Jackson et al. 2001, Loreau et al. 2001, Soulé et al. 2003, Hooper et al. 2005).

In addition, some have suggested that biodiversity is hardly identifiable without inconsistency, scientific uncertainty, and ethical judgment (Svancara et al. 2005, Tear et al. 2005, Wilhere 2008). These challenges are in part due to a far smaller amount of knowledge about biodiversity compared to the enormous rate of extinction, unknown diversity, biodiversity loss, and huge socioeconomic constraints (Ehrlich & Wilson 1991, Smith et al. 1993, Smith & Robert 1993, Ehrlich

1994, Tilman 1994, Roberts & Hawkins 1999, Purvis & Hector 2000, Periera et al. 2010).

Consequently, conservation planners are still facing the critical task of trying to identify and define biodiversity objectives despite a growing number of proposed measures and concepts. In this study, we used the commonly considered indices (i.e. species richness, evenness), as these can be reasonably provided by the available data and used in the proposed decision support method presented in Chapter 4. We also included measures of species abundance (i.e. density). We do not suggest that these measures are necessarily the most appropriate measures of biodiversity in all cases. Instead, it is worth noting that a chosen method of representing biodiversity must be open for discussions for two reasons. First, as mentioned above, each biodiversity measurement technique has limitations. For example, representing biodiversity features using the strict target-based approach may provide lower biodiversity protection across biodiversity features (Di Minin & Moilanen 2012). Also, strict targets imposed upon biodiversity features may be subjective and lack strong ecological support (Marris 2007). Second, if representing biodiversity feature is considered, an optimization algorithm is an option to consider as opposed to a non-optimal method.

To quantify the number of species (i.e. richness measure) across PUs, we obtained the number of species per sample set divided by the total number of sample sets at PU *i* following the entire dataset (e.g. 12-year groundfish survey). Hence, species richness is based on the corrected sampling effort. Note that a *sample* *set* refers to a unique vessel, trip, set (VTS) number in MSS or a unique WatchID in seabird surveys.

The diversity index of attribute *j* was calculated using the Shannon-Wiener Index (H_j) (Equation 3.5).

$$H_j = -\sum_{k=1}^{s} p_{kj} * lnp_k \qquad \text{Equation 3.5}$$

The quantity p_k is the proportion of individuals found in species *k,* and the natural logarithm of p_k is lnpk. The resulting product was summed across species and multiplied by -1 to get a positive result. Then, the evenness measure (E_j), which refers to the ratio of observed diversity to maximum diversity per sample set of attribute *j*, was calculated (Equation 3.6).

$$E_j = \frac{H_j}{H_{max}} = E_j = \frac{H_j}{lnS} \qquad \text{Equation 3.6}$$

Where lnS is the natural log of species richness.

The species evenness at PU *i* was estimated using Equation 3.3. Species abundance (or density) was calculated based on weight or count of individuals. To determine the relative abundance across PUs, Equation 3.3 was used. Note that the concepts of species richness and species evenness were also used in the calculation of fishing raster-based attributes (i.e. richness and evenness of fisher groups), as discussed in Section 3.4.2.2.

3.4 METHODS: SPECIFIC GIS-BASED METHODS AND CALCULATIONS

This section describes the specific methods applied for generating each of the 25 raster-based attributes. Section 3.4.1 presents the methods used to generate the biological attributes, and Sections 3.4.2 and 3.4.3 describe the methods used for generating the socioeconomic raster-based attributes.

3.4.1 BIOLOGICAL ATTRIBUTES

Each of the five groups of marine taxa was estimated and individually mapped. Four of these groups, including groundfish, corals, and commercially exploited invertebrates, were obtained from the DFO multispecies survey program (i.e. MSS). Seabirds' data were obtained from CWS surveys, more specifically the "Programme intégré de recherches sur les oiseaux pélagiques" (PIROP) and the Eastern Canada Seabirds at Sea (ECSAS) programs.

3.4.1.1 GROUNDFISH ATTRIBUTES

All MSS data considered in this study were sampled using a standard DFO survey gear, the Campelen 1800 shrimp trawl. This shrimp trawl has a small mesh size (80 mm in the wings, 44 mm in the square and the first bellies, and 60 mm in remaining bellies), a wingspread of 15-18 m and average vertical opening of 4-5 m, depending on depths sampled (McCallum & Walsh 1997, McCallum & Walsh 2002). Similarly, as noted by Brodie (2005), MSS were standardized using a constant trawl time (15 min.), vessel speed (3 knots), and hence distance towed (0.75 NM). Brodie and Stansbury (2007) also noted other potential sources of uncertainties associated with the MSS data, such as vessel effects (i.e. different vessels were used for sampling), gaps in coverage (i.e. strata that were not sampled or reduced number of sample sets), and changes in the timing of the survey (i.e. survey was done at a later time). This study, however, did not investigate how these potential sources of error may have affected the representation of spatial attributes generated for this study.

It should be noted that this study only used the MSS datasets from 1995 and later years. Pre-1995 MSS programs used a different trawl to conduct sampling (i.e. Engel trawl) which exhibited a different catchability due it its larger mesh size compared to the Campelen trawl used in surveys in 1995 and beyond (McCalum & Walsh 1996). This gear change made it difficult to compare the data used in this study with historic or pre-1995 data. A study of historic data (until 1990), however, showed that groundfish biomass did not change prior to the 1992 collapse of the Northern Cod fishery (Gomes et al. 1995). Another study also showed that groundfish species composition and spatial distribution in the northeast Newfoundland and Labrador shelf remained stable between 1978 and 1986 (Villagarcia 1995). A total of 14,989 unique VTS or sample sets based on a 12-year MSS survey (i.e. 1995-2007) were considered for this study.

The groundfish dataset includes a total of 203 taxa, of which 55 were not identified to the species level. Nonetheless, all sample sets were used in calculating the raster-based attributes for groundfish biomass, richness, and evenness across PUs (see Equation 3.3, Figure 3.5, Figure 3.6, and Figure 3.7).

SPECIES STATUS ATTRIBUTE

In addition, particular groundfish species (i.e. endangered, threatened, and of special concern) were identified and mapped to generate another spatial attribute called species status. (See Table 3.3 for the species list and references).
It should be noted that while insufficient data was available to classify Winter Skate being at risk in the study area, it was identified in assessment reports as endangered and threatened in neighboring regions. For this reason, Winter Skate was included in our data analysis for species status (Table 3.3).

The species status attribute considered 11 species listed by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC) or identified in existing peer-reviewed literature. To account for the different levels of endangerment associated with the different species statuses, the study used a weighting scheme based on a two-step process. First, it involved ranking the species based on their status. Endangered species were assigned with the highest rank, followed by threatened species, species of special concern, and finally data deficient species. Then, this ordinal ranking of species statuses was converted into numerical weights.

To generate the numerical weights, we used a rank reciprocal method, a popular method of assessing the importance weights of dimensions of an attribute (Stillwell et al. 1981).

Table 3.3 Groundfish species for species status attribute

Species Common Name	Scientific Name	Species Status	Reference	Species Status Rank	Species Status Numerical Weights
Atlantic cod	*Gadus morhua*	Endangered	Cosewic 2003	1	0.48
Blue hake	*Antimora rostrata*	Endangered	Devine et al. 2006	1	0.48
Rock (Roundnose) grenadier	*Coryphaenoides rupestris*	Endangered	Devine et al. 2006 Cosewic 2008	1	0.48
American plaice	*Hippoglossoides platessoides*	Threatened	Cosewic 2009	2	0.24
Northern wolffish	*Anarhichas denticulatus*	Threatened	Cosewic 2001a	2	0.24
Spotted wolffish	*Anarhichas minor*	Threatened	Cosewic 2001b	2	0.24
Cusk	*Brosme brosme*	Threatened	Cosewic 2003	2	0.24
Spiny dogfish	*Squalus acanthias*	Special Concern	Cosewic 2010	3	0.16
Roughhead grenadier	*Macrourus berglax*	Endangered Special Concern	Devine et al. 2006 Cosewic 2007	1	0.48
Atlantic wolffish	*Anarhichas lupus*	Special Concern	Cosewic 2000	2	0.24
Winter skate	*Raja ocellata*	Data deficient	Cosewic 2005	4	0.12

Note: The neighboring populations of winter skate in southern Gulf of St Lawrence, Eastern Scotian shelf, and from Georges Bank to Western Scotian shelf are considered as being endangered, threatened, and of special concern respectively.

The normalized weights for different species status were used to calculate the kernel surface across PUs for species status using an averaged sum (Equation 3.3). In this calculation, the biomass or individuals per species count were not considered; rather the density calculation was estimated based on species presence

(Figure 3.8). The sum of the weights in planning unit *i* was divided by the total number of observations recorded in planning unit *i* based on 12-year MSS data.

REGIONAL ENDEMISM AND REGIONAL RARITY

The identification and representation of rarity and endemism were restricted to the study region and did not consider global ranges, hence are hereby called *regional rarity* and *regional endemism*.

The spatial attributes for regional rarity and regional endemism were generated by combining three measures of rarity. These measures are density rarity, cell occupancy, and habitat specificity. Below, we present the operational definition for these measures and the methods used in obtaining them. Figure 3.2 shows the four major steps in obtaining the attributes for regional endemism and regional rarity.

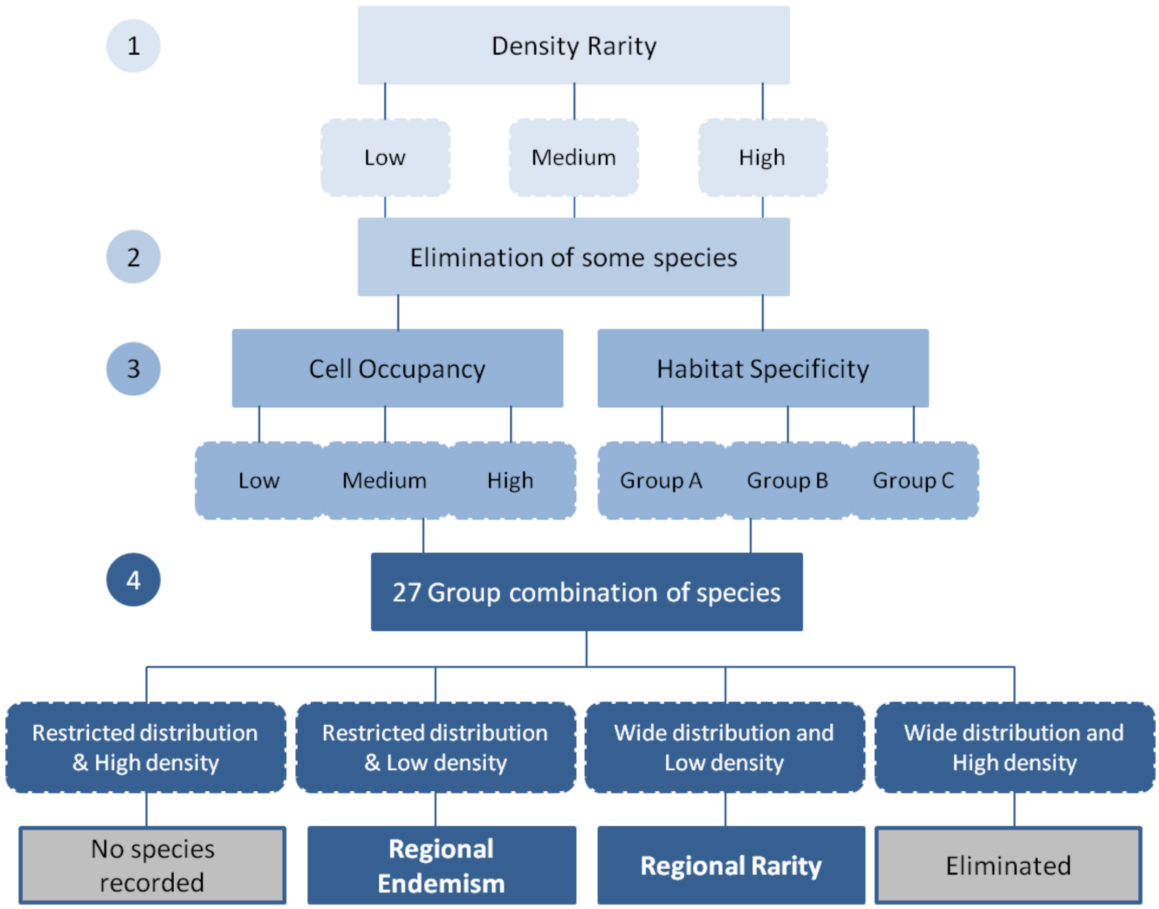


Figure 3.2 Four major steps in obtaining the regional endemism and regional rarity using three measures of rarity. Regional rarity refers to the groundfish species that are widely distributed but with low density while the regional endemism refers to groundfish species with restricted distribution and low density.

DENSITY RARITY

The count of individuals sampled for each rare species served as the basis for estimating density rarity. To date, there is no existing literature that identifies rare species for the study area. Hence, rare species were first identified from the MSS data. Identifying species rarity typically requires some rule for grouping species based on the number of individuals sampled for each species. In this study, we used a grouping technique proposed by Preston (1948) that uses a sequence of octaves of frequency. This sequence is a logarithmic series that helps determine the relative commonness of species.

Table 3.4 illustrates a sequence of octaves (1, 2, 3, etc.) based on the approximate number of individuals recorded for each species. It should be noted that this grouping technique uses Log_2. Preston (1948) demonstrated, based on empirical evidence, that the sequence of octaves follows a Gaussian curve. Hence, he suggested that the distribution of species (relative to their commonness) has a normal distribution.

Table 3.4 Sequence of octaves

Species group	1	2	3	4	5	6	7	8	Etc.
Approximate number of individuals sampled for each species	1-2	2-4	4-8	8-16	16-32	32-64	64-128	128-256	

This study generated 23 octaves for groundfish (Table 3.5). Only species that belong to the first eight octaves were considered rare species. These eight octaves were on the left of the modal octave (or highest point) of the Gaussian distribution serving as the marker between the sets of common and rare species (Figure 3.3). These eight octaves contain 56 species. From this list, however, cusk (*Brosme brosme)* was removed as it was already classified under species status. Also, species that are found in inshore areas that were inconsistently sampled were excluded.

Table 3.5 Species per octave based on the MSS Groundfish dataset

Octave (X)		Species per Octave (Y)
Number of Individuals	Species Group	
1-2	1	4.5
2-4	2	6
4-8	3	7
8-16	4	5
16-32	5	8
32-64	6	4.5
64-128	7	10.5
128-256	8	6
256-512	**9**	**11**
512-1024	10	3
1024-2048	11	9
2048-4096	12	7
4096-8192	13	7
8192-16384	14	5
16384-32768	15	8
32768-65536	16	4
65536-131072	17	8
131072-262144	18	4
262144-524288	19	2
524288-1048576	20	4
1048576-2097152	21	0
2097152-4194304	22	1
4194304-8388608	23	2

Note: The octave, in bold, is the value for the crest of the Gaussian distribution shown in Figure 3.3

These include Arctic shanny (*Stichaeus punctatus),* smooth flounder *(Liopsetta putnami),* and winter flounder *(Pseudoplueronectes americanus),* reducing the rare species list to 52.

Then, these 52 species were grouped into three levels of density rarity (i.e. low, medium, and high). Low includes species in the first three octaves (i.e. with individuals less than or equal to 8), medium includes the species in the next three octaves (i.e. with individuals greater than 8 and ≤ 64), and high includes the species in the last two octaves (i.e. with individuals greater than 64 and ≤ 256 individuals) per species.

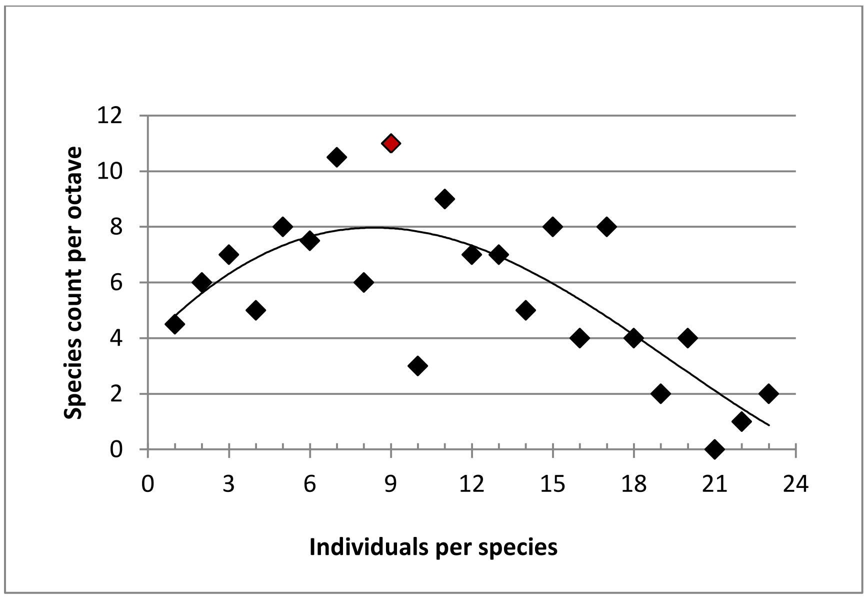


Figure 3.3 The distribution of the relative commonness (or rarity) of groundfish species. The modal octave is identified in red.

CELL OCCUPANCY

The number of PUs that a rare species occupies served as the basis for estimating cell occupancy. To determine the cell occupancy of species, the number of PUs that a species occupied (or sampled) was counted and divided by the total number of PUs for the study area. Then, the result was multiplied by 100 to obtain, in percent, the relative cell occupancy of species. Rare species, based on cell occupancy, were categorized into three groups: (Low) <0.5%, (Medium) 0.5 to ≤1%, and (High) >1%. This measure attempts to identify whether a rare species occupy a relatively large or small number of PUs.

HABITAT SPECIFICITY

With reference to the nine defined units of ecoregion, habitat specificity was defined based on the number of ecoregions occupied by species. Rabinowitz et al. (1986) referred habitat specificity to whether or not species occupy few habitats. In the absence of detailed studies on habitats in the study region, existing ecoregions were used as a surrogate for regional habitats. Two sets of defined units of marine ecoregion for the study area were identified in two separate studies: (a) Pepin et al. (2010) identified 5 ecoregion units for most of the Newfoundland and Labrador bioregion and (b) Perez-Rodriguez et al. (2010) identified two additional defined units in the Flemish Cap region following the same method employed by Pepin et al. (2010). It must be noted that some portions of the study area fall outside these identified seven ecoregion units. Subsequently, our study added two other units covering the Northwest Atlantic Fisheries Organization (NAFO) zones 3Pn, 3Ps, and 4Vn. The delineation of these NAFO zones into two units was based on the two unique ecological environments described for these areas in Mahon et al. (1988) (Figure 3.4).

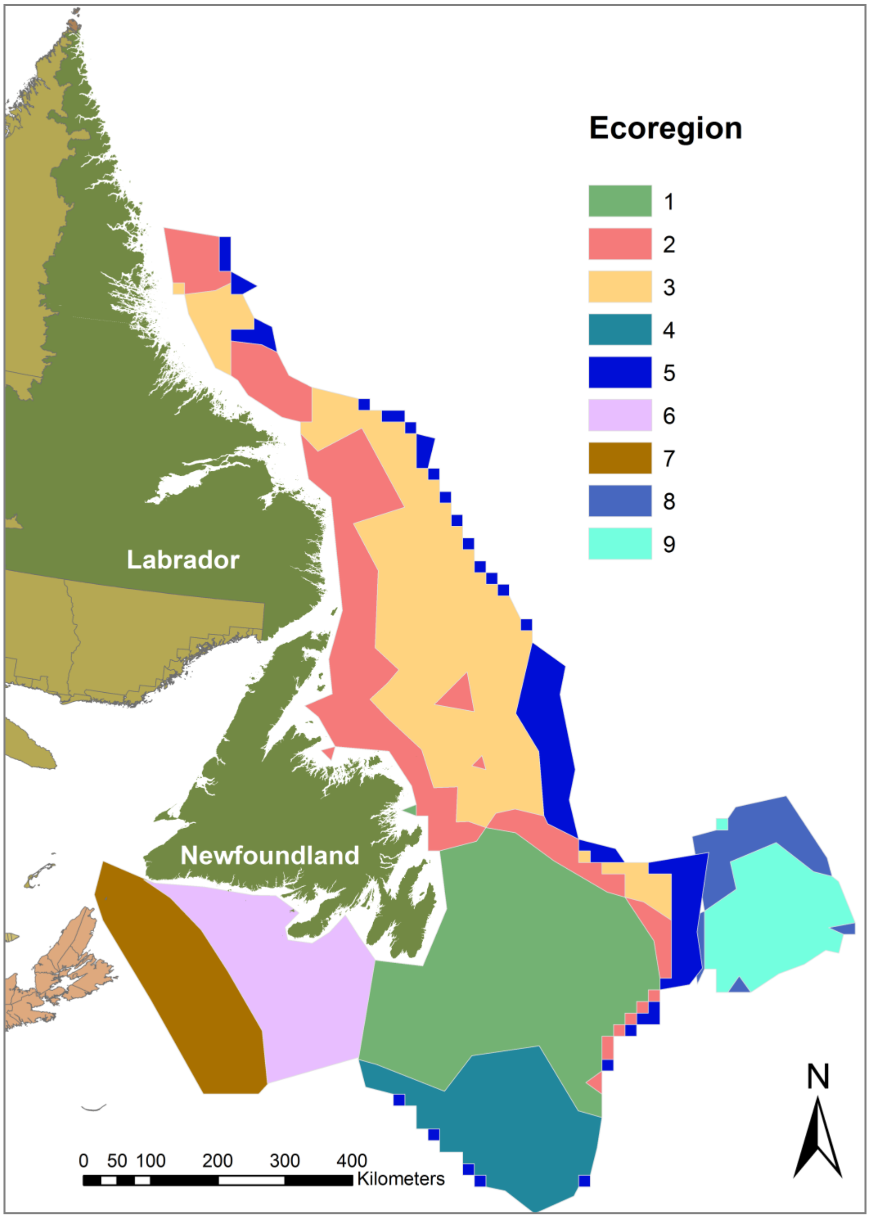


Figure 3.4 Nine ecoregion units used in this study. Units 1-5 and 8-9 were respectively identified and described in Pepin et al. (2010) and Perez-Rodriguez et al. (2010) while units 7 and 6 were added in this study.

This study defined habitat specificity based on the number of ecoregion units occupied by species. Then, groundfish species were classified into three groups: (Group A) species that occupy the least number of habitats (i.e. one unit of the ecoregion), (Group B) species that occupy two habitats (i.e. two units of the ecoregion), and (Group C) species that occupy three or more habitats (i.e. 3 or more units of the ecoregion).

The different groups of species obtained from the three measures of rarity described above were then combined. Twenty-seven unique group combinations of species are possible, although 15 groups were empty of species (Table 3.6).

Table 3.6 Group combinations of species based on density rarity, cell occupancy, and habitat specificity

Restricted distribution and high density (Class 1)	Restricted distribution and low density (Class 2)	Wide distribution and low density (Class 3)	Wide distribution and high density (more common) (Class 4)
*HαA	LαA	*LαC	*HαC
*HαB	LαB	LβC	HβC
*HβA	*LβA	MαC	HγC
*HβB	*LβB	MβC	
*HγA	MαA	*LγC	
*HγB	MαB	*MγC	
	MβA		
	MβB		
	*LγA		
	*LγB		
	*MγA		
	MγB		

In each group combination, the three letters are arranged accordingly representing density rarity, cell occupancy, and habitat specificity. The first letter refers to density rarity where count of individual per species is considered as: **L -** low, **M -** medium, and **H –** high. The second letter refers to cell occupancy where species is considered occupying: **α –** low, **β** – medium, and **γ -** high number of PUs. The third letter refers to habitat specificity where species occupy: **A -** 1 ecoregion, **B -** 2 ecoregions, and **C -** ≥3 ecoregions. *Group combinations with no species found.

Then, these 27 group combinations were further categorized into four classes: (Class 1) restricted distribution and high density, (Class 2) restricted distribution and low density, (Class 3) wide distribution and low density, and (Class 4) wide distribution and high density (Table 3.6). Based on the datasets used for this study, no species was listed under Class 1. The second class was referred to as "regional endemic," containing 24 species and the third class was referred to as "regional rare" containing 22 species. The last class is composed of six species considered as being more common; hence, they were excluded (Table 3.7).

Table 3.7 Classification of 52 rare species identified from MSS fish dataset

Regional Endemic	Regional Rare	Relatively Common
Acipenser oxyrhynchus	*Cryptosaras couesi*	*Gadus ogac*
Alepisaurus brevirostis	*Nansenia groenlandica*	*Anoplogaster cornuta*
Alosa pseudoharengus	*Raja laevis*	*Cottunculus thompsoni*
Benthodesmus simonyi	*Alepisaurus brevirostis*	*Dibranchus atlanticus*
Brevoortia tyrannus	*Bathypterois dubius*	*Raja lintea*
Caranx crysos	*Halargyreus johnsonii*	*Scomberesox saurus*
Gasterosteus wheatlandi	*Raja bathyphila*	
Himantolophus groenlandicus	*Raja mollis*	
Idiacanthus fasciola	*Rhectogramma sherborni*	
Molva brykelange	*Rhinochimaera atlantica*	
Parasudis truculentus	*Saccopharynx ampullaceus*	
Platytroctes apus	*Somniosus microcephalus*	
Raja erinacea	*Synodus poeyi*	
Apeltes quadracus	*Anotopterus pharao*	
Aphanopus carbo	*Caristius groenlandicus*	
Arctogadus glacialis	*Ceratius holboelli*	
Lipogenys gillii	*Coelorhynchus carminatus*	
Myoxocephalus scorpioides	*Eurypharynx pelecanoides*	
Nessorhamphus ingolfianus	*Hydrolagus affinis*	
Urophycis chuss	*Micromesistius poutassou*	
Argentina striata	*Petromyzon marinus*	
Diretmus argenteus	*Raja hyperborea*	
Centroscymnus coelolepis		
Lepidion (haloporphyrus) eques		

Kernel density analysis was used to generate the raster-based attributes (i.e. relative densities) of regionally endemic (Figure 3.9) and regionally rare species (Figure 3.10) for the study area. The kernel function was based on species presence, whereby each record of species was given a value of one using a 15 km search radius (Equation 3.1). It should be noted that the area unit for the output density values was set to square kilometers in ArcGIS™, resulting in relatively small values at the PU scale.

3.4.1.2 SEABIRD ATTRIBUTES

Unlike the MSS data, the spatial attribute for seabirds represents a combination of both historical and recent surveys acquired through the CWS PIROP and ECSAS survey programs (Table 3.1). The datasets obtained from the CWS Atlantic Region included a standardized calculation of all sample sets (K. Allard, personal communication). These standard estimates were based on the number of seabirds per km^2. Each sample set was identified in the dataset using a unique WatchID. The entire seabird dataset contains 11,600 sample sets from ECSAS and 11,643 sample sets from PIROP.

PIROP observations covered 1965-1990 (although most records were after 1970), while ECSAS records covered 2006-2011. PIROP recorded observations (counts of birds) based on two approaches (Gjerdrum et al. 2012). The first one, used in earlier surveys, involves 10-minute observations during which all birds were counted without considering a distance from a vessel. The second approach recorded birds within a particular band transect (300 m), scanning at 90° arc to one side of the ship, allowing the calculation of seabird densities (i.e. birds per square kilometer).

ECSAS implemented a standardized sampling protocol both for moving (e.g. vessels) and stationary platforms (e.g. oil rig) (Gjerdrum et al. 2012). Seabirds were recorded within a 300 m band transect, scanning at a 90° arc to one side of the ship and implemented a five-minute snapshot approach to capturing the flying birds. It should be noted that ECSAS is similar to the second approach under PIROP, except for the protocol implemented for flying birds.

To combine the sample sets from these two surveys that used different methods, the seabird count for both surveys was standardized based on a kilometer traveled per sample set. From this standardized count, the raster-based attribute for seabird density, richness, and evenness were calculated using Equation 3.3. For the respective attribute maps, see Figure 3.11, Figure 3.12, and Figure 3.13.

3.4.1.3 SPONGE ATTRIBUTE

Sponge data were obtained from the DFO MSS dataset. Sponges were recorded only as by-catch wet weight for the period 1995 to 2011. Records included weights in kilograms for each observation but did not identify sponges to the species level. Based on this limitation, the raster-based attribute for sponge was based only on aggregated biomass (Equation 3.3, Figure 3.14).

3.4.1.4 CORAL ATTRIBUTES

Cold-water coral data were also obtained from the DFO MSS dataset. Like sponges, corals sample sets were recorded as by-catch for the period of 2000 to 2010. Two types of records were provided: count or weight recorded at sea, and count or weight of sample sets on land, as recorded at DFO - St. John's, NL (V.E. Wareham, personal communication). While both measurements should be identical, they have differences. In response, the highest weight recorded either at sea or on land was used in this study. Data on weights are believed to be more reliable than count (V.E. Wareham, personal communication). Also, more sample sets containing "weight" data were found than sample sets containing count data. As a result, a raster attribute based on coral biomass (weight) was generated (Equation 3.3, Figure 3.15). For calculating the raster-based attribute for coral species richness, only those coral samples identified to the species level were included in the calculation (Equation 3.3, Figure 3.16).

Based on a previous study on coral growth rates (Sherwood & Edinger 2009), this study grouped coral species according to their level of sensitivity to damage, as a function of longevity. Then, sensitivity scores were assigned to the following groups: hard bottom gorgonian, hard bottom anthipatharian, hard bottom cup coral, soft bottom gorgonian, sea pens, soft bottom cup coral, and soft coral (Table 3.8). To determine the raster-based attribute for coral sensitivity, the sensitivity scores of coral samples were summed and divided by the total number of sample sets (N) for planning unit *i* (Equation 3.3).

Table 3.8 Sensitivity score assigned to each coral group

Coral Groups	Samples	Sensitivity Score
Hard bottom gorgonian	*Acanthogorgia armata* *Anthothela grandiflora* *Keratoisis grayi* *Paragorgia arborea* *Paramuricea grandis* *Primnoa resedaeformis*	5
Hard bottom antipatharian	*Antipatharian [ORDER]* *Bathypathes sp.* *Stauropathes arctica*	5
Hard bottom cup coral	*Fungiacyathus marenzelleri* *Scleractinia [ORDER]* *Vaughanella margaritata*	4
Soft bottom gorgonian	*Acanella arbuscula* *Chrysogorgia cf. agassizii* *Radicipes gracilis*	4
Seapen	*Anthoptilum grandiflorum* *Dischoptilum gracile* *Funiculinia quandrangularis* *Halipteris finmarchica* *Parastenella atlantica* *Pennatula phosphorea* *Pennatula sp.* *Umbellula lindahli*	3
Soft bottom cup coral	*Flabellum alabastrum* *Flabellum angulare* *Flabellum sp.*	2
Soft coral	*Duva florida* *Gersemia rubiformis,* *Nephtheid [FAMILY]* *Anthomastus sp.* *Heteropolypus insolitus* *Anthomastus grandiflorus* *Anthomastus agaricus* *Drifa glomerata* *Drifa sp.*	1

3.4.1.5 INVERTEBRATE ATTRIBUTES

Shrimp and crab are marine invertebrates currently being exploited in major commercial fisheries in the province of Newfoundland and Labrador, Canada (DFO 2004). Because of the relatively high commercial significance of these invertebrates for the study area, separate attributes were obtained for shrimp and crab, rather than being simply combined into one attribute such as landed biomass. Generating the attributes for each of these species was important from the standpoint of sustaining these species via conservation options. Shrimp species in the datasets include *Pandalus sp., Pandalus borealis, Pandalus montagui*, and *Pandalus propinquus*. However, most observations are from *P. borealis* and *P. montagui,* the two commonly exploited species for commercial purposes. Records for crab are solely for snow crab *(Chionoecetes opilio)*. All MSS observations from 1995 to 2011 were combined when calculating the biomass for shrimp and crab (Equation 3.3). Hence, weights (kg) served as the basis for kernel density calculations to generate the raster-based attributes for shrimp (Figure 3.18) and crab (Figure 3.19).

3.4.2 FISHING ATTRIBUTES

So far, there has been little exploration as to the various ways of measuring socioeconomic gain. In most SCP methods, this is typically measured using aggregated fishing effort. This study, however, derived from fishing activities six raster-based attributes whereby economic (i.e. monetary) gain was measured separately from socially-oriented (e.g. fishing employment, fishing business) gain. The fishing attributes include: (a) landed catch based on weight, (b) fishing revenue, (c) fishing business based on unique vessels, (d) fishing employment based on crews, (e) richness distribution of fisher groups (FGs), and (f) evenness distribution of FGs. It should be noted that the richness and evenness distribution of FGs were calculated in similar ways, as species richness and species evenness respectively, as presented below.

The fishing activities were obtained from the commercial fishing data (i.e. the DFO logbook) from DFO-St. John's, NL. The DFO logbook contains information on fishing trips, such as location (or the geographic coordinates for the "start" and "end" of the fishing trip), species caught, including weight and landed monetary value, the length of fishing time, and the size of the vessels. Logbook datasets excluded fishing vessels under 18 feet long (e.g. the lobster fishery) that were hence excluded from the analysis. All socioeconomic data layers were generated from a combined set of data points covering 2001 to 2010. The following subsections present the two groups of fishing attributes used in the study.

3.4.2.1 LANDED CATCH

This study considered two sets of information from the DFO logbook for measuring the economic gain from fishing activities. These include the weight (kg) and the dollar value (C$) of landed fish. From these two sets of information, we respectively derived the raster-based attributes for landed catch based on biomass (Figure 3.20) and fishing revenue (Figure 3.21). It should be noted that landed fish refers to both shellfish, such as crab and shrimp, and groundfish.

Both layers used a prorated calculation made by DFO-Policy and Economics Branch in St. John's, NL. Table 3.9 describes the calculation of prorated value and prorated weight of landed catch.

Table 3.9 Example calculation of prorated weights of landed catch

Estimates at sea (e.g. Redfish)		
Date	Weight (kg)	Prorated weight (kg)
June 10	10,000	9,772
June 11	12,000	11,726
June 12	15,000	14,658
June 13	22,000	21,498
June 14	8,000	7,818
June 15	31,000	30,293
June 16	20,000	19,544
Total	118,000	115,309

Actual amount landed: 115,310 kg. The factor used to calculate the prorated weight of fish is obtained by dividing the actual weight of fish landed at port by the total estimated weight recorded in the logbook. For instance, 115,310/118,000 = 0.9772. N.B. Weights of daily fish catches are then obtained by multiplying the estimated weights by the proration factor. Example for June 10: 10,000 * 0.97720 = 9,772 kg. The same calculation was applied in determining prorated value (C$). Courtesy of Anne-Marie Russell, DFO Policy and Economics Branch, St. John's, NL.

The fishing biomass and fishing revenue per PU were averaged based on the number of trips recorded for each PU over the 10-year commercial fishing history spanning from 2001 to 2010 (Equation 3.3). Thus, the landed catch based on biomass and monetary value at PU *i* refer respectively to the average weight and Canadian dollar value of landed catch per trip. CPUE calculation was considered, but the use of fishing per trip provides a straightforward description of the landed catch. Defining effort using the logbook information present challenges due to the varying motor power, vessel size, vessel type, and number and types of gears used in actual fishing.

3.4.2.2 DISTRIBUTION OF FISHING BUSINESS AND FISHERS

The distribution of fishing business and fishers can provide insight as to how space-based conservation options such as MPAs impacts fishers and fishing vessel owners at sea.

In this regard, this study generated one attribute that estimates the spatial distribution of fishing business and three attributes that can determine the distribution of fishers using the logbook data. Below we present the methods in obtaining these attributes and their relevance in capturing the socioeconomic activities at sea.

FISHING BUSINESS

Each fishing business or business ownership was represented using a unique vessel code per PU. The reason for doing this is the assumption that each vessel represents a single fishing business. The fishing business layer intends to estimate the distribution of fishing business across PUs. A single vessel can, however, use several types of fishing licenses, and different vessels can be part of the same fishing business. Such information was however not made available by DFO for this study for privacy reasons. However, this attribute can identify areas that are important to the majority of fishing businesses, irrespective of individual ownership and licenses. In the study area, this is potentially useful in identifying PUs relevant to smaller-scale and larger-scale fishing enterprises.

From the logbook data, we generated a modified set of data whereby a unique vessel ID was counted only once, in a PU, through a random selection of latitude and longitude coordinates. This rule was implemented for PUs with multiple records of the same vessel ID based on the 10-year logbook data. The resulting set of data was then used to estimate the relative densities of fishing business and fishing employment across PUs.

In determining the raster-based attribute for fishing business, the unique fishing vessels were first represented in each PU with one geographic data point that was selected randomly. Then, the resulting dataset was used in calculating the kernel density surface (Equation 3.2, Figure 3.22). Each data point was given a value of one. The area unit of analysis and the search radius were respectively set to square kilometer and 15 km. Hence, the values shown on the map refer to the number of unique vessels per square kilometer relative to each PU. We should note that the attribute for fishing business assumes that each vessel, whether large or small, is given the same level of importance.

FISHING EMPLOYMENT

In identifying the distribution of fishers, we considered mapping the distribution of fishing employment based on direct engagement with fishing activities at sea (i.e. crew members per fishing vessel) to identify PUs with high fishing employment. In protected area planning, making this information available can be useful in leveraging the short-term socioeconomic impacts against protecting biodiversity.

The number of crew per vessel was derived from the vessel size using multipliers (Table 3.10). Multipliers refer to a standard approximation of crew members based on the five categories of vessel size. A set of multipliers has been used in survey analysis conducted by DFO (DFO 2005). In this study, however, we used an updated set of multipliers (Table 3.10) obtained from the DFO regional office in St. John's, NL (S. Allen, personal communication). This new set of multipliers provides an improved estimate of vessel crew compared to the older set (B. Best, personal communication).

Table 3.10 Multipliers for vessel crew members per vessel category. Multipliers include the skipper (S. Allen, pers. comm.)

Vessel length (ft)	Multipliers (vessel crew)
Under 35 feet	1.75
35-44	2.50
45-64	5.50
65-99	7.00
Over 100 feet	14.00

The set of geographic points used in mapping the fishing employment is the same as those used when generating fishing business distribution in which each unique vessel is represented only once in a unique PU. This layer provides an estimate of the relative number of fishers that are likely directly employed per PU. Using the original logbook data points could introduce counting the same set of crew members more than once if they happen to be in that PU more than once. Kernel density calculations used the multipliers in generating the kernel surface (Figure 3.23). Hence, fishing employment for planning units was determined by obtaining the sum of the fishing vessel crews (estimated through multipliers) per square kilometer in planning unit *i* (Equation 3.2).

FISHER GROUPS RICHNESS AND EVENNESS

Another way to approximate the distribution of fishers in the study area is to classify them into groups based on major fisheries and vessel size, hereby called fisher groups (FGs). Six major commercial fisheries were identified by directed species or species groups with the expert advice from the Fish, Food and Allied Workers (FFAW) (K. Sullivan, personal communication). These major fisheries included: crab, shrimp, cod, other groundfish, other shellfish, and other pelagic species.

Table 3.11 Fisher groups used for calculating richness and evenness of fisher groups

Fisher Categories	Fisher Groups
1	Inshore crab
2	Inshore cod
3	Inshore shrimp
4	Inshore other groundfish
5	Inshore other shellfish
6	Inshore pelagics
7	Nearshore crab
8	Nearshore cod
9	Nearshore shrimp
10	Nearshore other groundfish
11	Nearshore other shellfish
12	Nearshore pelagics
13	Midshore cod
14	Midshore crab
15	Midshore shrimp
16	Midshore other groundfish
17	Offshore crab
18	Offshore cod
19	Offshore shrimp
20	Offshore other groundfish
21	Offshore other shellfish

The range of vessel sizes considered was: (1) inshore (18-34 ft), (2) near shore (35-64 ft), (3) midshore (65-99 ft), and (4) offshore (≥ 100 ft). Combining the four categories of vessels and six major fisheries, 21 FGs were identified from the commercial fishing dataset (Table 3.11).

These groups were subsequently used when calculating the richness and evenness of FGs. These two measurements were computed in a similar way as the species richness and species evenness discussed in Section 3.3.3. For comparison purposes, the definitions of FG-Richness and FG-Evenness, as applied to social and biological datasets, are described below. It should be noted that fisher groups and fishers were used analogously with species and individuals belonging to a species respectively.

In biological datasets:

Species richness refers to the average number of species per sample set per PU (Equation 3.3). *Species evenness* refers to the ratio of observed diversity to maximum diversity per sample set (Equation 3.5). Across PUs, an average value for evenness (E_H) was obtained (Equation 3.3).

In fishing datasets:

FG-Richness refers to the sum of unique *fisher groups* per PU for the period 2001-2010 (Equation 3.2). It should be noted that the density of FG-Richness in PU *i* uses sum while species richness uses average. The reason for this is that unique FGs were identified for each PU prior to the density analysis. In contrast, unique groundfish species was determined for each sample set (not PU) to which obtaining the average was necessary for correction purposes. *FG-Evenness* refers to the ratio of observed diversity of FGs to maximum FGs diversity per PU.

To calculate the FG-Evenness, the FG-diversity was quantified first based on the Shannon-Wiener index (Equation 3.4), then re-expressed through the following equation.

$$H_f = -\sum_{f=1}^{n} pf * \ln pf \qquad \text{Equation 3.7}$$

The quantity p_f is the proportion of fishers found in fisher group *f,* and ln*pf* is the natural logarithm of this ratio. The resulting product was summed across FGs and multiplied by -1 to get a positive outcome. Then, a measure of FG-Evenness (E_f) was calculated using Shannon's equitability (Equation 3.5), re-expressed as the following equation:

$$E_f = \frac{H_f}{H_{max}} = E_f = \frac{H_f}{lnF} \qquad \text{Equation 3.8}$$

where lnF is the natural log of FG-Richness.

The relative density of FG-Evenness across PUs uses summation (Equation 3.2). Accordingly, FG-Richness can be used to approximate the relative importance of PUs with respect to the number of fisher groups interested in each PU (Figure 3.24). PUs with higher FG-Richness score implies that these PUs support more fisher groups. On the other hand, FG-Evenness offers a relative importance of PUs with respect to the equitability among FGs that are interested in each PU (Figure 3.25). PUs with higher FG-Evenness score implies that they support an equal (or nearly equal) proportion of fishers that belongs to various FGs.

In general, the above four attributes aim at identifying planning units that are essential for socioeconomic activities but not necessarily based on monetary gain. They highlight social benefits in order to capture broader facets of socioeconomic benefit per PU.

3.4.3. ATTRIBUTES FOR OTHER HUMAN USES

Four raster-based attributes representing other human uses were generated for the study area: (1) gear impact, (2) marine commercial traffic, (3) oil and gas activities, and (4) gear conflict.

3.4.3.1 GEAR IMPACT

PUs may have different levels of negative ecological impact caused by fishing gears, and those highly impacted should be minimized in conservation scenarios based on the concept of naturalness (Callicot 1998, Angermeier 2000, Willis & Birks 2006). To determine negative ecological impacts of fishing gears across PUs, we used the severity scores or negative ecological impact of fishing gears for each PU based on 10-year fishing dataset (Figure 3.26). The severity score was adopted from Fuller et al. (2008).

Gear impacts were calculated based on the four fishing gears having the highest severity scores, being bottom trawl (98%), bottom gillnet (79%), dredge (74%), and bottom longline (62%). These percent severity scores were then converted into decimal severity scores (i.e. 0.98, 0.79, 0.74, and 0.62). Decimal severity scores corresponding to each record of fishing gear were summed for each PU based on the 10-year logbook records. Then, based on the sums of severity scores the relative densities of PUs were generated using Equation 3.2. Thus, the negative ecological impact of fishing gear is the sum of gear severity scores at PU *i*.

3.4.3.2 MARINE COMMERCIAL TRAFFIC

Identifying areas for marine commercial traffic is an important consideration in siting conservation areas (Crowder et al. 2006, Douvere 2008). In this study, we determine these areas using the long range identification and tracking (LRIT) data spanning from February 2010 to February 2011. LRIT is a system that tracks vessels on an international voyage including all passenger ships, mobile offshore drilling units, high-speed craft, and cargo ships of ≥ 300 gross tonnages. These ships are required, under the Safety of Life at Sea (SOLAS) convention, to send out their geographic locations to the LRIT Data Center on a 6-hour interval basis (UN 1980, Maritime Safety Committee 2008). We acquired these data through the Maritime Security Branch of the Canadian Coast Guard.

To determine the raster-based attribute (or density) of marine commercial traffic, via the one-year LRIT data, we obtained the length (in kilometers) of vessel tracks per km^2 for each PU. This process was carried out by first converting the 6-hour geographic points into lines based on the date/time stamp for each commercial vessel using the *Point to Line* tool in ArcGIS™ 10. Then, the relative densities of all commercial vessel tracks *(MT)* in PUs were calculated using the *Line density* tool in ArcGIS™ 10 (Figure 3.27). Search radii were set to 10 km (Equation 3.9).

$$MT_i = \frac{\sum_{q=1}^{n} K_{iq} * V_{iq}}{SR} \qquad \text{Equation 3.9}$$

Where K_{iq} refers to the length (km) of vessel track q intersecting the search radius for planning unit i, and V_{iq} refers to the value for each vessel track in PU i. In this study, all vessel tracks were equal, hence, V_{iq} was given with a value = 1. SR refers to search radius used in the data analysis (i.e. 10 km). Hence, MT_i (marine commercial traffic for planning unit i) was expressed as length of vessel tracks in kilometers per km^2.

3.4.3.3 OIL AND GAS ACTIVITIES

Oil and gas activities (O&G) have been significant economic activities in the study area (Sawyer & Stiebert 2010). In this study, the densities of O&G activities were estimated from the locations of O&G wells and several types of licenses. The geographic datasets obtained from the Canada-Newfoundland and Labrador Offshore Petroleum Board (C-NLOPB) represented O&G wells as points, while the O&G licenses (i.e. production, significant discovery, and exploration) were represented using polygons to show area coverage.

To determine the raster-based attribute of O&G activities across PUs, the following procedures were used: (1) polygons (representing areas for O&G licenses) were converted to points, (2) points were merged and weighted, and (3) the resulting point layer was used to approximate the density of O&G activities using kernel density.

First, we observed that the O&G wells serve as a useful proxy in indicating the spatial extent of current O&G areas, particularly for the production and significant discovery licenses. Some exploration areas, however, do not contain wells. To account for these areas, we converted all polygons to points using the *Feature to Point* tool in ArcGIS™ 9.2. This tool creates a point to represent each polygon. While this conversion may work well when polygons have similar or relatively small area size, it creates a limitation when the area size of some polygons exceeds the mean polygon size.

This restriction of this procedure is noticeable in the data layer for the exploration license whose average area amounts to 1.58 km^2, but having few large areas (e.g. 11.6 km^2, 5.3 km^2). Accordingly, these few vast areas would not be fairly represented in the density calculation. Second, the points generated from polygon layers (i.e. for exploration, production, and significant discovery) were merged to the data points representing the geographic locations of O&G wells. Then, the various O&G activities were ranked and weighted using the rank reciprocal method (Table 3.12). The weights were subsequently used to generate the weight for each of the data points. Third, the weight of the data points served as the basis for calculating the densities of O&G activities for each PU using kernel density (Equation 3.2, Figure 3.28).

Table 3.12 Weighted scores of O&G Activities

Activities	Rank	Weight
Production	1	0.48
Wells	2	0.24
Significant discovery	3	0.16
Exploration	4	0.12
Total		1.0

3.4.3.4 GEAR CONFLICT

The issue on gear conflict has raised policy issues and concerns in managing marine resources (Kangas et al. 2012, Kaiser 2014). The overlap between mobile and fixed gears (i.e. fixed gears and shrimp gears) is known to be an issue in the study area, such as in the Hawke channel (DFO 2002). Such overlap can for instance become an issue when mobile bottom trawls affect fixed gears. From a standpoint of siting protected areas such as MPAs, these areas with potential conflict issues can either be minimized or maximized in conservation areas. They can be kept to a minimum, if, for example, they may require another type of restriction (e.g. closing the areas for mobile bottom gear) as in the Hawke Channel case (DFO 2002). However, they can be maximized if setting these areas for conservation purposes could be an appropriate resolution.

To determine PUs that may have potential gear conflict issues, we mapped the spatial overlap of these fishing gears using a two-step procedure. First, the spatial distribution of shrimp and crab gears was calculated separately based on effort (vessel length, feet) using the 10-year commercial fishing data points. Second, the spatial overlap between these effort calculations was estimated using the logical AND operation that returns the Boolean value true (presence of spatial overlap) if both operands (i.e. quantities, other than zero, for both gears) are present otherwise, it returns false (absence of spatial overlap).

We used MCE-FLOWA to carry out this computation, a software extension developed for ArcGIS™ 9.2 (Equation 4.3, Boroushaki et al. 2008). Prior to calculating the spatial overlap, both datasets were normalized from 0 to 1. Hence the gear conflict issues across PUs ranges from 0-1, with 0 representing the absence of spatial overlap and 1 representing complete spatial overlap (Figure 3.29).

3.5 RESULTS

This section presents the raster-based attributes representing biological diversity based on five marine taxa, fishing-based activities and other human uses identified for the study area. Since the attributes are in the form of maps, the dominant spatial patterns observed are highlighted in the presentation of the 25 raster-based attributes.

3.5.1 GROUNDFISH

The highest groundfish biomass is observed on the southern portion of Newfoundland and the Grand Banks. A few scattered PUs with high value appears in other NAFO areas as well (Figure 3.5).

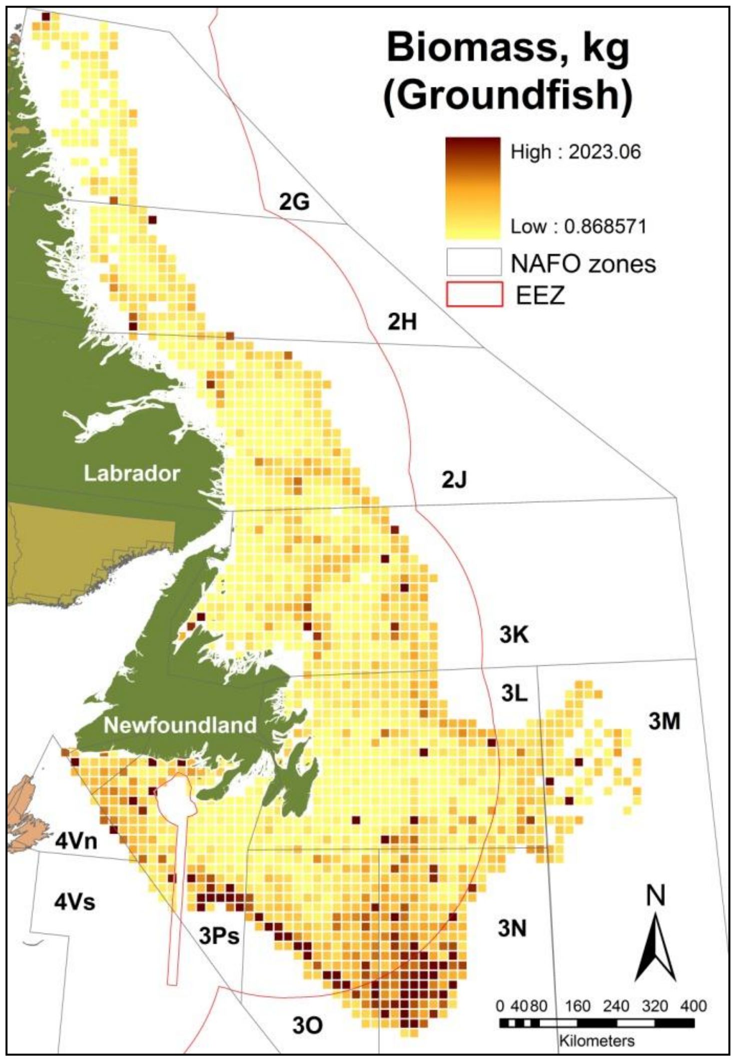


Figure 3.5 Groundfish biomass (kg), expressed as average, per sample set in a planning unit. Source Data: DFO MSS, 1995-2007.

The groundfish species richness is relatively high along the continental shelf edge, including the Flemish Cap region, a portion of NAFO 3M. Some high values are also observable in NAFO 3K, 3L, and 2J. The southern portions of Newfoundland particularly around NAFO 3Pn (see Figure 3.1 for the reference of this area) and 3Ps have moderate values (Figure 3.6).

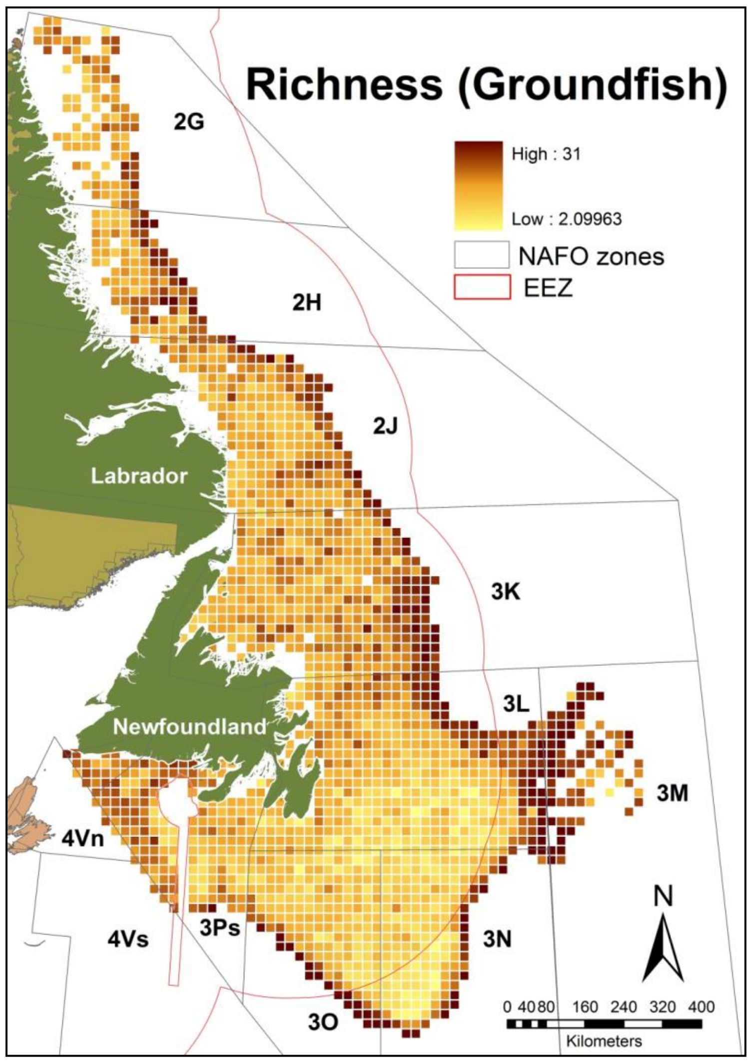


Figure 3.6 Groundfish richness expressed as the average number of groundfish species per sample set in a planning unit. Source Data: DFO MSS, 1995-2007.

The species evenness for groundfish species shows relatively low evenness in the Grand Banks region (NAFO 3O, 3N, and portion of 3L). The rest of the PUs show relatively high values, particularly the NAFO zone 3Ps, Flemish Pass (situated at the boundary between 3L and 3M), the lower portion of NAFO zone 3K, and the upper part of NAFO zone 3L (Figure 3.7).

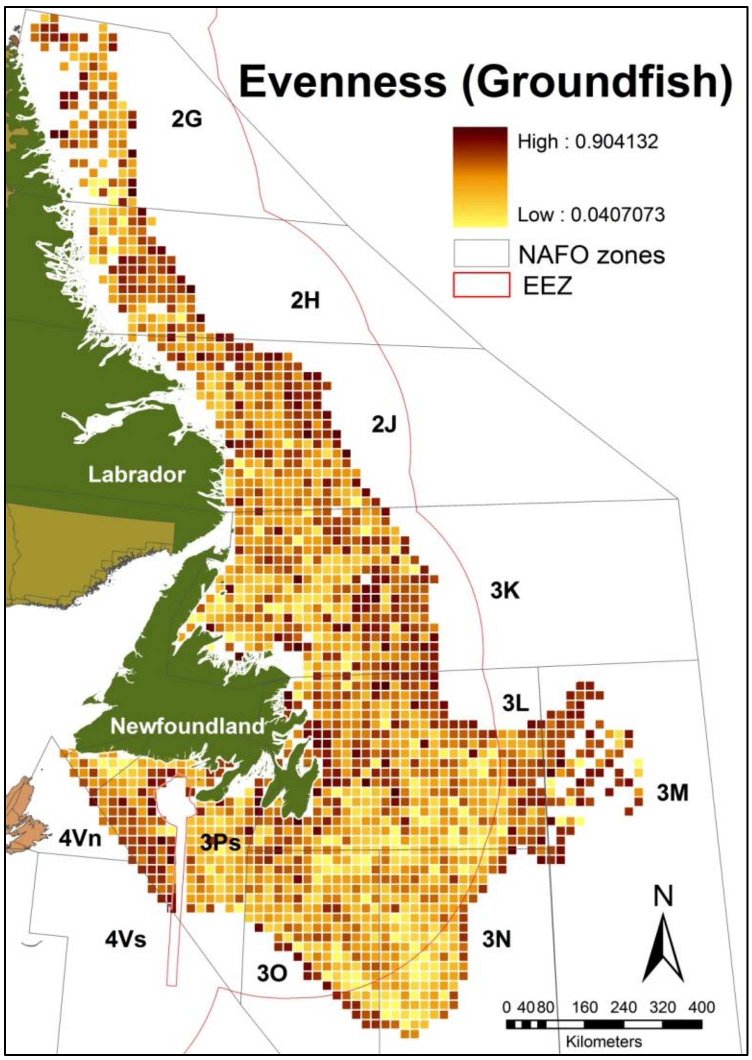


Figure 3.7 Evenness of groundfish species, expressed as average, per sample set in a planning unit. Source Data: DFO MSS, 1995-2007.

The relative densities of PUs with regard to the species status (i.e. composed of species classified as endangered, threatened, and of special concern) show that all the shelf edge from NAFO zone 2G down to 3Ps and nearshore areas in southern Labrador and northern Newfoundland are critical locations to species that are likely at risk (Figure 3.8).

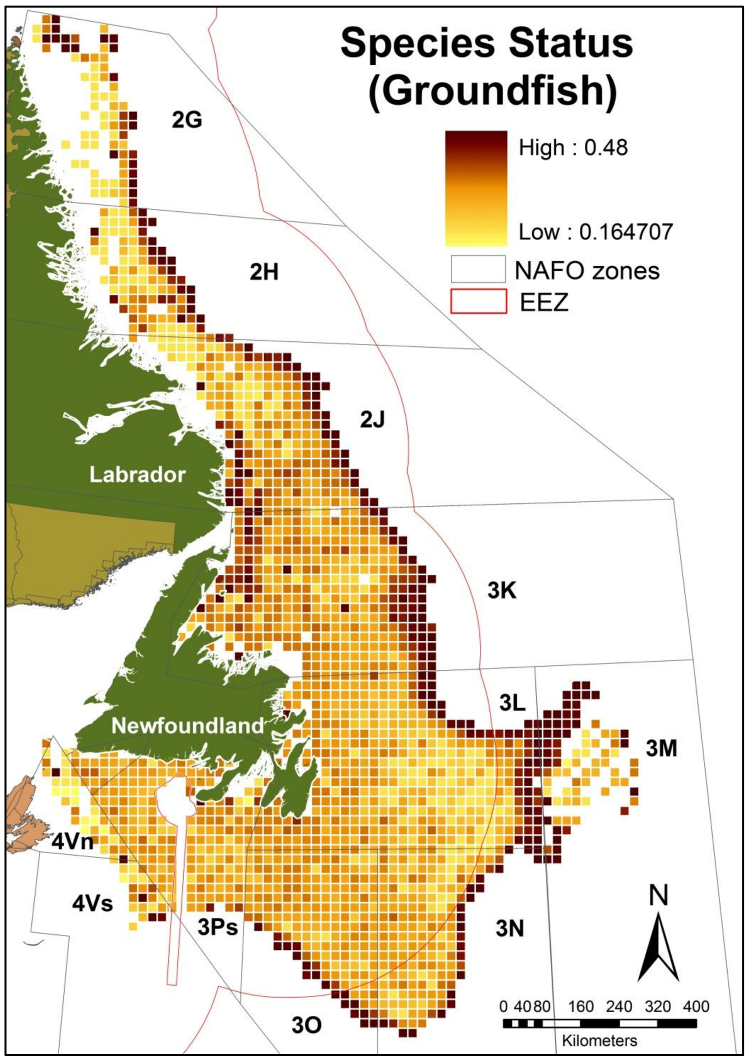


Figure 3.8 Species status expressed as average, per planning unit, of numerical weights assigned to three species status (i.e. endangered, threatened, of special concern) and one data deficient species. Source Data: DFO MSS, 1995-2007.

High concentration of regional endemic groundfish species tends to occur in patches at the shelf edge, particularly from NAFO zones 2H down to 3Ps, and on the Flemish Pass and around the edge of Flemish Cap in NAFO 3M. A few PUs in the southern portion of Newfoundland also show importance for regional endemic species (Figure 3.9).

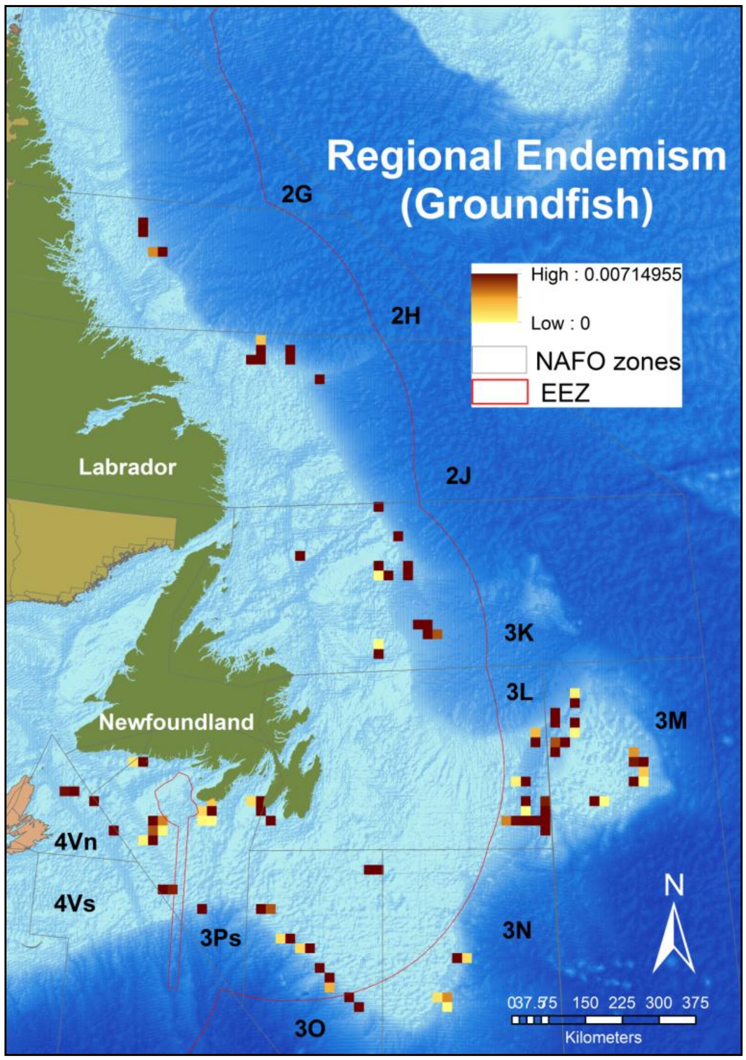


Figure 3.9 Presence of regionally endemic groundfish species per square kilometer in each planning unit. Source Data: DFO MSS, 1995-2007. GEBCO bathymetry in light to dark blue indicates shallow to deep areas.

High concentration of regionally rare species also appears along the shelf edge from NAFO 2H down to 3Ps. Relatively higher concentrations of PUs relevant to regional rare species are observable along the shelf edge of NAFO 3K, 3L, and 3M particularly within Flemish Pass (Figure 3.10).

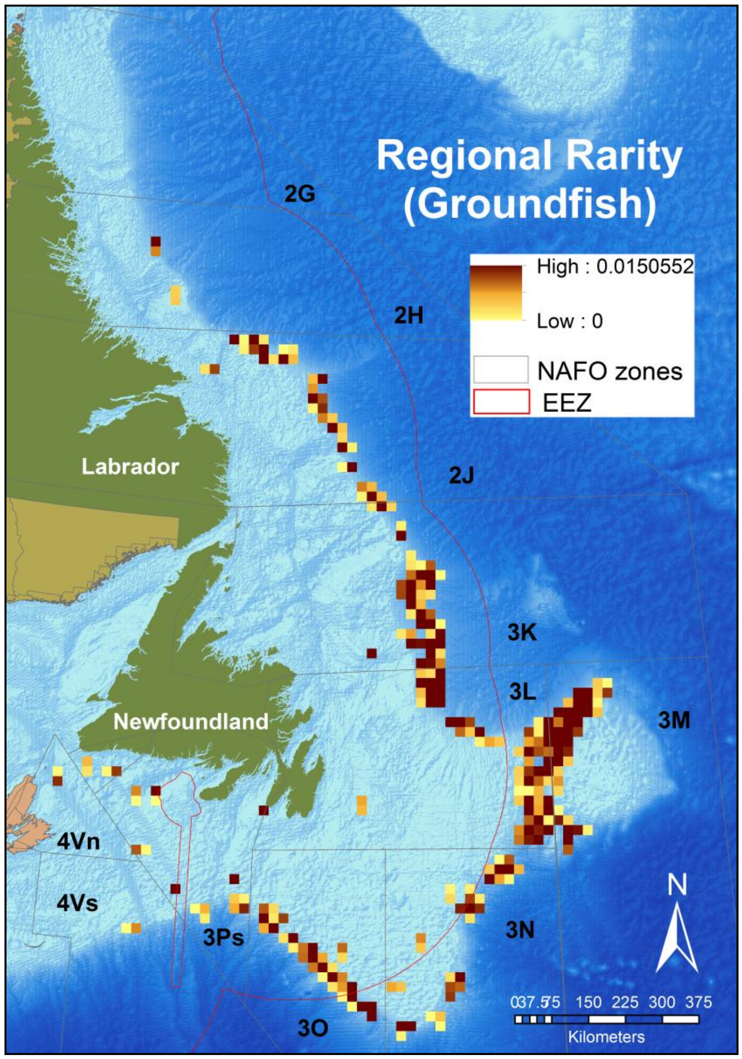


Figure 3.10 Presence of regionally rare groundfish species per square kilometer in a planning unit. Source Data: DFO MSS, 1995-2007. GEBCO bathymetry in light to dark blue indicates shallow to deep areas.

3.5.2 SEABIRDS

Seabird density also tends to follow the shelf edge, particularly along NAFO zones 2G, 2J, 3L (especially the Flemish Pass), and 3O. The inshore areas of NAFO zone 3K also show some PUs with high scores (Figure 3.11).

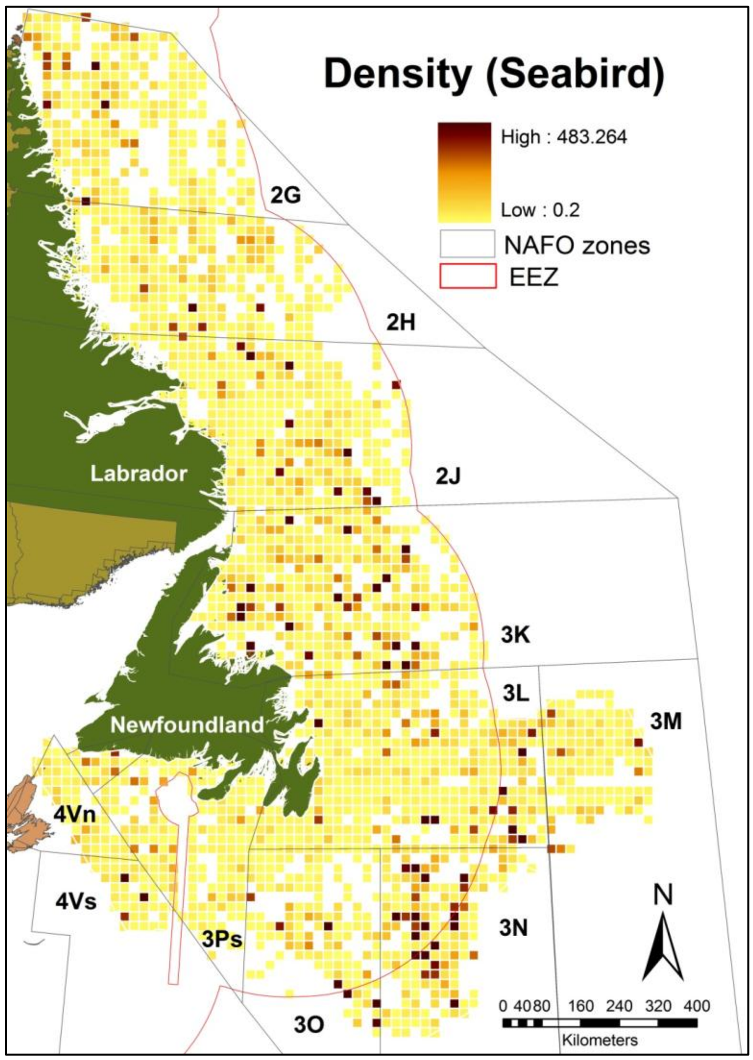


Figure 3.11 Density of seabirds expressed as average counts of individual seabirds per sample set in a planning unit. Source Data: PIROP and ECSAS, 1965-2010.

Seabird species richness appears higher in areas where there is a significant count of seabirds (density) namely: NAFO zones 3K, 2J, 3N, and 3O (Figure 3.12). Other important seabird rich areas include 4Vn, 3Pn, 2G, and 2H.

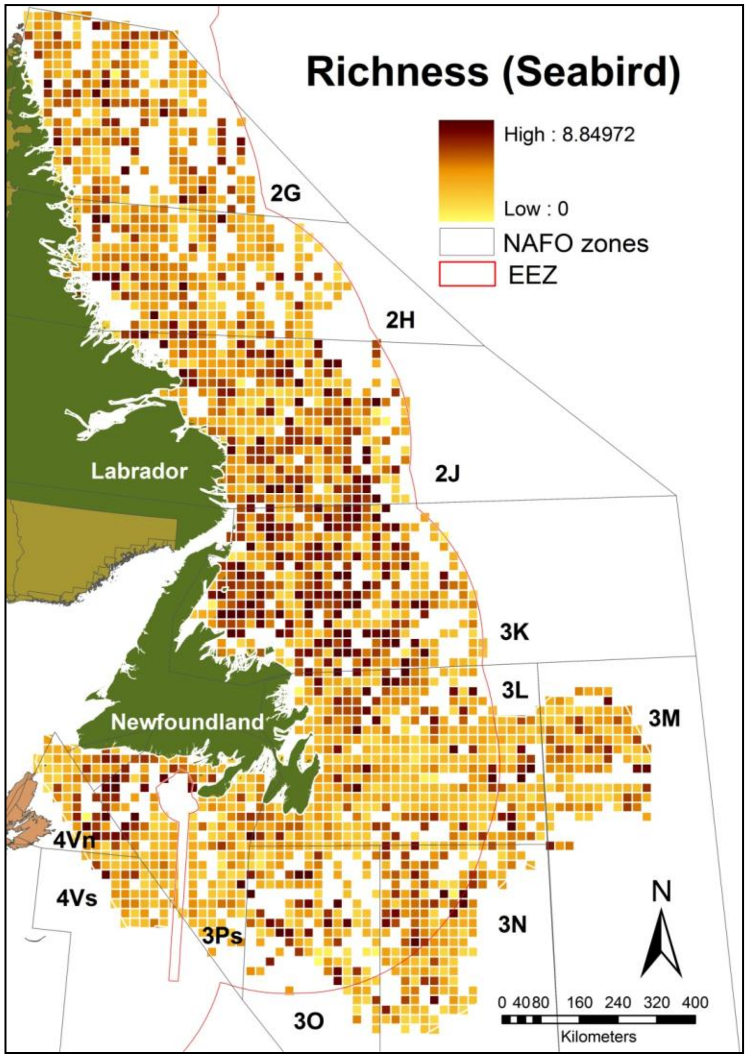


Figure 3.12 Richness of seabird species, expressed as average number of seabird, per sample set for each planning unit. Source Data: PIROP and ECSAS, 1965-2010.

Species evenness of seabirds does not show special areas of concentration. Instead, it appears that in most of the NAFO zones species evenness tends to have PUs with varying values from low to high (Figure 3.13).

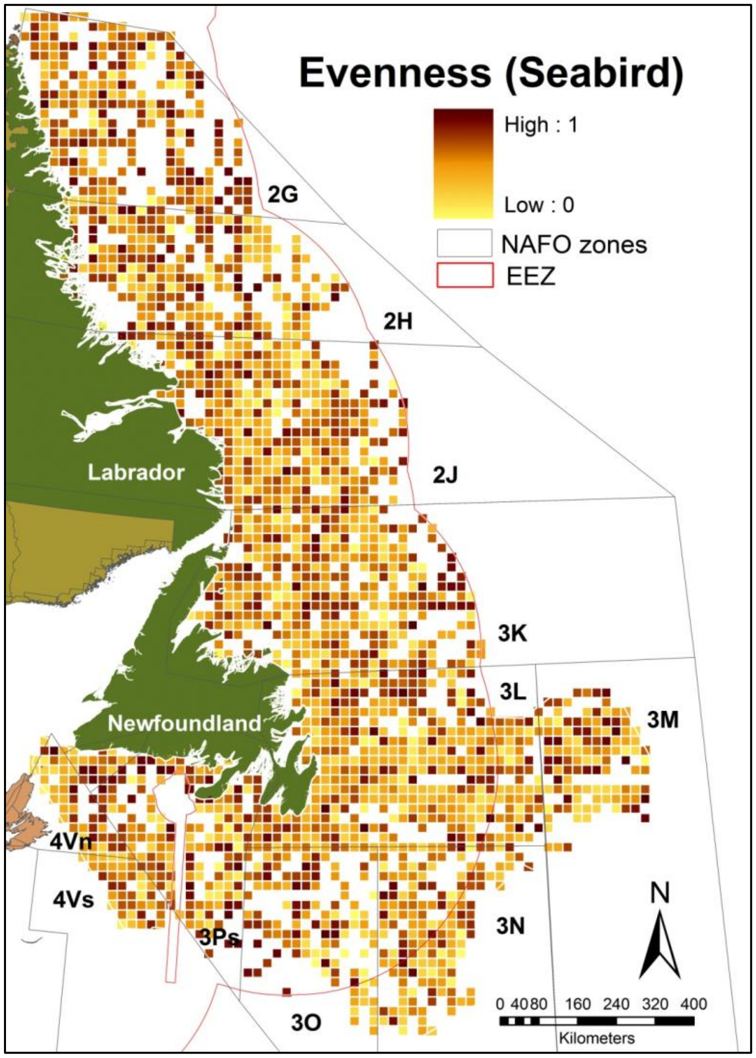


Figure 3.13 Evenness of seabirds expressed as average per sample set in a planning unit. Source Data: PIROP and ECSAS, 1965-201.

3.5.3 SPONGES

Sponge biomass tends to concentrate on the shelf edge, particularly in NAFO zones 3N, 3L, 2J, 2H, and in northern portions of NAFO 3K and 2G (Figure 3.14).

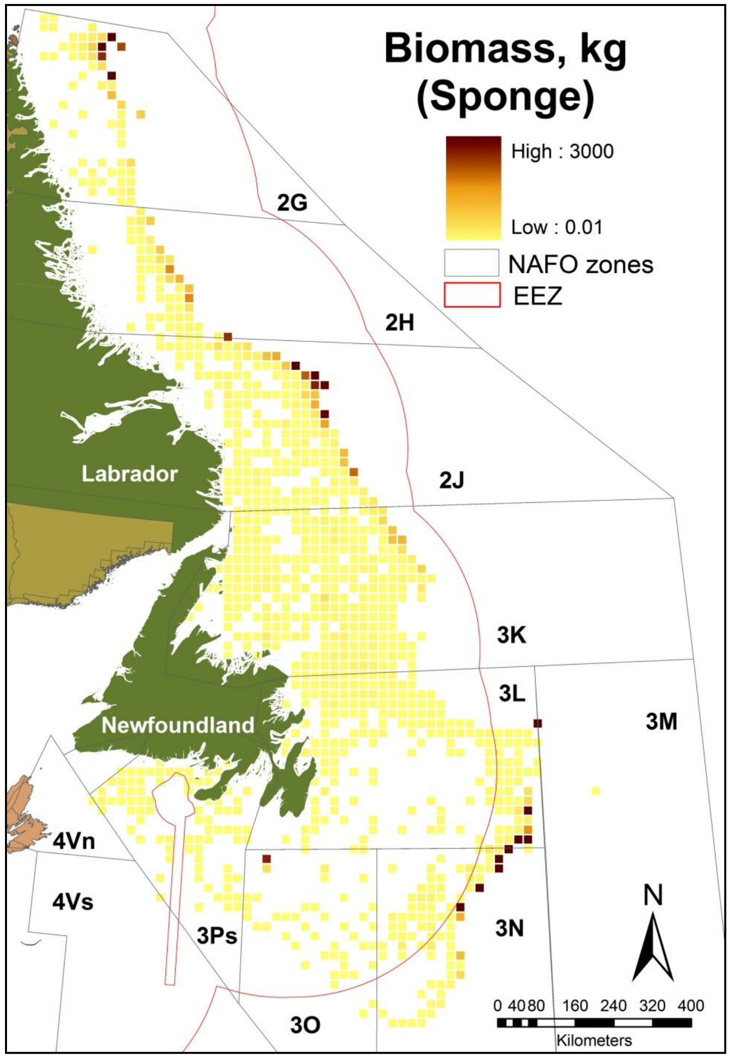


Figure 3.14 Sponge biomass expressed as average weights (kg) per sample set in a planning unit. Source Data: DFO MSS, 1995-2011.

3.5.4 CORALS

Coral biomass is particularly high in PUs along the shelf edge of NAFO zones 2G, 3Ps, and some of the PUs in 3Pn, 2J, and 2H (Figure 3.15)

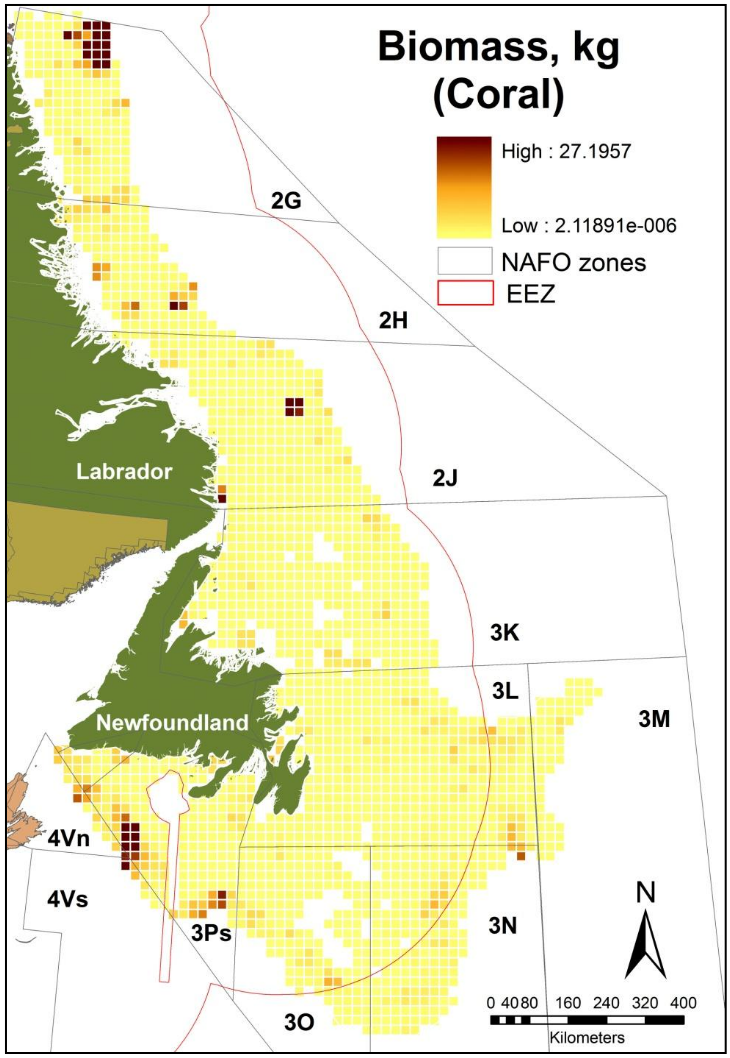


Figure 3.15 Coral biomass expressed as average weight (kg) per sample set in a planning unit. Source Data: DFO MSS by catch, 2000-2010.

Coral species-rich PUs include those in the shelf edge (except for NAFO zones 2H and 2J) together with the Flemish Pass. Some PUs in the outer shelf of NAFO zones 2J, 3Ps and 3O as well as most of NAFO zone 2G, show higher coral species richness scores (Figure 3.16).

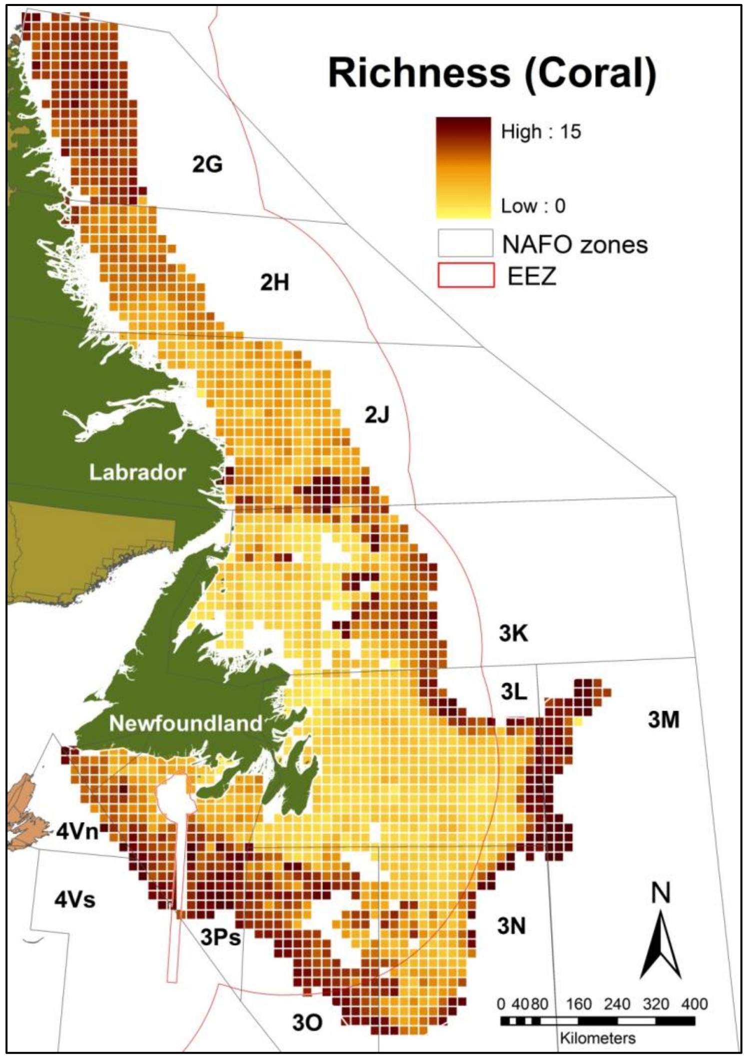


Figure 3.16 Coral species richness expressed as average number of species per sample set in a planning unit. Source Data: DFO MSS by catch, 2000-2010.

The coral sensitivity map suggests that most shelf edge areas, except for some PUs in NAFO 3N, serve as the habitats for highly sensitive coral groups. The outer shelf of NAFO zones 3O and 3Ps and the inshore areas in the southern region of Newfoundland also showed PUs with a relatively high sensitivity score (Figure 3.17).

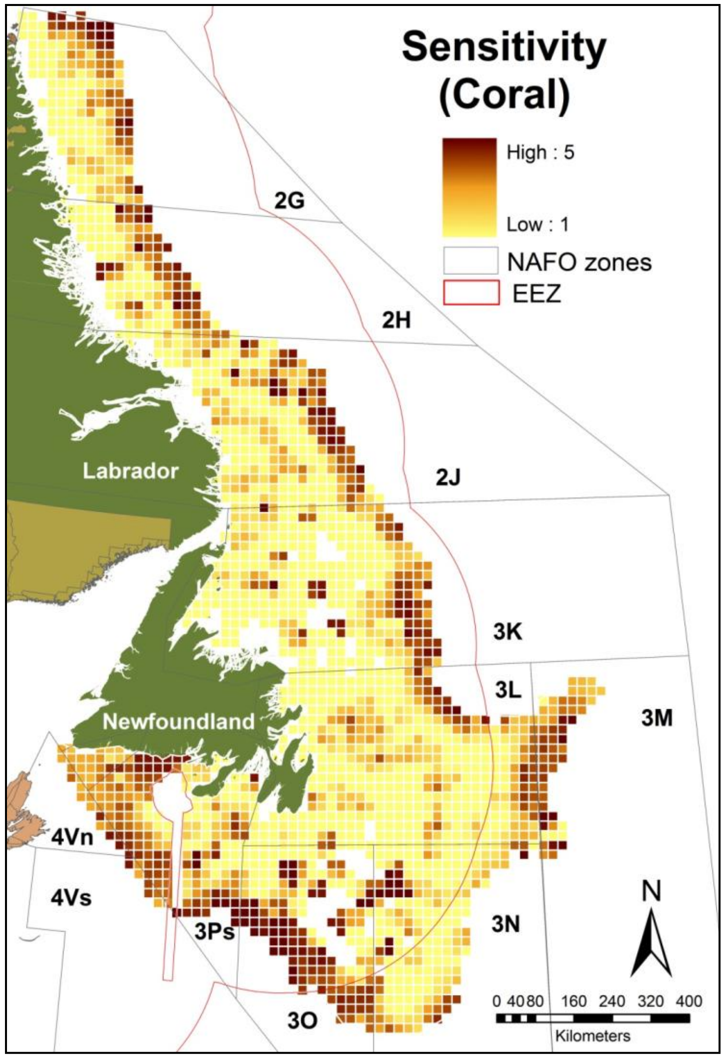


Figure 3.17 Coral sensitivity expressed as average sensitivity scores of coral groups per planning unit. Source Data: DFO MSS by catch, 2000-2010.

3.5.5 INVERTEBRATES

Shrimp biomass tends to show concentration in certain areas of the study area, including the continental shelf edge of NAFO zones 3L and 2G. Also, high biomass is observed in the nearshore towards the offshore regions of NAFO zones

3K, 2J and 2H (Figure 3.18). We should note that the southern portion of the study area shows the lowest presence of shrimp.

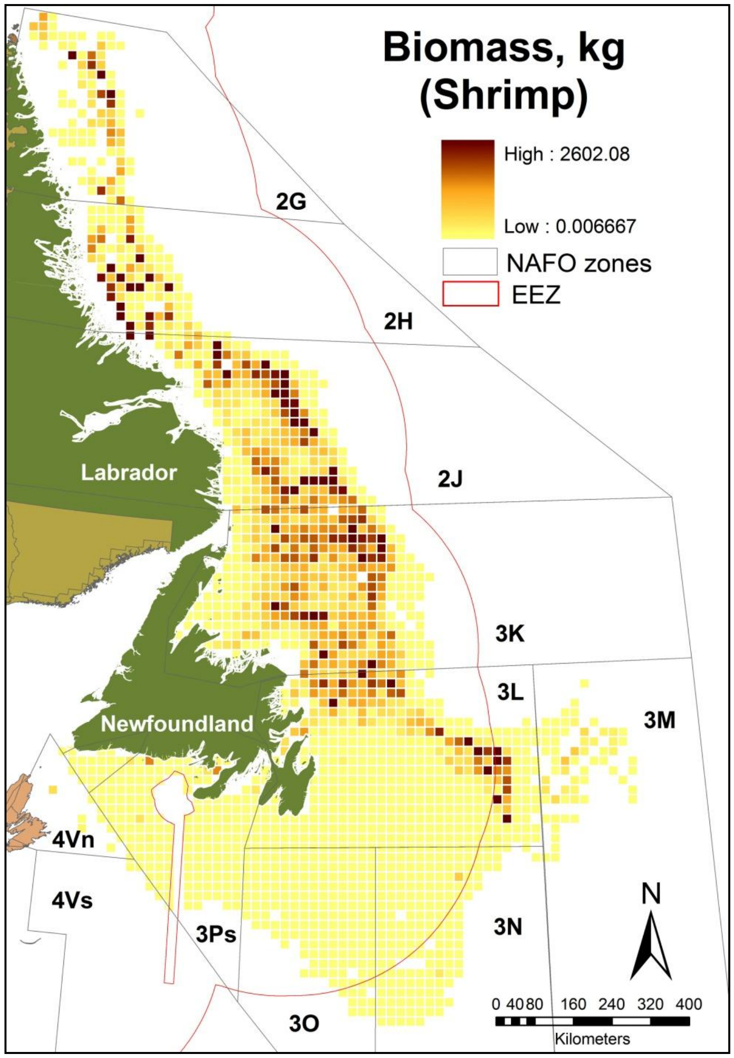


Figure 3.18 Shrimp biomass expressed as average weights of shrimp per sample set in a planning unit. Source Data: DFO MSS, 1995-2011.

Crab biomass shows higher scores in PUs of NAFO zones 2J, 3K, and 3L down to the shelf edge (NAFO zone 3N and a portion of 3O) and west outer shelf of Grand Banks. Crab biomass also covers broad inshore areas particularly in NAFO zones 3K, 3L, and 3Ps, and is almost non-existent in the northern NAFO zones of 2H and 2G (Figure 3.19).

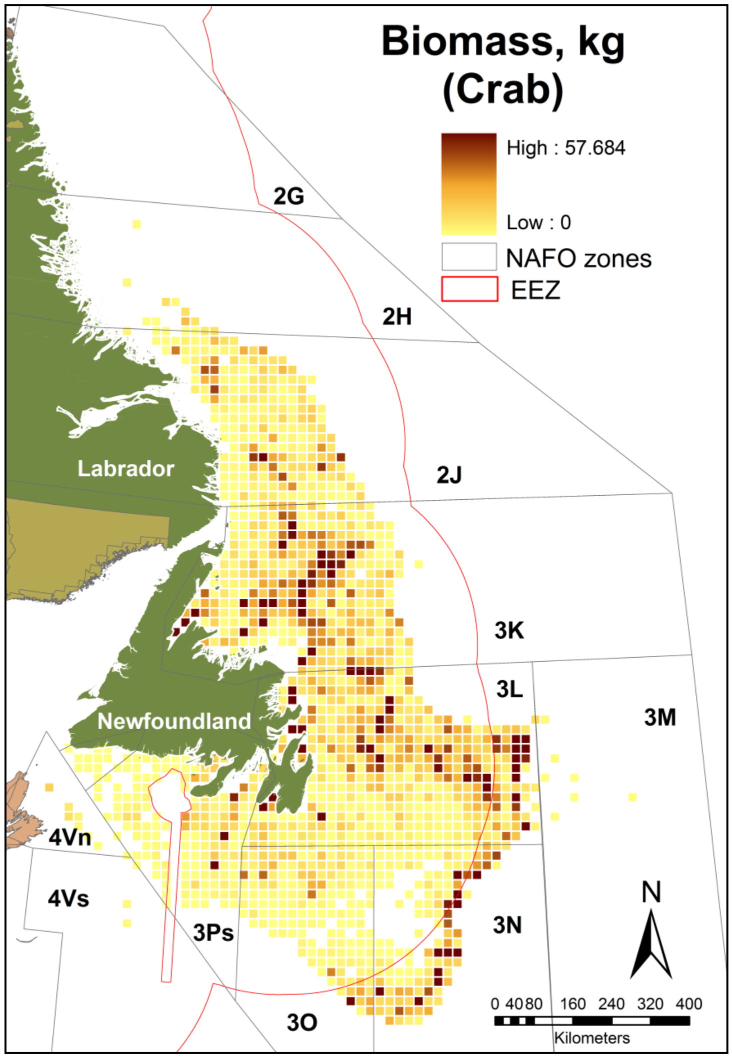


Figure 3.19 Crab biomass expressed as average weights (kg) of snow crab per sample set in a planning unit. Source Data: DFO MSS, 1995-2011.

3.5.6 LANDED CATCH

The landed catch based on biomass shows high scores for the northern NAFO zones (i.e. 2G, 2H, and 2J) capturing shrimp fishery, the major economic activities in these areas. The inshore PUs of NAFO zone 3K and 3Pn show relatively high scores mostly from the inshore crab fishery. Other small areas around the NAFO zones 4Vn, 3O, and 3N also showed high scores obtained from multiple types of fishery (e.g. cod, redfish, halibut, hake) (Figure 3.20).

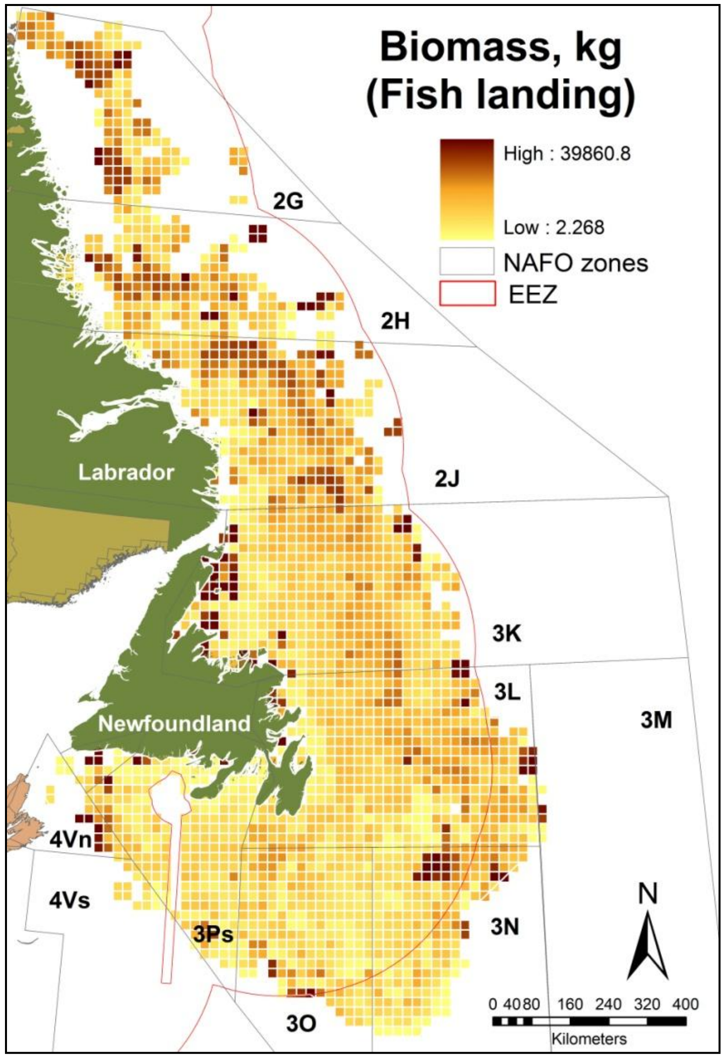


Figure 3.20 Landed catch based on biomass is expressed as average weights (kg) of landed fish per trip in a planning unit. Source data: DFO logbook, 2001-2010.

The landed value, in Canadian dollars, shows similar patterns in areas for shrimp fishing, particularly in NAFO zones 2G, 2H, and 2J, and crab fishing areas at NAFO zones 3L and the Flemish Pass. Other relatively important, but small, patches of PUs occur around the inshore region of NAFO zone 3O and 3Pn. The relative densities across PUs for fishing revenue suggest that the landed monetary value from the inshore region is relatively small compared to those from the offshore region. We should note that the logbook data excludes the lobster fishery, a dominant source of fishing revenue in the inshore areas, thus this fishery is not considered in this analysis (Figure 3.21).

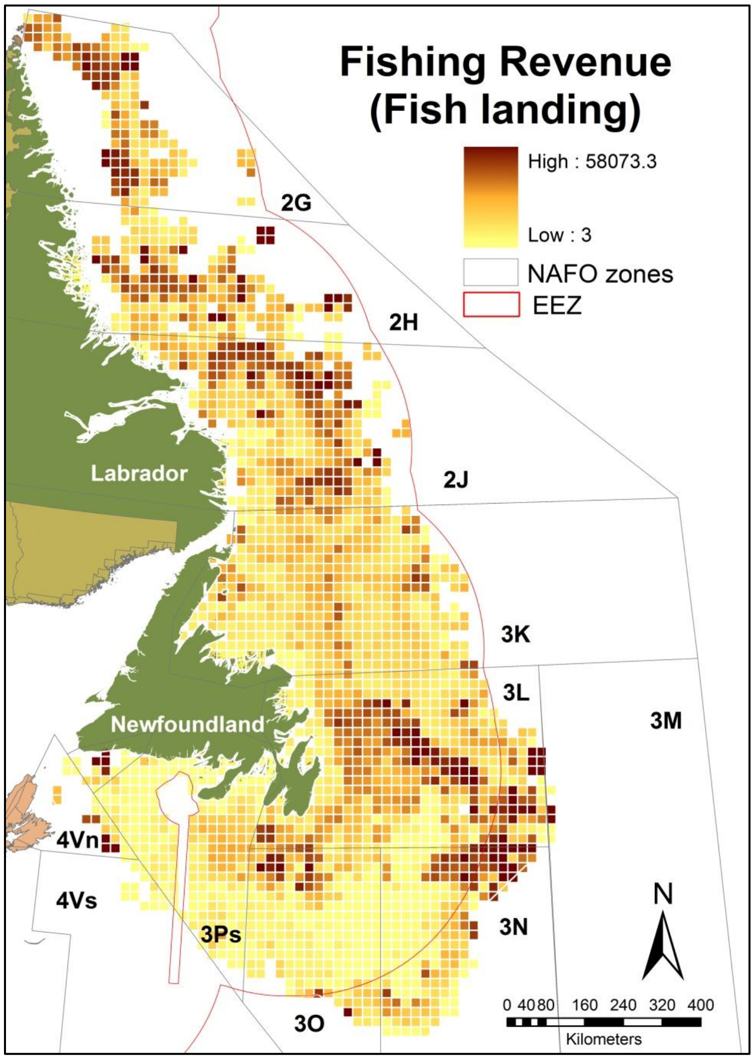


Figure 3.21 Fishing Revenue expressed as average value (C$) of landed fish per trip in a planning unit. Source Data: DFO logbook, 2001-2010.

3.5.7 FISHING BUSINESS AND FISHERS DISTRIBUTION

PUs with high importance for the fishing business mostly occur in offshore areas where shrimp and crab fishing are predominant. It is important to note, however, that the fishing business attribute was able to capture small-scale fishing activities in the inshore regions of Newfoundland (Figure 3.22).

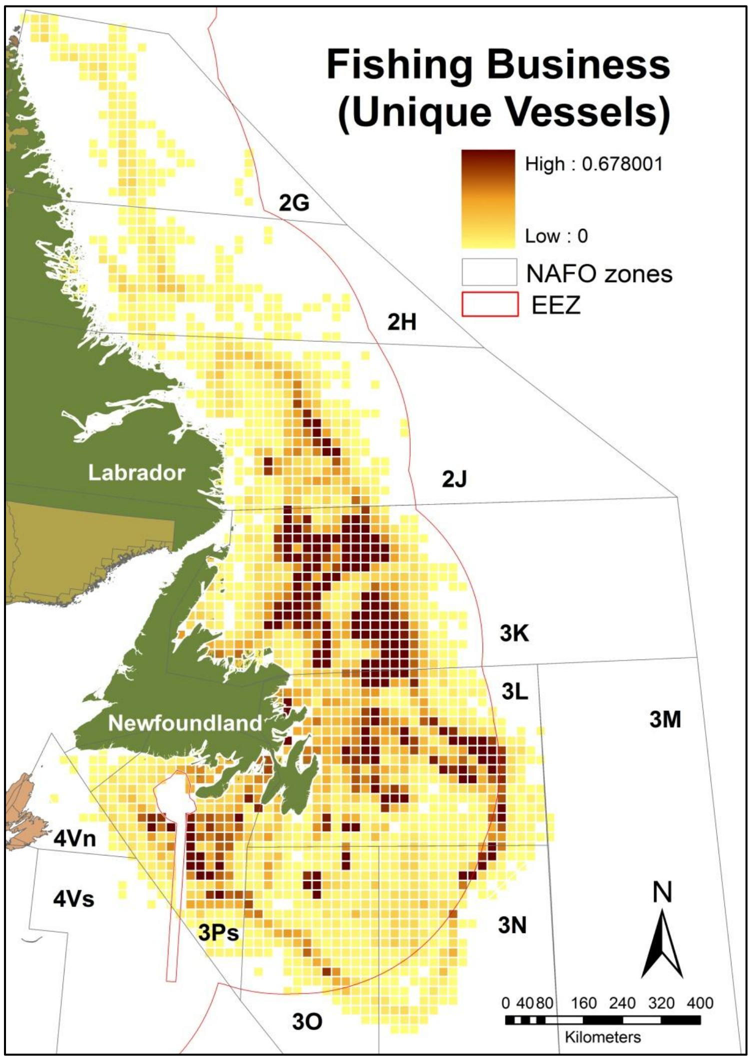


Figure 3.22 Density of fishing business, based on unique fishing vessels, per square kilometer in a planning unit. Source Data: DFO logbook 2001-2010.

Similar to the fishing business, shrimp and crab fisheries drive areas of high importance regarding fishing employment in the offshore regions. It should be noted, however, that the southern Grand Banks is relatively more important to fishing employment than fishing business (Figure 3.23).

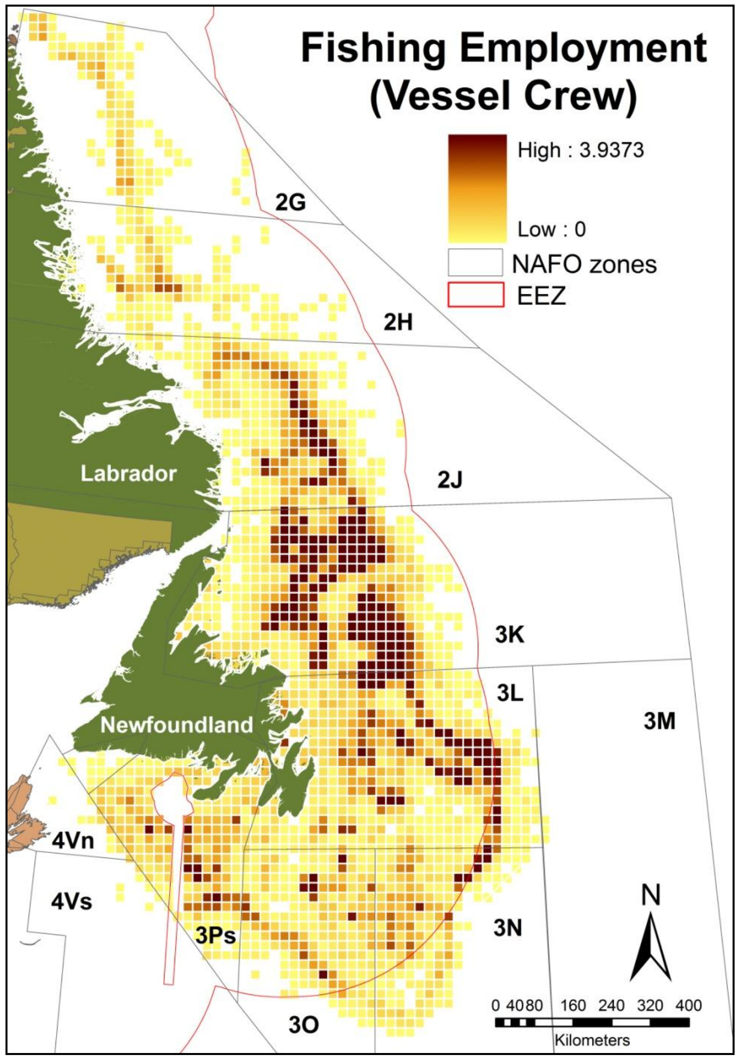


Figure 3.23 Density of fishing employment based on summed fishing vessel crew per square kilometer in a planning unit. Source Data: DFO logbook 2001-2010.

The FG-Richness suggests that the PUs off southern Newfoundland (i.e. NAFO 3Ps), both inshore and offshore regions, are areas with the most diversity of FGs. Other areas with a significant number of FGs are NAFO 2J, 3K, and 3L, with relatively high concentration toward the shelf edge. Also, some of the shelf edge of NAFO zones 2H and 3O, extending to some areas of 3N, are of medium relative importance to a variety of fisher groups (Figure 3.24).

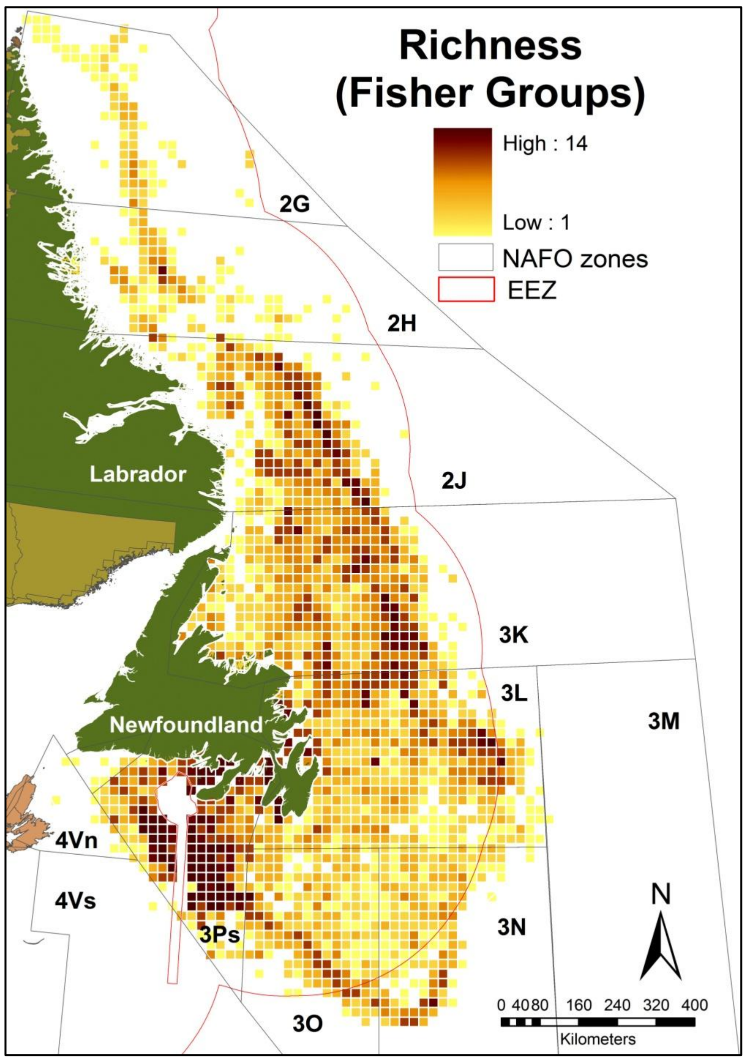


Figure 3.24 FG-Richness expressed as average number of fisher groups per planning unit. Source Data: DFO logbook 2001-2010.

The evenness of fisher groups indicates that NAFO zones 3Ps, 3O, and 2J as well as those in the shelf edge of NAFO zone 2G have a high evenness score – that is, a high proportion of fishers among FGs may have an equal share of these areas (Figure 3.25).

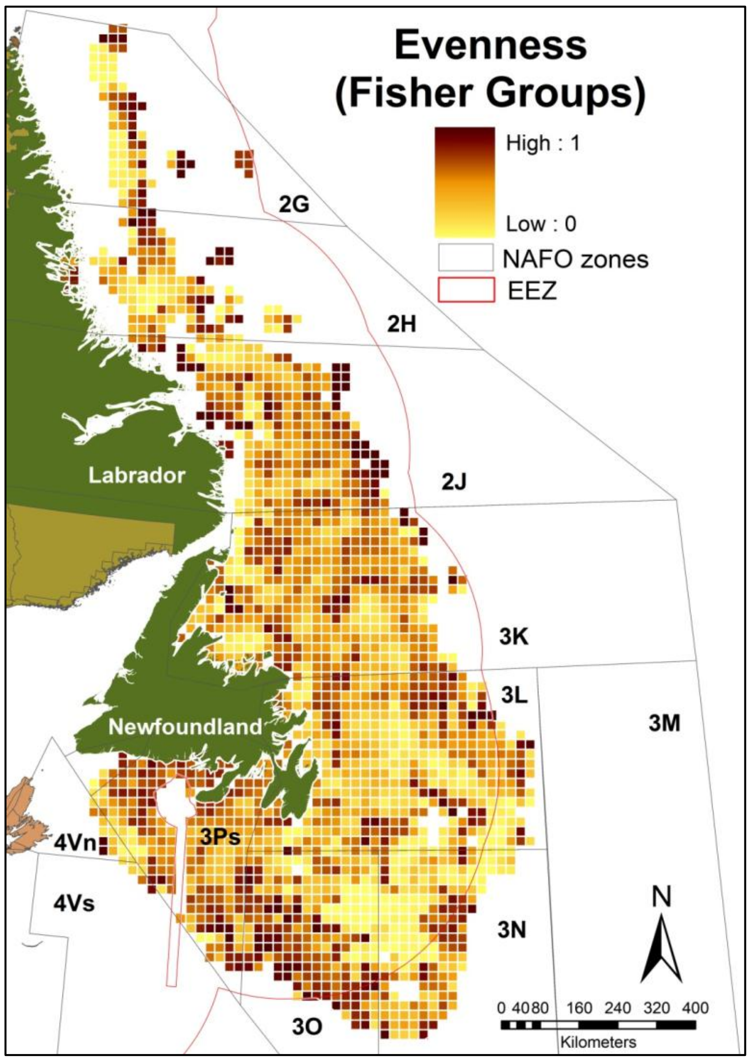


Figure 3.25 FG-Evenness of fisher groups in a planning unit. Source Data: DFO logbook 2001-2010.

3.5.8 OTHER HUMAN USES

Areas with potentially high adverse ecological impacts from the use of fishing gears are mostly around NAFO 3K and 2J, which corroborates with shrimp fishing zones. Other areas that may also have suffered from adverse ecological impacts of gears are observed along the shelf edge of NAFO zones 3Ps and 3O, where trawling of other types of directed species is also high (Figure 3.26).

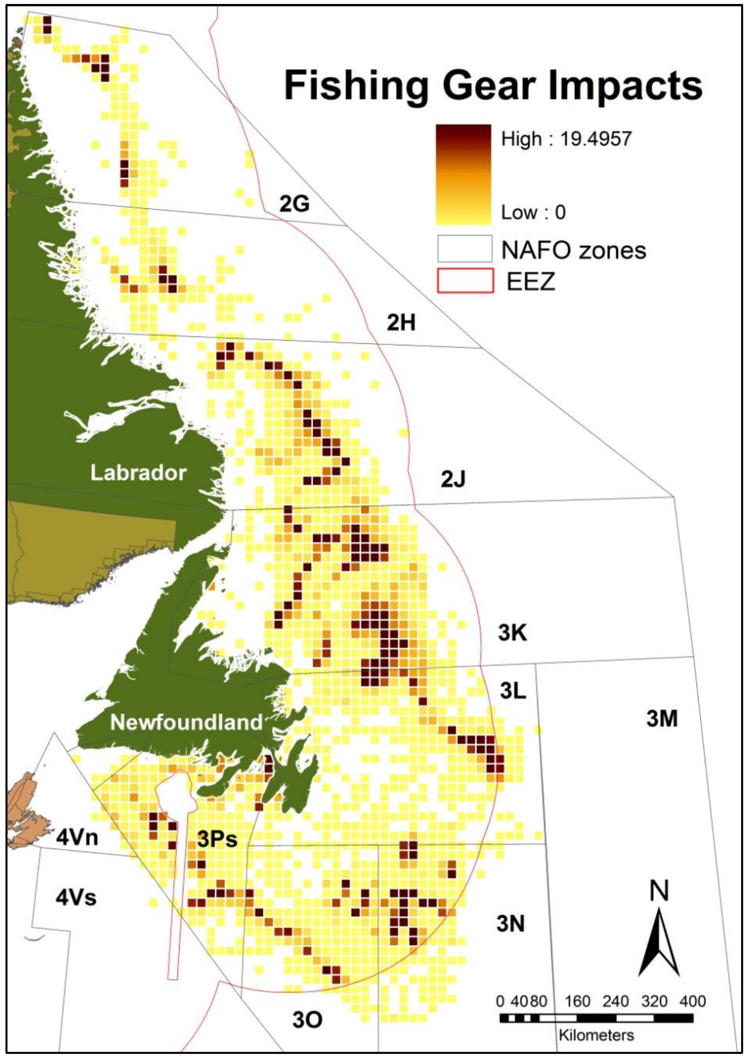


Figure 3.26 Fishing gear impacts per square kilometer, based on the sum of severity scores assigned to fishing gears, in a planning unit. Source Data: DFO logbook, 2001-2010.

Areas of higher marine commercial traffic concentrate off of southern Newfoundland, including NAFO 3Pn, 3Ps, 3O, 3N, and 3M (Figure 3.27). This pattern largely reflects maritime transportation between Europe and cities located along the St. Lawrence River. We should note that this map resembles the oil spill risk areas identified in the south coast of Newfoundland following a separate assessment report (Transport Canada 2007). Also, along the beaches of this area, a higher linear density of oiled birds was reported relative to other areas in the world (Weise & Ryan 2003).

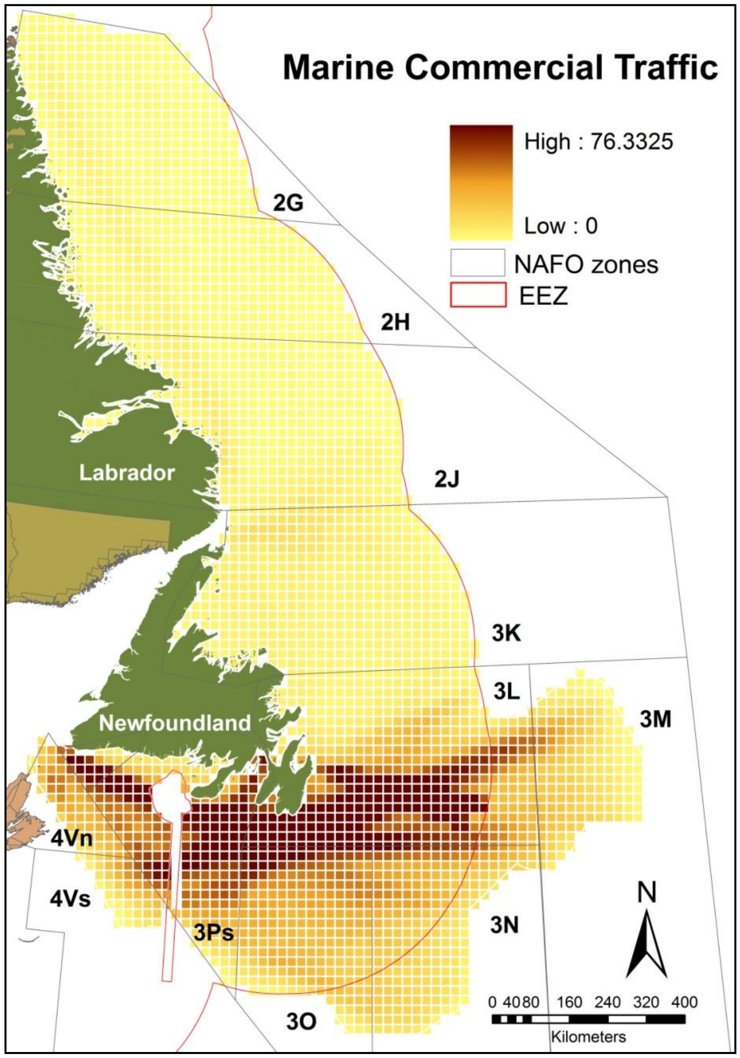


Figure 3.27 Marine commercial traffic is expressed as density of vessel tracks in kilometers per square kilometer in a planning unit. Source Data: Long Range Identification and Tracking (LRIT), Feb 2010 - Feb 2011.

Oil and gas activities are concentrated in a small portion of the study area located mainly around the northeastern Grand Banks, where large oil rigs are located and where significant oil production is ongoing. Some patches of activities are also present in NAFO zones 2H, 2J, 3O, and 3N, but these areas are mostly locations for exploratory and significant discovery licenses (Figure 3.28).

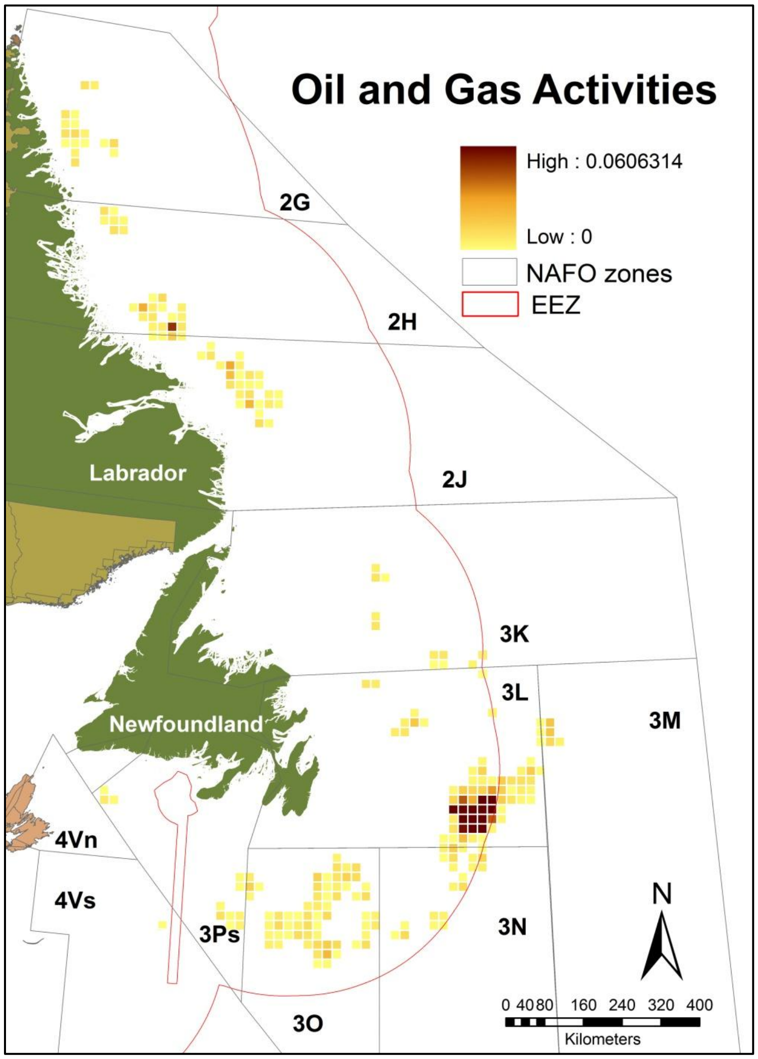


Figure 3.28 Density of oil and gas activities per square kilometer, based on weights assigned to each license category (Table 3.12), for each planning unit. Source Data: C-NLOPB, 1986-2010.

As for areas of potential gear conflict, a relatively high spatial overlap between shrimp and crab fishing are observed around NAFO 3K and 2J (Figure 3.29). Overlap scores [0-1] implies zero to complete spatial overlap.

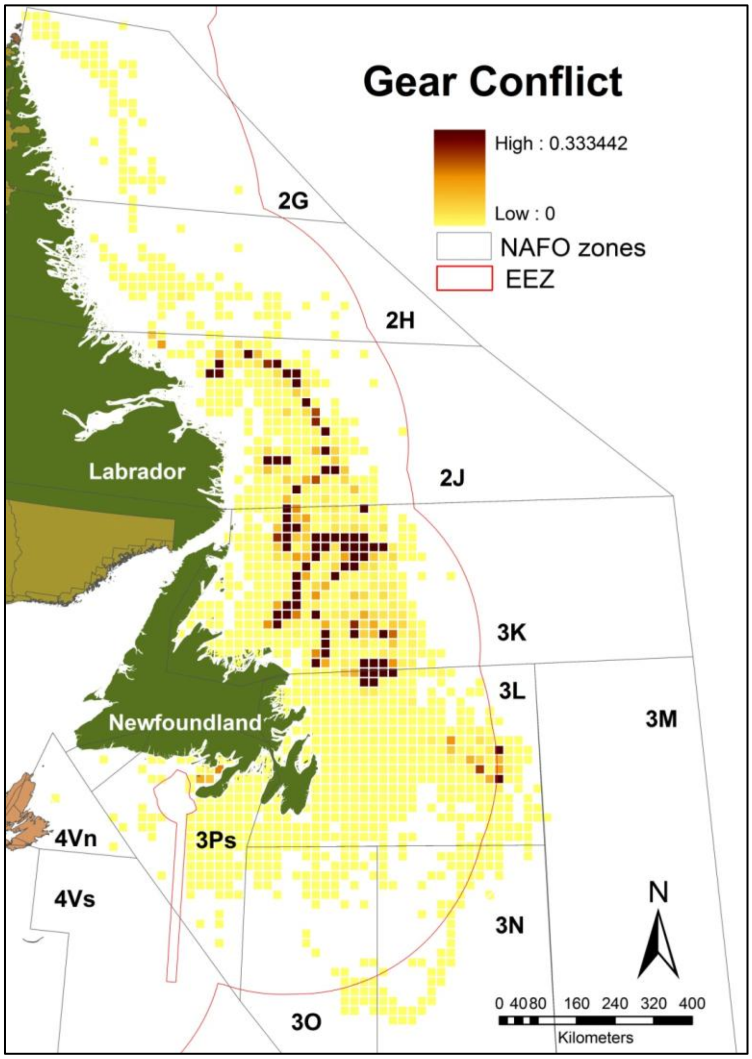


Figure 3.29 Gear conflict refers to the overlap of effort between shrimp and crab fishing. Source Data: DFO logbook, 2001-2010.

The 15 raster-based biodiversity attributes identified certain portions of the study area that are relatively more important than others. The shelf edge shows that it is rich in many uncommon and rare groundfish species, as well as the sensitive and habitat-forming species such as corals and sponges. One potential reason could be its unique environment, including its slope, depth, and current, providing this region with distinctive physical processes (Smith & Sandstrom 1988), and possibly a rare habitat for its diverse species that may have less flexibility in living outside this area. In addition to the biological diversity observed at the shelf-edge, species at risk are also found in PUs in the Flemish Pass.

The central portion of the study area (i.e. NAFO zones 2J and 3K) showed relatively higher importance (compared to the northern NAFO zones 2G and 2H) to groundfish species that are at risk, rare, and with higher biomass, to different species of seabirds, and to commercially exploited invertebrates including crab and shrimp. The southern portions of the study area (i.e. NAFO zones 3L, 3M, 3N, 3O, and 3Ps) have several distinctive patches relevant to biodiversity. These include the southern portion of the Grand Banks showing high groundfish biomass, as well as high seabird density and seabird species richness. NAFO zone 3Ps has high groundfish evenness, with PUs important to rare species, seabird, coral, and crab. NAFO zone 3O is distinctively rich in coral species and supports those in more sensitive groups.

The ten socioeconomic raster-based attributes also showed that certain portions of offshore areas are more important than others. The northern NAFO zones including 2G, 2H, and 2J are important sources of fishing revenue as well as high biomass of landed fish, but less significant for marine commercial traffic and

O&G activities. The central portion of the study area (i.e. NAFO zones 2J, and 3K) are of particular importance to the fishing business and multiple types of fisher groups, but is less important for fishing employment. These areas are also highly trawled; hence, they may have the most adverse impacts from fishing gears. The southern portion of the study area showed higher importance to multiple types of socioeconomic activities including fishing, marine traffic, and O&G.

In terms of spatial overlap between biodiversity and socioeconomic attributes, certain areas are notable. These include the shelf edge of the study area, identified as species-rich but also showing high economic importance to commercially exploited crab and shrimp fisheries. NAFO zone 3K (both inshore and outer shelf) shows high importance biologically, but is also critical for fishing businesses, fishing employment, and landed biomass. The southern portion of Newfoundland waters (i.e. NAFO zones 3Ps and 3O) is rich in seabirds and coral species, but is also relevant to different types of fisher groups as well as commercial transport.

This study also provides evidence that modern means of fishing may have a highly adverse impact on areas with high biodiversity. This overlap is particularly notable along the shelf edge and the outer shelf of the NAFO zone 3K. In addition, the study confirms that two commercial fisheries, shrimp and crab, drive most of the fishing revenue, fishing business, and fishing employment in this province.

3.6 DISCUSSION AND SUMMARY

When generating the raster-based attributes, the kernel density estimations were primarily used to represent the relative value of individual measures or attributes (e.g. species richness, fishing employment, and gear impact) for each PU. Describing biodiversity and human activities through spatial attributes has associated limitations, as do the interpretations generated from them. We should note that the analysis and results of this study are highly dependent on survey effort, datasets made available for the project, attributes' scale of analysis, and the density calculation using the GIS-based raster analysis. Thus, application of the results to actual planning efforts should be done with some degree of caution and appropriate understanding of the data and possible limitations in terms of interpretations.

While existing literature and consultation of expert opinions played roles in deciding which information to generate for this study, the availability of particular datasets was the primary factor in identifying the spatial attributes. The 25 GIS raster-based attributes used in the study cannot provide an exhaustive view of the biodiversity and socioeconomic activities in the study area. However, these attributes represent one of the largest data collection exercises ever attempted in the study area. It is also worth noting that this study assumed that the temporal differences between datasets do not make a significant difference as to the spatial overlap of biodiversity and socioeconomic activities.

The mapping of the attributes allowed visualizing different patterns of biodiversity and socioeconomic activities. Similar to biodiversity, socioeconomic activities also have different facets that are difficult to quantify using a single attribute. In fact, this study considered socioeconomic costs only from three types of datasets excluding other recently growing socioeconomic activities, particularly in inshore areas, such as aquaculture, tourism, and recreational fishing. Developing appropriate methods for multiple types of socioeconomic attributes that can occur at various scales and being able to integrate all of them in planning analysis is a future challenge in data-driven planning. This is important to note, as a rich representation of socioeconomic activities remains a gap in MPA planning methods.

Overall, this chapter provides a bird's eye view of a far more challenging aspect of conservation decision-making – that is, when socioeconomic competing issues are considered in a similar way as biodiversity issues. The raster-based attributes presented in this study show that there are potentially significant spatial overlaps between species-rich areas and areas with high socioeconomic human activities. This spatial overlap is a challenge to deal with for most planners, decision makers, and policy makers. Chapter 4 presents a method that offers an alternative approach to help planners and stakeholders discuss and make decisions using the attributes generated in this chapter.

3.7 LITERATURE CITED

Ando, A., J. Camm, S. Polasky, and A. Solow. 1998. Species distributions, land values, and efficient conservation. Science **279**:2126-2128.

Angermeier, P. 2000. The natural imperative for biological conservation. Conservation Biology **14**:373-381.

Ban, N., and C. Klein. 2009. Spatial socioeconomic data as a cost in systematic marine conservation planning. Conservation Letters **2**:206-215.

Boroushaki, S., and J. Malczewski. 2008. Implementing an Extension of the Analytical Hierarchy Process Using Ordered Weighted Averaging Operators with Fuzzy Quantifiers in ArcGIS. Computers and Geosciences **34**:399-410.

Brodie, W. 2005. A Description of the Autumn Multispecies Surveys in SA2+ Divisions 3KLMNO from 1995-2004. Northwest Atlantic Fisheries Organization. NAFO SCR Document 05/8. Northwest Atlantic Fisheries Organization.

Boroushaki, S., and J. Malczewski. 2008. Implementing an extension of the analytical hierarchy process using ordered weighted averaging operators with fuzzy quantifiers in ArcGIS. Computers and Geosciences **34**:399-410.

Brodie, W. 2005. A description of the autumn multispecies surveys in SA2+ divisions 3KLMNO from 1995-2004. Northwest Atlantic Fisheries Organization. NAFO SCR Document 05/8. Northwest Atlantic Fisheries Organization.

Brodie, W., and D. Stansbury. 2007. A brief description of Canadian multispecies surveys in SA2+ divisions 3KLMNO from 1995-2006. NAFO SCR Document 07/18. Northwest Atlantic Fisheries Organization.

Callicot, J., L. Crowder, and K. Mumford. 1998. Current normative concepts in conservation. Conservation Biology **13**:22-35.

COSEWIC 2000. COSEWIC assessment and status report on the Atlantic wolffish *Anarhichas lupus* in Canada. Committee on the Status of Endangered Wildlife in Canada, Ottawa, 21 pp.

COSEWIC 2001a. COSEWIC assessment and status report on the northern wolffish *Anarhichas denticulatus* in Canada. Committee on the Status of Endangered Wildlife in Canada, Ottawa, 21 pp.

COSEWIC 2001b. COSEWIC assessment and status report on the spotted wolffish *Anarhichas minor* in Canada. Committee on the Status of Endangered Wildlife in Canada, Ottawa, 22 pp.

COSEWIC 2003. COSEWIC assessment and update status report on the Atlantic cod *Gadus morhua* in Canada. Committee on the Status of Endangered Wildlife in Canada, Ottawa, 76 pp.

COSEWIC 2003. COSEWIC assessment and status report on the cusk *Brosme Brosme* in Canada. Committee on the Status of Endangered Wildlife in Canada, Ottawa, 30 pp.

COSEWIC 2005. COSEWIC assessment and status report on the winter skate Leucoraja ocellata in Canada. Committee on the Status of Endangered Wildlife in Canada, Ottawa, 41 pp.

COSEWIC 2007. COSEWIC assessment and status report on the roughead grenadier *Macrourus berglax* in Canada. Committee on the Status of Endangered Wildlife in Canada, Ottawa, 40 pp.

COSEWIC. 2008. COSEWIC assessment and status report on the Roundnose Grenadier *Coryphaenoides rupestris* in Canada. Committee on the Status of Endangered Wildlife in Canada, Ottawa, 42 pp.

COSEWIC. 2009. COSEWIC assessment and status report on the American Plaice *Hippoglossoides platessoides*, Maritime population, Newfoundland and Labrador population and Arctic population, in Canada. Committee on the Status of Endangered Wildlife in Canada, Ottawa, 74 pp.

COSEWIC. 2010. COSEWIC assessment and status report on the Spiny Dogfish *Squalus acanthias*, Atlantic population, in Canada. Committee on the Status of Endangered Wildlife in Canada, Ottawa, 50 pp.

Crowder L., et al. 2006. Resolving mismatches in U.S. ocean governance. Science **313**:617-618.

De Smith, M., M. Goodchild, and P. Longley. 2007. Geospatial analysis: A comprehensive guide to principles, techniques and software tools. 2nd edition. British Library Cataloguing in Publishing Data, UK.

Devine, J., K. Baker, and R. Haedrich. 2006. Fisheries: Deep-sea fishes qualify as endangered. Nature **439**:29-29.

DFO (Fisheries and Oceans Canada). 2002. Proceedings of the workshop on greenland halibut. Canadian Science Advisory Secretariat, Proceeding series 2001/036.

DFO (Fisheries and Oceans Canada). 2004. Costs and earnings survey. Atlantic Region report, Economic Analysis and Statistics. DFO Policy Sector, Ottawa, Canada.

DFO (Fisheries and Oceans Canada). 2005. Socioeconomic consideration to inform a decision whether or not to list three populations of Atlantic cod under SARA. Technical document. Policy Sector, DFO, 143 pp.

DFO (Fisheries and Oceans Canada). 2009. Development of a framework and principles for the biogeographic classification of Canadian marine areas. DFO Canadian Scientific Advisory Secretariat 2009/056.

Di Minin, E. and A. Moilanen. 2012. Empirical evidence for reduced protection levels across biodiversity features from target-based conservation planning. Biological Conservation **153**:187-191.

Douvere, F. 2008. The importance of marine spatial planning in advancing ecosystem-based sea use management. Marine Policy **32**:762– 771.

Edinger, E., K. Baker, R. Devillers and V. Wareham. 2007. Coldwater corals off Newfoundland and Labrador: Distribution and fisheries impacts. WWF-Canada, Toronto.

Ehrlich, P. and E. Wilson. 1991. Biodiversity studies: Science and policy. Science **253**:758-762.

Ehrlich, P. 1994. Energy use and biodiversity loss. Philosophical Transactions of the Royal Society of London. Series B: Biological Sciences **344**:99-104.

Fuller, S., C. Picco, J. Ford, C. Tsao, M. Lance, D. Hangaard, and R. Chuenpagdee. 2008. How we fish matters. Ecology Action Centre, Living Oceans Society, and Marine Conservation Biology Institute, British Columbia, Canada.

Gjerdrum C., D. Fifield and S. Wilhelm. 2012. Eastern Canada seabirds at sea (ECSAS) standardized protocol for pelagic seabird surveys from moving and stationary platforms. Canadian Wildlife Service, Technical Report Series Number 515. CWS, Atlantic Region.

Gomes, M., R. Haedrich, and M. Villagarcia. 1995. Spatial and temporal changes in the groundfish assemblages on the north-east Newfoundland/Labrador Shelf, north-west Atlantic, 1978–1991. Fisheries Oceanography **4**:85-101.

Goulet, P., M. Walsh, A. Cuff, R. Devillers, and E. Edinger. 2010. Data analysis towards GIS-based ecoregion delineation in the NL shelf, 1995-2007. Final reports to Fisheries and Oceans Canada, St. John's, NL. St. John's, NL, Canada.

Holmlund, C. and M. Hammer. 1999. Ecosystem services generated by fish populations. Ecological economics **29**:253-268.

Hooper, D., et al. 2005. Effects of biodiversity on ecosystem functioning: a consensus of current knowledge. Ecological Monographs **75**:3-35.

Hutchings, J. and R. Myers. 1994**.** What can be learned from the collapse of a renewable resource? Atlantic cod, *Gadus morhua*, of Newfoundland and Labrador. Canadian Journal of Fisheries and Aquatic Resources **51**:2126-2146.

Jackson, J., et al. 2001. Historical overfishing and the recent collapse of coastal ecosystems. Science **293:**629–638.

Kaiser, M. 2014. The conflict between static and mobile gear in inshore fisheries. Policy Department B: Structural and Cohesion Policies. Available from http://www.europarl.europa.eu/studies (accessed May 2015).

Kangas, M., A. Chandrapavan, Y. Hetzel, E. Sporer. 2012. Minimising gear conflict and resource sharing issues in the Shark Bay trawl fisheries and promotion of scallop recruitment. Fisheries Research Report No. 229. Fisheries Research Division, Western Australian Fisheries and Marine Research Laboratories

Kukkala, A. and A. Moilanen. 2013. Core concepts of spatial prioritization in systematic conservation planning. Biological Reviews **88**:443-464.

Leader-Williams, N., W. Adams, and R. Smith. 2010. Trade-offs in conservation: Deciding what to save. Wiley-Blackwell, Oxford, UK.

Loreau, M., et al. 2001. Biodiversity and Ecosystem Functioning: Current Knowledge and Future Challenges. Science **294**:804-808.

Mahon, R., S. Brown, K. Zwanenburg, D. Atkinson, K. Buja, L. Claffin, G. Howell, M. Monaco, R. O'Boyle, and M. Sinclair. 1988. Assemblages and biogeography of demersal fishes of the east coast of North America. Canadian Journal of Fisheries and Aquatic Sciences **55**:1704-1738.

Malczewski, J. 1999. GIS and multicriteria decision analysis. John and Wiley Sons, Inc., USA.

McCallum, B., and S. Walsh. 1996. Groundfish survey trawls used at the northwest Atlantic Fisheries Center, 1971 to present. NAFO Scientific Council Studies **29**:93-104.

Margules, C. and R. Pressey. 2000. Systematic conservation planning. Nature **405**:203-253.

Margules, C. and S. Sarkar. 2007. Systematic Conservation Planning. University Press, Cambridge.

Maritime Safety Committee. 2008. Revised performance standards and functional requirements for the long-range identification and tracking of ships, Resolution MSC.263 (84). Available from http://www.imo.org/ourwork/safety/navigation/documents/lrit/msc.263(84).pdf (accessed December 2014).

Marris, E. 2007. Conservation priorities: What to let go. Nature **450**:152-155.

McCallum, B. and S. Walsh. 1997. Groundfish survey trawls used at the Northwest Atlantic Fisheries Center, 1971-present. NAFO Scientific Council Studies **29**:93-104.

McCallum, B. and S. Walsh. 2002. An update on the performance of the Campelen 1800 during the bottom trawl survey in NAFO subareas 2 and 3 in 2001. NAFO SCR Document 02/32. Northwest Atlantic Fisheries Organization.

Mitchell, A. 1999. The ESRI guide to GIS Analysis: Volume 1: Geographic Patterns and relationships. ESRI Press, California.

Moilanen, A. 2008. Generalized complementarity and mapping of the concepts of systematic conservation planning. Conservation Biology **22**:1655-1658.

Moilanen, A., and A. Arponen. 2011. Setting conservation targets under budgetary constraints. Biological Conservation **144**:650-653.

Myers N., R. Mittermeier, C. Mittermeier, G. da Fonseca, and I. Kent. 2000. Biodiversity hotspots for conservation priorities. Nature **403**:853-858.

Pepin, P., A. Cuff, M. Koen-Alonso, and N. Ollerhead. 2010. Preliminary analysis for the delineation of marine ecoregions on the NL shelves. NAFO SCR Document 10/72. Northwest Atlantic Fisheries Organization.

Pereira, H., et al.2010. Scenarios for global biodiversity in the 21st century. Science **330**: 1496-1501.

Perez-Rodriguez, A., A. Cuff, N. Ollerhead, P. Pepin, and M. Koen-Alonso. 2010. Preliminary analysis towards the delineation of marine ecoregions in the Flemish Cap, Northwest Atlantic. NAFO SCR Document 10/ 073. Northwest Atlantic Fisheries Organization.

Pressey, R., and A. Nicholls. 1989. Efficiency in conservation: Scoring versus Iterative approaches, Biological Conservation **50:**199-218.

Preston, W. 1948. The Commonness, and rarity, of species. Ecology **29**:254-283.

Purvis, A. and A. Hector. 2000. Getting the measure of biodiversity. Nature **405**:212-219.

Rabinowitz, D., S. Cairns, and T. Dillon. 1986. Seven forms of rarity and their frequency in the flora of the British Isles.

Roberts, C., and J. Hawkins. 1999. Extinction risk in the sea. Trends in Ecology & Evolution **14**:241-246.

Roberts, C., et al. 2002. Marine biodiversity hotspots and conservation priorities for tropical reefs. Science **295**:1280-1284.

Rose, G. 2003. Fisheries resources and science in Newfoundland and Labrador: An independent assessment. Royal Commission on Renewing and Strengthening Our Place in Canada. Available from http://www.gov.nl.ca/publicat/royalcomm/research/Rose.pdf (accessed May 2015).

Sawyer, D. and S. Stiebert. 2010. Fossil fuels-at what cost? Government support for upstream oil activities in three Canadian provinces: Alberta, Saskatchewan and Newfoundland & Labrador. Government support for upstream oil activities in three Canadian provinces: Alberta, Saskatchewan and Newfoundland & Labrador. Global Subsidies Initiative (GSI) of the International Institute for Sustainable Development (IISD), Geneva, Switzerland

Sherwood, O. and E. Edinger. 2009. Ages and growth rates of some deep-sea gorgonian and antipatharian corals of Newfoundland and Labrador. Canadian Journal of Fisheries and Aquatic Sciences **66**:142-152.

Silverman, B. 1986. Density estimation for statistics and data analysis. Chapman and Hall, New York.

Singleton, S. 2009. Native people and planning for marine protected areas: How "stakeholder" processes fail to address conflicts in complex, real-world environments. Coastal Management **37**:421-440.

Smith, F., et al. 1993. How much do we know about the current extinction rate? Trends in Ecology & Evolution **8**:375-378.

Smith, F. and M. Robert 1993. Estimating extinction rates. Nature **364**:494-496.

Smith, P. and H. Sandstrom. 1988. Physical processes at the shelf edge in the Northwest Atlantic. Journal of Northwest Atlantic Fishery Science **8**:5-13.

Starr, M. and M. Zeleny. 1977. MCDM: State and future of the arts. Pages 5-9 in M. K. Starr, and M. Zeleny, editors. Multiple criteria decision making. Amsterdam, North-Holland.

Stewart, R. and H. Possingham. 2005. Efficiency, costs and trade-offs in marine reserve system design. Environmental Modeling and Assessment **10:**203-213.

Soulé, M. 1986. Conservation biology and the "real world". Pages1-12 in M. E. Soulé editor. Conservation biology. The science of scarcity and diversity. Sinauer Associates, Massachusetts.

Soulé, M. 1987. Introduction. Pages 1–10 in M. E. Soulé, editor. Viable populations for conservation. Cambridge University Press, New York.

Soulé, M., et al. 2003. Ecological effectiveness: Conservation goals for interactive species. Conservation Biology **17**:1238-1250.

Stillwell, W., D. Seaver, and W. Edwards. 1981. A comparison of weight approximation techniques in multiattribute utility decision making. Organizational Behavior and Human Performance **28**:62-77.

Svancara, L., R. Brannon, J. Scott, C. Groves, R. Noss, and R. Pressey. 2005. Policy-driven versus evidence-based conservation: A review of political targets and biological needs. BioScience **55:**989– 995.

Tear, T., et al. 2005. How much is enough? The recurrent problem of setting measurable objectives in conservation. BioScience **55:**835–849.

Tilman, D. 1994. Competition and biodiversity in spatially structured habitats. Ecology **75**: 2-16.

Transport Canada. 2007. Synopsis Report - Environmental oil spill risk assessment for the south coast of Newfoundland. 1st Edition. TP 14740E, Ottawa, 43 pp.

UN (United Nations). 1980. International convention for the safety of life at sea, 1974. London. Available from http://treaties.un.org/doc/Publication/UNTS/Volume1184/volume-1184-I-18961-English.pdf (accessed March 2013).

United Nations Environment Program (UNEP). 1992. Global biodiversity strategy. World Resources Institute, New York.

United Nations Environment Program (UNEP). 1995. Global biodiversity assessment. Cambridge University Press, Cambridge

Villa, F., L. Tunesi, and T. Agardy. 2001. Zoning marine protected areas through spatial multiple-criteria analysis: The case of the Asinara Island National marine Reserve of Italy. Conservation Biology **16**:515-526.

Villagarcia, M. 1995. Structure and distribution of demersal fish assemblages on the northeast Newfoundland Labrador Shelf. MSc. thesis. Memorial University of Newfoundland.

Wand, M. and M. Jones. 1995. Kernel smoothing. Monographs on statistics and applied probability. Chapman & Hall, New York.

Wiese, F. and Ryan, P. 2003. The extent of chronic marine oil pollution in southeastern Newfoundland waters assessed through beached bird surveys 1984–1999. Marine Pollution Bulletin **46:**1090–1101.

Wilhere, G. 2008. The how-much-is-enough myth. Conservation Biology **22**:514-517.

Willis, K. and H. Birks. 2006. What is natural? The need for long term perspective in biodiversity conservation. Science **314:**1261-1265.

Wilson, K., M. Westphal, H. Possingham, and J. Elith. 2005. Sensitivity of conservation planning to different approaches to using predicted species distribution data. Biological Conservation **22**:99–112.

Zucchini, W. 2003. Applied smoothing techniques, Part 1: Kernel density estimation.

World Resources Institute (WRI). 1992. Global biodiversity strategy. World Resources Institute, Washington, D.C.

Worm, B., et al. 2006. Impacts of biodiversity loss on ocean ecosystem services. Science **314**:787-790.

Zeleny, M. 1982. Multiple ranking decision criteria. McGraw-Hill Book Company, New York.

CHAPTER 4 A CONSERVATION DECISION-SUPPORT METHOD FOR MAKING TRADEOFFS AND HARD DECISIONS SPATIALLY EXPLICIT

4.1 INTRODUCTION

Global increase of human activities, including the increasing exploitation of coastal and marine resources, contributes to a global decline in marine biodiversity and ecosystem services (Roberts & Hawkins 1999, Crowder & Norse 2005, Worm et al. 2006, Halpern et al. 2008). Without effective policies, this trend is expected to continue during the 21st century (Pereira et al. 2010). The international community, cognizant of this situation, continues to push for more stringent conservation measures, including an effort to establish networks of marine protected areas (MPAs) that could protect important ecosystems from the direct impacts of human activities (UNEP 2004, COP X 2010). In this context, the development of systematic conservation planning (SCP) methods that can identify the best terrestrial and marine protected areas to create has been a subject of interest among conservation researchers and practitioners (Margules & Pressey 2000, Margules & Sarkar 2007, Moilanen et al. 2009).

For three decades, SCP methods have largely aimed to produce optimal solutions. The optimization models used are typically designed to generate a solution whereby biodiversity targets can be achieved at a minimal socioeconomic cost (Pressey & Nichols 1989, Margules & Pressey 2000, Stewart & Possingham 2005). Such an approach has been the cornerstone of modeling biodiversity objectives, being based on the concepts of representativeness, representation, and complementarity (Williams 2004, Moilanen 2008).

Other concepts used in SCP are also of social nature, such as effectiveness and flexibility (Kukkala & Moilanen 2012). Existing SCP tools are however limited in the way they consider socioeconomic factors, requiring competing costs to be aggregated and represented as a single cost (Moilanen et al. 2009). As a result, optimization models do not handle complex and competing socioeconomic costs, such as fishing, oil and gas, and transportation, in an effective and flexible way.

This limitation of optimization models results in several issues. First, it largely ignores the underlying compromises among the competing socioeconomic interests in conservation planning (Ban & Klein 2009, Adams et al. 2011). Non-monetary losses (e.g. sense of culture, varying levels of resilience of stakeholders) are hardly captured when costs are aggregated into a single measure of economic adverse impacts (Singleton 2009). Second, it can hardly inform the distribution of negative impacts to various stakeholders. For example, the use of an optimal model may be challenging in situations where different communities want an equitable distribution of the negative impacts of MPAs (Weeks et al. 2010).

One of the advantages of optimizations approaches is that they can generate numerous solution scenarios; each one achieves the stated goal, hence providing stakeholders options to discuss. However, it is assumed that a chosen scenario, to be biologically efficient, should be implemented at once (Meir et al. 2004). In practice, the networks of MPAs proposed by optimal models are rarely implemented all at once (Meir et al. 2004, Visconti et al. 2010). One reason is that other social factors, such as community and institutional capacity and lack of political will can overtake the implementation of MPAs and often lead to implementing only a portion of the optimal solution (Jameson et al. 2002, Svancara et al. 2008), making this subset not necessarily optimal in terms of biological conservation objectives.

Along with the popularity of optimization models, a planning approach based on the "win-win" principle, assuming that all stakeholders should win from a decision, has also been encouraged (Kiss 1990, IUCN 2002, Fisher 2012, MBI 2014). However, providing solutions that can benefit all stakeholders proves to be difficult at best, and impossible in many cases (Bailey & Jentoft 1990, Hulme & Murphree 2002, McShane et al. 2011). If all stakeholders cannot win, balancing loss and gain among stakeholders (i.e. negative impacts vs. gains of a policy) becomes important and controversial in conservation planning processes (Leader-Williams et al. 2010).

It is suggested that the competing interests in conservation policy typically require answering questions like "whose interest counts?" (Schmid 2002, Yates 2003), "who is the winner?" (Lackey 2006) or "conservation for whom?" (Kaimowitz & Sheil 2007). These questions indicate that decisions based on tradeoffs in which loss and gain compensate one another may not be the only decisions available for conservation planners. It also suggests that a hard decision (HD), an "either-or" type decision requiring planners to choose between incompatible options, can also commonly exist in conservation planning (Kooiman & Jentoft 2005, McShane et al. 2011).

The motivation of this study is to develop a decision-support method that addresses some of the limitations of optimal models and, at the same time, explores the practicability of the "win-win" option. Specifically, we aim at testing an alternative method designed to accommodate: (1) competing interests (not only among biological interests but also among socioeconomic interests) and (2) varying levels of tradeoffs (or lack thereof) between the conservation objectives.

In testing our proposed alternative method, the Spatial Tier Framework-Ordered Weighted Averaging (STF-OWA), we intend to better understand (1) the benefits of accommodating the socioeconomic competing interests, (2) the importance of varying levels of tradeoffs (or lack thereof) among competing conservation objectives, and (3) to what extent "win-win" solutions are a viable option to pursue in conservation planning.

Section 4.2 presents the case study selected to test our method, the Newfoundland and Labrador region, Canada. Section 4.3 introduces the conceptual, structural, and quantitative foundations of STF-OWA, based on the concepts of goal hierarchy and multi-criteria technique as implemented in a GIS-based environment. Then, we present the details of the workshop in which conservation priorities were elicited in preparation for the test-run of the STF-OWA. Finally, the section presents the methods for evaluating the STF-OWA scenarios. Section 4.4 presents the results of the case study concerning how conservation priorities agreed on at the workshop were made spatially explicit through the STF-OWA conservation scenarios. Section 4.5 presents a discussion on how STF-OWA scenarios can inform planning decisions. Finally, Section 4.6 concludes and draws on the relevance of our findings, discussing the increasing challenges that conservation planners and policy makers have to deal with if they are to achieve the global goals for MPAs.

4.2 STUDY REGION AND DATASETS

To provide a real application context for our method, the STF-OWA was tested in the context of the identification of potential conservation areas in the Newfoundland and Labrador (NL) continental shelf and slope bioregion in Canada. The NL region is located in the northwest Atlantic, representing one of the 12 major marine biogeographic regions in Canada (DFO 2009). Our study area included this bioregion as well as the Flemish cap area, an extension of the continental shelf outside of Canadian waters, covering a total of around 1.2×10^6 km^2.

Most of the study area, except inshore areas and the Northwest Atlantic Fisheries Organization (NAFO) Zone 2G (Figure 3.1), is data rich, being surveyed annually since 1977 under the multispecies benthic survey program of Fisheries and Oceans Canada (DFO) (Brodie 2005). Other major geo-referenced data, such as commercial landed catch, have also been recorded in this region.

Based on the spatial distribution of available data, the study area was divided into a 20 km resolution grid, resulting in about 3000 grid cells used as planning units (PUs). Grid resolution was selected based on the spatial distribution of the different datasets and is similar to the one used in other studies that looked at the same region (Edinger et al. 2007, Goulet et al. 2010).

In the STF-OWA framework, decision variables (i.e. spatial criteria) are evaluated for each individual PU. Existing geographic datasets (see Table 3.1) were collated from private and government agencies and organized by three themes: (1) biological, based on the twelve years of multispecies trawl surveys and historical and recent seabirds surveys, (2) socioeconomic, based on ten years of commercial fishing records, and (3) other marine uses, based on the ten years of fishing records, one year of marine commercial traffic, and data on oil and gas activities (Appendix 2).

The above thematic layers reflect the three broad conservation objectives explored in this study, which are (1) to maximize biodiversity preservation, especially in species-rich areas and areas with sensitive, rare, endemic, and endangered species; (2) to minimize adverse impacts on socioeconomic activities particularly on fisheries employment, fisheries landed value, number of fishing businesses, and fisher groups; and (3) to minimize adverse impacts on other marine uses such as marine transportation, oil and gas activities, and minimize areas that are negatively impacted by fishing gear and areas with gear conflict.

These three objectives are captured using 25 GIS-based attributes (see Figure 4.2 in Results section). Spatial (GIS-based) criteria were identified based on existing literature and on the availability of datasets. It should be noted that each spatial criterion in the STF is represented and evaluated quantitatively using a continuous value based on GIS-based raster cells, whereby each raster cell represents a PU. Details of data processing for generating the spatial criteria used in this study were presented in Chapter 3.

4.3 METHODS

This section is composed of three subsections: (1) explaining the technical details of the spatial tier framework-ordered weighted averaging (STF-OWA), the method developed in this study, (2) providing details about how the conservation priorities, used in testing the STF-OWA method, were elicited from a workshop, and (3) presenting the quantification of STF-OWA scenarios.

4.3.1 CONCEPTS AND QUANTIFICATION OF STF-OWA FOR CONSERVATION PLANNING

A decision-making process involving various and possibly competing objectives is a traditional problem tackled by the field of decision theory that has been approached by Keeney & Raiffa (1976) using the concept of goal hierarchy. This concept offers a framework that allows unpacking high-level objectives into lower level objectives. For example, a biodiversity conservation objective can be broken down into protecting seabirds and corals, by identifying areas where a variety of species concentrate. In this example, the high-level biodiversity objective is expressed through sub-objectives (i.e. protecting specific marine taxa) that are achieved using biodiversity attributes, such as a measure of species richness.

In decision analysis, objectives (or goals) are referred to as the desired state while attributes are similar to indicators of desired outcomes in the future (Newell & Simon 1972, Starr & Zeleny 1977). Hence, in the context of goal hierarchy, the low-level objectives can be seen as "means," to achieving a higher objective or "end" (Figure 4.1). In this paper, we refer to attributes and objectives as "criteria" *(*see items in Tier 1 to Tier 3, Figure 4.1). In multi-criteria decision analysis, (MCDA), criteria serve as the basis for which courses of actions or alternatives are evaluated. For this reason, "criteria" is a generic term that encompasses both the concepts of objectives and attributes (Malczewski 1999, p 82).

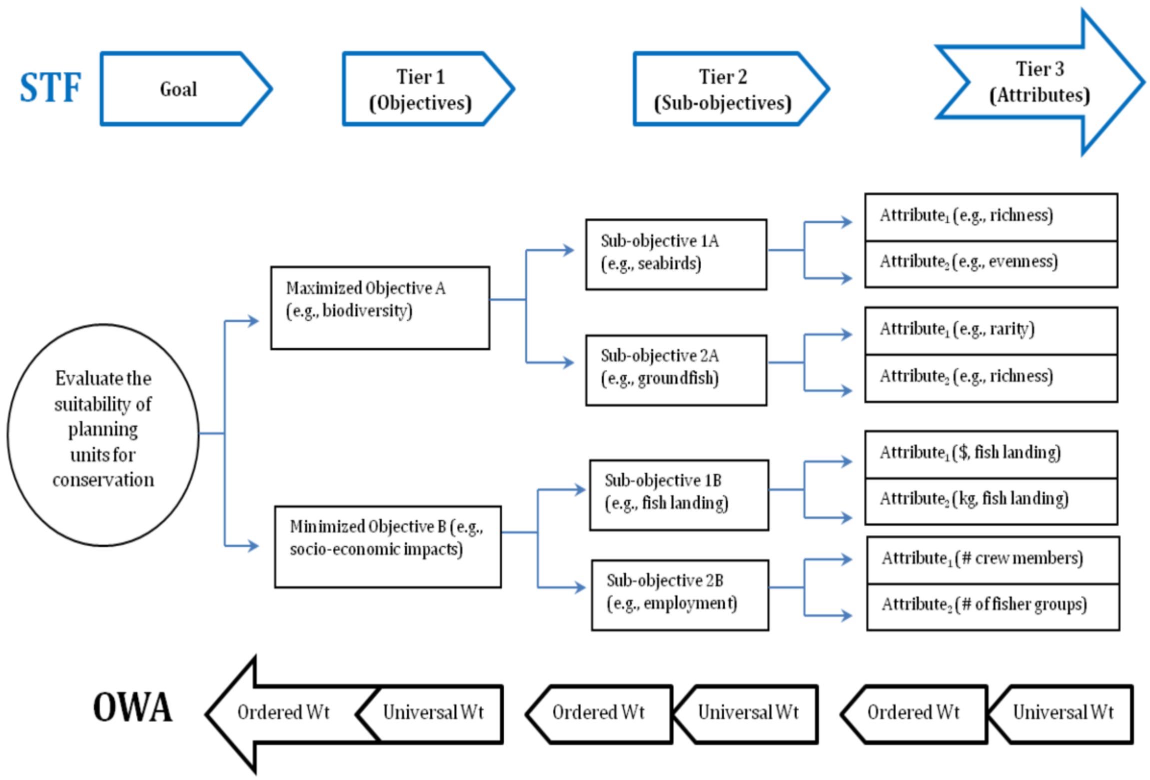


Figure 4.1 General organization of the Spatial Tier Framework - Ordered Weighted Averaging (STF-OWA) showing two directions: (1) how STF unpacks the goal into attributes and (2) how OWA aggregates the attributes leading to objectives and the final goal. OWA allows two ways to handle values through stakeholders' preferences (i.e. universal weight and ordered weight) among and between sets of criteria.

4.3.1.1 SPATIAL TIER FRAMEWORK: UNPACKING A CONSERVATION GOAL

This study considered identifying suitable areas for conservation as the high-level conservation planning objective, referred to as the conservation goal. Using the concept of goal hierarchy, a goal can be achieved by unpacking or breaking down the goals into small or measurable components called tiers: (1) objective, (2) sub-objectives, and (3) attributes. The numbers of tiers may vary depending on the appropriate levels necessary to unpack the high level-objectives. For example, the fishing objective was unpacked directly into attributes, while the biological objective was unpacked into two low-level criteria (see Figure 4.2 in Results section). It should be noted that each tier is composed of georeferenced data layers, hence referred to as spatial tiers. Thus, the spatial tier framework (STF) used in this study is defined as a framework for unpacking high-level spatially explicit conservation objectives into low-level spatially explicit attributes.

4.3.1.2 ORDERED WEIGHTED AVERAGING: INTEGRATING CONSERVATION PRIORITIES

The STF criteria can also be thought of as a hierarchy of values following the assumption that values are linked to objectives (Keeney 1992, Seip & Wenstop 2006). For example, when prioritizing conservation areas, decision-makers should consider identifying objectives based on stakeholders' value systems or preferences. In conservation planning these preferences are typically associated with how biodiversity gain and socioeconomic adverse impacts are regarded by stakeholders.

The Ordered Weighted Averaging (OWA) method (Yager 1988) was used to integrate stakeholders' preferences into the spatial tier framework. OWA is an MCDA method that can integrate stakeholders' preference in two ways: (1) by assigning weights to criteria, something done in most MCDA methods, but also (2) by identifying a level of tradeoff to apply among and between sets of criteria. The OWA method has been used extensively in decision-making (Emrouznejad and Marra 2014), including in combination with GIS (e.g. Malczewski 2006).

A few studies used OWA in terrestrial conservation (e.g. Valente and Vettorazzi 2008). While some work looked at using MCDA methods for marine conservation (e.g. Wood and Dragicevic 2007), this thesis presents the first use, to our knowledge, of OWA in marine conservation. Combined with the STF framework, the approach proposed in this thesis is called STF-OWA, combining the spatial tier framework of the goal hierarchy with a method that allows aggregating elements of each tier using various levels of tradeoff. The STF-OWA requires a clear statement as to the direction (or attainment) of a given objective, whether it is minimized or maximized in suitable areas for conservation (Malczewski 1999). SF-OWA minimizes objectives representing the cost or adverse impacts of conservation (e.g., foregone benefits) (Equation 4.1). Minimization means that PUs having minimal adverse economic impacts is preferred. In contrast, STF-OWA maximizes objectives considered as a benefit or conservation gain (e.g., biodiversity protection) (Equation 4.2). Maximization means that a PU presenting a high biodiversity is preferred.

$$X'_{ij} = \frac{X_j^{max} - X_{ij}}{X_j^{max} - X_j^{min}} \qquad \text{Equation 4.1}$$

$$X'_{ij} = \frac{X_{ij} - X_j^{min}}{X_j^{max} - X_j^{min}} \qquad \text{Equation 4.2}$$

Where X'_{ij} is the standardized score for PU *i* and criterion *j*, X_{ij} is the raw score, and X_j^{max} is the maximum score for criterion *j*, X_j^{min} is the minimum score for criterion *j*, $X_j^{max} - X_j^{min}$ is the range of a given criterion. The standardized values can range from 0 to 1.

As previously noted, the STF is used to break down a high-level goal into low-level/simpler and clearly attainable objectives. In contrast, to determine the suitable areas for conservation in the goal tier, OWA combines low-level criteria, within and between spatial tiers, into a high-level goal. This direction of criteria combination is indicated by the hollow arrow at the bottom of Figure 4.1.

OWA supports the combination of three operators: logical AND, logical OR, and the standard weighted linear combination (WLC). Logical AND and OR operations are non-compensatory aggregation methods in the sense that a single criterion is allowed to meet an outcome, hence, prohibiting the rest of the criteria to compensate (Eastman 2009). In GIS operations, the logical AND is similar to the intersection operations whereby the high priority areas are strictly required to meet all criteria while the logical OR is similar to the union operation whereby the high priority areas are required to meet at least one criterion. In this sense, the logical AND can be considered as a "risk-averse" operation, making sure that all objectives are satisfied, while the logical OR can be seen as a "risk-taking" operation, accepting the risk that not all objectives are satisfied in the priority areas.

WLC is the most commonly used decision rule in GIS-based MCDA (Malczewski 1999). In this operation, stakeholders' preferences are expressed using universal weights representing the level of importance of individual criteria. Universal weights typically range from 0 to 1 and for a set of criteria, a set of corresponding universal weights must sum to 1. For example, three criteria, A, B, and C, could be assigned weights of: A=0.3, B=0.6 and C=0.1, the sum of the weights being 1. The expert has in this case assessed that criteria B is twice as important as A, and A is three times as important as C. The universal weight assigned to each criterion is applied to all PUs of the study area. Therefore, WLC evaluates each PU based on the weighted average of all criteria. Given the normalized universal weights for attribute *j*, w_j, and the standardized criteria scores with respect to PU *i* and criteria *j*, X'_{ij}, where j = 1, 2, … *n*, WLC evaluates PU *i* as follows:

$$WLC_i = \sum_{j=1}^{n} w_j \, X'_{ij}, \text{ where } \sum w_j = 1 \qquad \text{Equation 4.3}$$

Tradeoffs involve balancing loss and gain. In general, tradeoffs are anchored to the notion of substitutability, where a low score on one criterion is compensatory (or substitutable) by a high score in another criterion (Jiang & Eastman 2000). Compensability is a concept supported by WLC. However, as explained in Table 4.1, WLC offers equal tradeoffs among the criteria. By combining the above three operators into a single OWA operation, varying different levels of tradeoff (or lack thereof) among criteria has become possible (Yager 1988, Jiang & Eastman 2000). It should be noted that by anchoring a tradeoff in the concept of substitutability, OWA implements substitutability in two different ways. First, when all criteria are allowed to substitute for one another, and as used in the study, OWA generates five various levels of tradeoffs. Second, when a single criterion is not allowed to be substituted by other criteria, OWA generates two cases of no-tradeoffs.

To implement these various levels of tradeoff and no-tradeoff calculations, OWA uses a different type of weighting mechanism called ordered weight. Ordered weights result from a quantitative manipulation of universal weights and an α

(alpha) parameter (Equation 4.5). The universal weights agreed on by stakeholders are specific to each criterion. In contrast, ordered weights are calculated using universal weights, arranged based upon the highest to lowest criteria scores that are specific to each PU and an α value.

Order weights were obtained using a fuzzy linguistic quantifier approach, where natural linguistic terms can be given equivalent formal quantitative expressions (Zadeh 1983). For example, a fuzzy subset of quantities (e.g. 0 to 1) can be associated with a corresponding set of fuzzy linguistic terms such as: "All" (i.e. all desired criteria must be achieved in the priority areas), "Most" (i.e. most of the desired criteria must be achieved in the priority areas), "Many" (i.e. many of the desired criteria must be achieved in the priority areas), "Half" (i.e. half of the desired criteria must be achieved in the priority areas), "Some" (i.e. some of the desired criteria must be achieved in the priority areas), "Few" (i.e. few of the desired criteria must be achieved in the priority areas), and "At Least one" or ALO " (i.e. at least one of the desired criteria must be achieved in the priority areas). The quantifier used in this study follows the operation suggested by Yager (1996 pp 49-73) (Equation 4.4).

$$Q(p) = p^{\alpha}, \alpha > 0 \qquad \text{Equation 4.4}$$

where Q is a linguistic quantifier (e.g. "All") represented within a unit interval [0, 1] and p expresses a set membership (or achieving a range of criteria in high priority areas) indicated by Q.

Table 4.1 Seven ordered weights (or linguistic quantifiers) corresponding to seven conservation scenarios. The parameter α is a quantity associated with a linguistic quantifier (e.g. All) or a set of order weights, see equation 4.4.

α	Linguistic quantifiers	Aggregation operators	Tradeoff and no-tradeoff rules	Tradeoff and no-tradeoff in siting conservation PUs
$\rightarrow \infty$	All	Logical AND (MIN)	No Tradeoff (No compensation among or between criteria) "**Most risk-averse**"	▪No tradeoff on lowest score, ***either*** the minimized socioeconomic score ***or*** *the* ***maximized*** biodiversity score. This means *no compromise* on ideal biological or socioeconomic criteria (i.e. lowest economic and highest biological score). ▪It finds the relatively high biological gain with the relatively low adverse impacts on socioeconomic activities. Thus, it searches for the relatively ***"win-win"*** PUs. ▪Case study result: The suitable areas for conservation are patchy and minimal at best.
10	Most	**	The *lower* criterion score compensates more	▪Very weak compromise on low socioeconomic scores *and* **high** biological scores.
2	Many	**	The *lower* criterion score compensates more	▪Weak compromise on low socioeconomic scores *and* **high** biological scores.
1	Half	WLC	*Equal* Tradeoff (Equal compensation among and between criteria) "**Neutral**"	▪Biological and socioeconomic criteria are ***equally*** compromised. ▪Case study result: It finds a mix of PUs with high biological gain and high adverse impact on socioeconomic impact activities.
0.5	Some	**	The *higher* criterion score compensates more	▪Weak compromise on high biological scores or *low* socioeconomic scores.
0.1	Few	**	The *higher* criterion score compensates more	▪Very weak compromise on high biodiversity scores or low socioeconomic scores.
$\rightarrow$ **0**	At Least One or ALO	Logical OR (MAX)	No Tradeoff (No compensation among and between criteria) "**Most risk-taking**"	▪No compromise on highest score ***either*** *the* maximized biological scores ***or*** the minimized socioeconomic scores. This means *no compromise on* whichever is the best criterion score (i.e. *either* highest biological score *or* lowest socioeconomic score). ▪Case study result: It tends to find PUs with the least adverse impacts on socioeconomic activities. Unfortunately, these PUs got relatively low biodiversity gain. Thus, it searches for the relatively ***'cheap'*** PUs.

Given the criteria universal weights, w_j, Equation 4.5 was used to obtain order weights, V_j, (Yager 1997, Malczewski 2006).

$$V_j = \left(\sum_{k=1}^{j} u_k\right)^{\alpha} - \left(\sum_{k=1}^{j-1} u_k\right) \qquad \text{Equation 4.5}$$

$\left(\sum_{k=1}^{j-1} uk\right)$ where u_k is the re-ordered w_j. V_j values range from 0 to 1, and $\sum V_j$ values for each set of order weights must equal to 1 (Yager 1996, Malczewski 2006).

In this study, OWA was calculated at a PU level. By multiplying the ordered weight with the re-arranged criteria values for planning unit *i*, Equation 4.6 obtains the OWA score for PU *i.* (Yager 1997, Malczewski 2006).

$$OWA_i = \sum_{j=1}^{n} \sum \left(\left(\sum_{k=1}^{j} u_k\right)^{\alpha} - \left(\sum_{k=1}^{j-1} u_k\right)^{\alpha}\right) Z_{ij} \qquad \text{Equation 4.6}$$

Where Z_{ij} is the rearranged X'_{ij}, a standardized criteria value at PU *i*. Z_{ij} values were obtained by re-ordering the standardized criteria values from highest to lowest (X'_{i1}, X'_{i2}, $X'_{i3} \ldots X'_{in}$). Obtaining OWA scores in this way would mean that the same criterion can be associated with different order weights across PUs. Appendix 1 explains in more details and with an example how OWA weights are calculated at the PU level.

This study used the seven linguistic quantifiers presented in Table 4.1, each with a corresponding ordered weight. These seven linguistic quantifiers correspond to various levels of conservation tradeoffs (i.e. five levels of tradeoffs and two cases of no-tradeoffs) as applied in identifying priority areas for conservation (Table 4.1). These seven linguistic quantifiers were implemented in the MCE-FLOWA tool of Boroushaki and Malczewski (2008), an ArcGIS extension used in this study.

4.3.2 WORKSHOP FOR GENERATING CONSERVATION PRIORITIES

To assess the usability of the STF-OWA approach, a one-day workshop was conducted in December 2012. Prior to the workshop, ethics approval was obtained. Out of 28 email invitations sent, 15 people accepted the invitation, including 4 academics, 5 environmental NGO employees, and 6 members of governmental agencies. Participants have direct knowledge about the biology and/or fisheries of the study area and/or limited or direct experience in conservation planning.

A workshop protocol was prepared and provided to the participants prior to the workshop (Appendix 3). This protocol provided participants with background information about the study, the STF-OWA method, and descriptions of the criteria involved in testing. Finally, in order to keep the workshop duration to a day, the universal and ordered weights for biological attributes in spatial Tier 3 were determined by the authors prior to the workshop.

The workshop schedule was divided into four parts: (1) presentation of STF criteria, (2) elicitation of participants' conservation priorities among the given STF criteria via the universal weight and ordered weight, (3) presentation of STF-OWA scenarios, and (4) collection of feedback on the utility of the STF-OWA method in setting conservation priorities.

The criteria that composed the STF were identified prior to the workshop based on the data available for the case study region. For these reasons, criteria were presented and discussed before the workshop participants were asked to discuss conservation priorities. The elicitation of values (or preferences), in the form of weights, was done in three ways. First, weights were assigned on an individual basis, whereby participants provided weights based on their individual preferences. Second, weights were based on groups, where all members of a group discussed and agreed on weights. Finally, weights were based on plenary discussion whereby the workshop facilitator got a satisfactory agreement on weights from all the participants. Two types of weighting schemes were used: (1) direct weighting where participants directly assigned weights to each criterion and (2) the analytic hierarchy process (AHP) (Saaty 1980). Weights, obtained using the AHP method, through the individual workshop participants, were averaged. Weights obtained in groups, using direct weighting, were further discussed and agreed upon in plenary session.

The workshop participants were guided with two questions to help them think about and translate their priorities into numerical weights. The first type of priority, expressed as universal weights, involved answering the question "How important is one criterion over the other?" The second type of priority, expressed as ordered weight, involved answering the question "How do you want each criterion, in each set of criteria, to tradeoff with one another?" To further guide the workshop participants in eliciting their answers to the second question, they were presented with illustrations of how tradeoffs are implemented in each of the seven linguistic quantifiers used by OWA (Table 4.1, Appendix 1 and 3). The second question was asked in order to combine, one at a time, the sets of criteria in tiers 2 and 3. After generating universal and ordered weights, the tool MCE-FLOWA was used for generating maps for each scenario. Then, maps representing three specific conservation scenarios ("All," "Half," "ALO") were presented at the end of the workshop for comparison and discussion. Finally, participants shared their feedback on the method used in the workshop and a short questionnaire (Appendix 4) was distributed to participants to collect formal feedback on the utility of the STF-OWA method for conservation planning.

4.3.3 METHODS IN ANALYZING STF-OWA SCENARIOS

As indicated in Figure 4.1, the goal tier of STF-OWA ultimately generates the spatially explicit evaluation of PUs regarding their suitability for conservation purposes. Seven sets of evaluations, corresponding to the seven conservation scenarios (i.e. All, Most, Many, Half, Some, Few, ALO) were generated for this study. To understand these scenarios, further analyses were conducted in three levels: (1) comparison of the seven conservation scenarios, (2) analyses of conservation scenarios based on percent area targets, and (3) identification and analyses of regions of interest (ROIs) identified from one of the conservation scenarios. To do these analyses, several calculations were used. These include quantifying suitability, biodiversity gain, adverse impacts on fishing activities, adverse impacts on other marine uses and cost-effectiveness of seven conservation scenarios, percent area targets, and ROIs.

To compare the seven conservation scenarios, a suitability score was derived from the OWA scores. Another method used to compare the conservation scenarios was through the use of percent area targets. These targets refer to four sets of area targets, namely, 5%, 10%, 20%, and 30% of the PUs. These percentages respectively contain the following PU counts: 54, 108, 216, and 324. For example, identifying the 5% area target means selecting 5% of PUs with the highest suitability scores. To further understand the STF-OWA conservation scenarios, we identified groups of thirteen adjacent PUs referred to as the regions of interest (ROIs). These ROIs were identified from the "Some" conservation scenario using 30% percent area target. The basis for choosing thirteen PUs for these ROIs was that thirteen is the relatively consistent number of adjacent PUs with higher suitability scores across the study area. The identified four conservation sites include Northern Labrador (NLab), Flemish Pass (FP), Southwest Grand Banks (SWGB) and the south coast of Newfoundland (SNfld) (Figure 4.8). The following sections show the calculations used in exploring the STF-OWA scenarios.

4.3.3.1 QUANTIFYING SUITABILITY

In a conservation scenario, a higher OWA score theoretically means higher suitability for conservation purposes. Hence, the suitability score referred to in the quantitative analyses, results, and discussion sections is the same as or is derived from the OWA score as presented below.

The suitability for PU i is the same as the OWA score for PU i. Suitability is also calculated beyond PU level to which the following equations apply. As shown below, the suitability scores were also obtained for the 28 combined scenarios generated by joining a conservation scenario and a percent area target (Equation 4.7). Suitability is also obtained based on the four percent area targets (Equation 4.8), and the four ROIs (Equation 4.9).

$$S_{kr} = \frac{\sum_{i=1}^{n} Z_{irk}}{n} \qquad \text{Equation 4.7}$$

Where Z_{irk} is the OWA score at PU i belonging to a conservation scenario r and percent area target k. Thus, S_{kr} (or suitability$_{kr}$) refers to the average of OWA scores of a conservation scenario r and a percent area target k.

$$S_k = \frac{\sum_{r=1}^{R} S_{kr}}{R} \qquad \text{Equation 4.8}$$

Where the total number of R conservation scenarios considered in this study was seven (i.e. R = 7). Hence, S_k (or suitability$_k$) refers to the average of OWA mean scores for a percent area target k.

$$S_q = \frac{\sum_{i=1}^{P} Z_{iq}}{P} \qquad \text{Equation 4.9}$$

Where Z_{iq} is the OWA score at PU i of an ROI q. The total number of PUs considered for each ROI was thirteen (i.e. P = 13). Hence, S_q (or suitability$_q$) refers to the average of OWA scores for ROI q.

4.3.3.2 QUANTIFYING BIOLOGICAL GAIN AND ADVERSE IMPACTS ON SOCIOECONOMIC ACTIVITIES

Biological gain and adverse impacts on socioeconomic activities were derived from the standardized scores of 15 biological attributes and 10 socioeconomic activities attributes. These standardized scores were generated using Equation 4.2. It should be noted that socioeconomic adverse impacts were calculated in two categories, namely, adverse impacts on fishing and on other marine uses.

Biological gain and adverse impacts were obtained for the four ROIs, the four percent area targets, the seven conservation scenarios and the 28 combined scenarios.

The biological gain and adverse impacts on fishing and on other marine uses were summed across four percent area targets (Equation 4.10, Equation 4.11, and Equation 4.12) and the seven conservation scenarios (Equation 4.13, Equation 4.14, and Equation 4.15).

$$B_k = \sum_{r=1}^{R} \overline{M_{rk}} \quad \text{Equation 4.10}$$

$$F_k = \sum_{r=1}^{R} \overline{N_{rk}} \quad \text{Equation 4.11}$$

$$O_k = \sum_{r=1}^{R} \overline{W_{rk}} \quad \text{Equation 4.12}$$

Where $\overline{M_{rk}}$, $\overline{MN_{rk}}$, $\overline{W_{rk}}$, respectively refer to the averages of the PUs' standardized scores for biodiversity, fishing, and other marine uses for a conservation scenario *r* and a percent area target *k.* The total number of R scenarios considered for this study was seven (i.e. R= 7). Hence, B_k, F_k, and O_k respectively refer to the summed averages for biological gain, fishing adverse impacts, and impacts on other marine uses across the seven scenarios for percent area target *k.*

$$B_r = \sum_{i=1}^{n} \overline{M_{ir}} \quad \text{Equation 4.13}$$

$$F_r = \sum_{i=1}^{n} \overline{N_{ir}} \quad \text{Equation 4.14}$$

$$O_r = \sum_{i=1}^{n} \overline{W_{ir}} \quad \text{Equation 4.15}$$

Where $\overline{M_{ir}}$, $\overline{N_{ir}}$, $\overline{W_{ir}}$ respectively refer to the averages of the standardized scores for biodiversity, fishing, and other marine uses at PU *i* of a conservation scenario *r.* Hence, B_r, F_r, and O_r respectively refer to the summed averages for biological gain, fishing adverse impacts, and impacts on other marine uses of a conservation scenario *r.*

The biodiversity gain and socioeconomic adverse impacts were also calculated, using averages (Equation 4.16, Equation4.17, and Equation 4.18). It should be noted that results from these calculations were used in calculating the relative cost-effectiveness of the percent area targets as presented in Section 4.3.3.3.

$$Bio_k = \frac{B_k}{R} \quad \text{Equation 4.16}$$

$$FI_k = \frac{F_k}{R} \quad \text{Equation 4.17}$$

$$OU_k = \frac{O_k}{R} \quad \text{Equation 4.18}$$

Bio_k, FI_k, and OU_k respectively refer to the biodiversity gain, adverse impacts on fishing and on other marine uses for percent area target *k* obtained as averages for the seven conservation scenarios.

Finally, the biological gain and the adverse impacts on fishing activities were determined for the 28 combined scenarios (Equation 4.19 and Equation 4.20) and for the four ROIs (Equation 4.21, Equation 4.22, and Equation 4.23).

$$Bio_{rk} = \frac{\sum_{i=1}^{n} \overline{M_{irk}}}{n} \qquad \text{Equation 4.19}$$

$$FI_{rk} = \frac{\sum_{i=1}^{n} \overline{N_{irk}}}{n} \qquad \text{Equation 4.20}$$

Where $\overline{M_{irk}}$, and $\overline{N_{irk}}$ respectively refer to the averages of biodiversity and fishing standardized scores at PU *i* belonging to a percent area target *k* and a conservation scenario *r.* Hence, Bio_{rk} and FI_{rk} respectively refer to the averages of the mean scores for biodiversity gain and adverse impacts on fishing activities for a percent area target *k* and a conservation scenario *r*.

$$Bio_{q} = \frac{\sum_{i=1}^{n} \overline{M_{iq}}}{P} \qquad \text{Equation 4.21}$$

$$FI_{q} = \frac{\sum_{i=1}^{n} \overline{N_{iq}}}{P} \qquad \text{Equation 4.22}$$

$$OU_{q} = \frac{\sum_{i=1}^{n} \overline{W_{iq}}}{P} \qquad \text{Equation 4.23}$$

Where $\overline{M_{iq}}$, $\overline{N_{iq}}$, and $\overline{W_{iq}}$ respectively refer to the averages of standardized scores for biodiversity, fishing, and other marine uses with respect to PU *i* and ROI *q*. The total number of PUs considered for each ROI was thirteen (i.e. P = 13). Hence, Bio_q, FI_q, and OU_q respectively refer to the averages of mean scores for biodiversity, fishing, and other marine uses with respect to an ROI *q*.

4.3.3.3 QUANTIFYING COST-EFFECTIVENESS

Cost-effectiveness was obtained as the ratio between biodiversity gain and socioeconomic adverse impacts. Higher scores mean higher cost-effectiveness. Since socioeconomic scores were categorized into two, namely, adverse impacts on fishing and adverse impacts on other marine uses, cost-effectiveness was separately calculated for each of these categories. Cost-effectiveness was determined for the following: four percent area targets (Equation 4.24 and Equation 4.25), 28 combined scenarios (Equation 4.26), and four ROIs (Equation 4.27 and Equation 4.28).

$$CE_k^{FI} = \frac{Bio_k}{FI_k} \quad \text{Equation 4.24}$$

$$CE_k^{OU} = \frac{Bio_k}{OU_k} \quad \text{Equation 4.25}$$

$$CE_{rk}^{FI} = \frac{Bio_{irk}}{FI_{irk}} \quad \text{Equation 4.26}$$

$$CE_q^{FI} = \frac{Bio_q}{FI_q} \quad \text{Equation 4.27}$$

$$CE_q^{OU} = \frac{Bio_q}{OU_q} \quad \text{Equation 4.28}$$

4.4 RESULTS

The results are presented in three sections below. Section 4.3.1 presents the conservation priorities considered for the case study following the universal and ordered weights elicited at the workshop. Section 4.3.2 presents the trends and the three categorizations of the seven conservation scenarios resulting from the seven ordered weights. Finally, Section 4.3.3 shows how STF-OWA can make conservation priorities more spatially explicit through the four ROIs.

4.4.1 CONSERVATION PRIORITIES

In the STF-OWA decision-support method, conservation priorities are expressed in value-based weights, especially the universal weights (level of importance) and ordered weights (levels of tradeoff and lack thereof).

4.4.1.1 RELATIVE IMPORTANCE OF CONSERVATION CRITERIA USING UNIVERSAL WEIGHTS

In Tier 3, as agreed upon by the authors, the attributes for biodiversity, such as richness, endemism, rarity, and sensitivity were generally considered more important (thus, given higher weights) than other attributes such as density, evenness, and species status (see Figure 4.2). Based on individual responses using AHP, employment (0.25) was identified as being the most important fishing attribute (i.e. highest weight), followed by fishing revenue (0.22), richness of fisher groups (0.16), number of fishing businesses (0.16), evenness of fisher groups (0.13), and landed biomass (0.11). Based on group weighting, gear impact (0.41) was the most important attribute for other marine uses, followed by oil and gas (0.32), marine traffic (0.16), and gear conflict (0.11).

In Tier 2, only the biodiversity criteria were aggregated into high-level criteria (sub-objectives) such as the five marine taxa. Based on individual direct weighting, the weights were determined to be the following, in order of importance: coral (0.31), groundfish (0.21), sponges (0.2), seabirds (0.14), and exploited invertebrates (0.14).

In Tier 1, based on plenary consensus, the workshop participants valued protecting biodiversity (0.7) several times more than protecting the fishing-based attributes (0.2), and seven times more than protecting or being concerned with other marine uses (0.1).

4.4.1.2 CONSERVATION TRADEOFFS USING ORDERED WEIGHT

The authors agreed on using the ordered weight "Half" to express tradeoffs among biodiversity attributes in Tier 3. The three other ordered weights were determined by the workshop participants based on consensus: "All" was agreed upon for fishing attributes and "Some" for other marine uses in Tier 3, and "Some" in trading off the five marine taxa (i.e. seabirds, groundfish, corals, sponges, and exploited invertebrates) in Tier 2. Finally out of the three objectives, in Tier 1, this case study ran the seven sets of ordered weights to generate the seven corresponding conservation scenarios. These seven conservation scenarios show how priority areas appear when different levels of tradeoffs are applied based on the risk-taking ("ALO," most risk-taking) and the risk-averse ("All," most risk-averse) attitudes of decision-makers. Figure 4.3 shows the three conservation scenarios representative of this continuum of seven conservation scenarios.

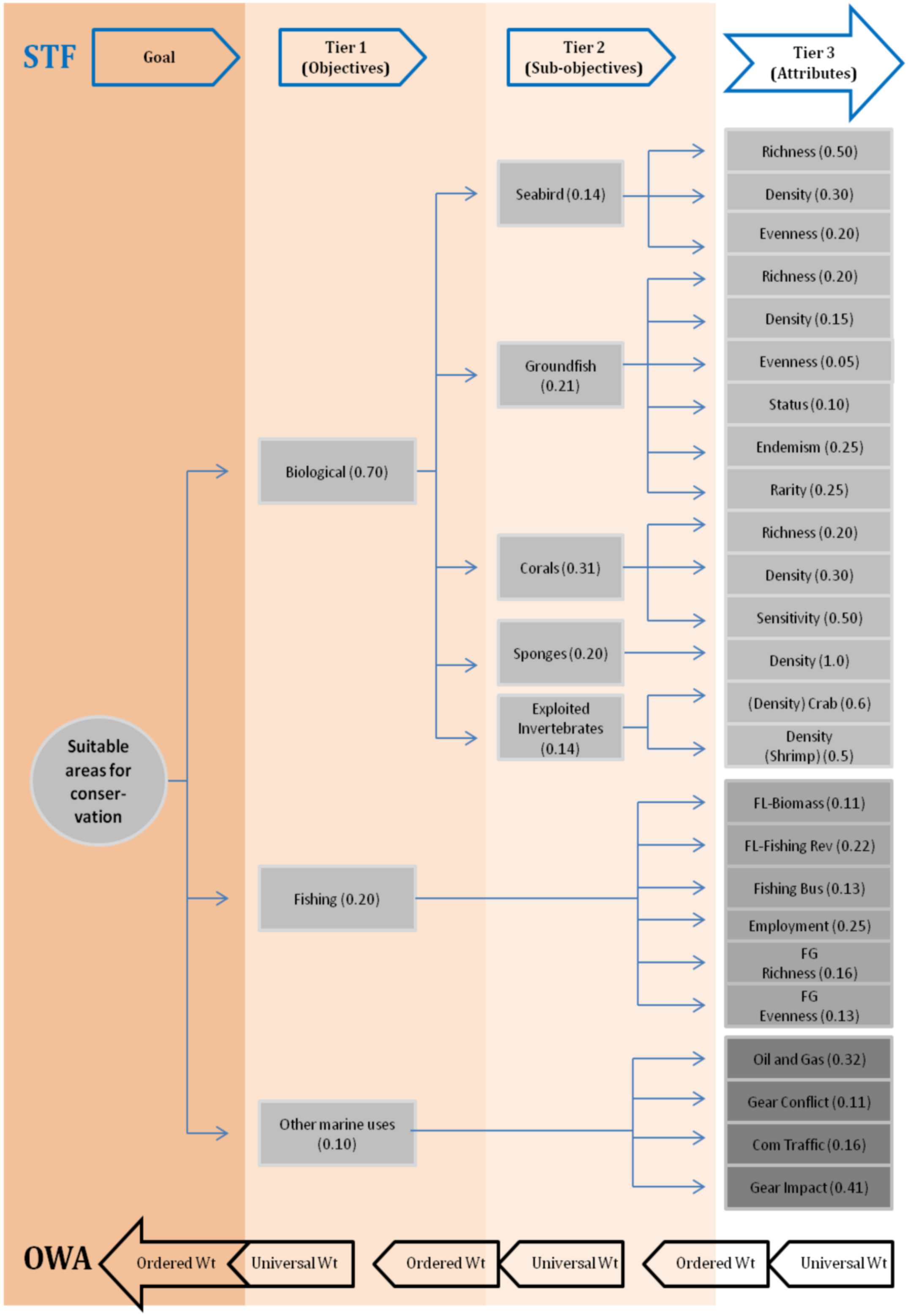


Figure 4.2 STF-OWA showing the categories and hierarchical arrangement of spatial criteria used in the study. The universal weights are shown for each criterion.

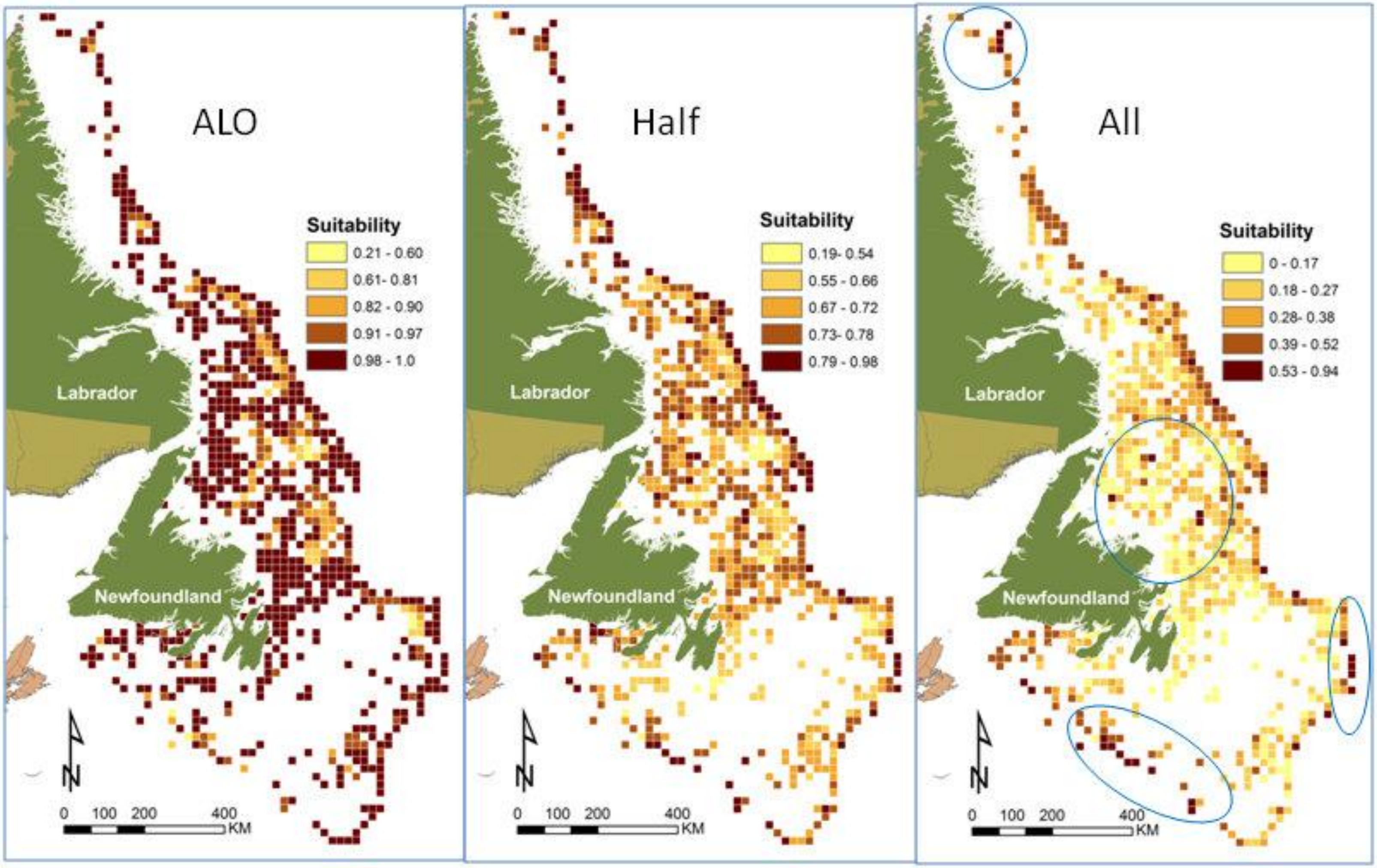


Figure 4.3 "At Least One (ALO)," "Half," and "ALL" respectively represent the risk-taking, neutral, and risk-averse scenarios. Areas circled in the "All" scenario (right) are areas with PUs displaying high suitability scores across seven scenarios.

4.4.2 ANALYSES OF THE SEVEN CONSERVATION SCENARIOS

With the ability of the STF-OWA to vary levels of tradeoffs (or lack thereof) among the three sets of objectives, we generated seven alternative conservation scenarios. In the following sections, we present the results in two parts. The first part shows the trends observed among the 28 conservation scenarios and four percent area targets. The second part presents the three categorizations of these seven conservation scenarios.

4.4.2.1 TRENDS ACROSS SEVEN CONSERVATION SCENARIOS

Figure 4.3 shows the suitability for conservation purposes of PUs. Conservation scenarios showed that the number of PUs with relatively high suitability scores (≥ 0.85) tends to decrease from "ALO" to "All" scenarios. When increasing the percent area targets, each scenario yielded decreasing suitability scores (Figure 4.4). Despite these decreasing trends of suitability scores, a small number of similar PUs scored ≥ 0.53 across the seven scenarios. The "All" scenario explicitly delineates these areas implying that only a small portion of the study area can be risk-averse, that is, PUs with relatively high biodiversity gain and low socioeconomic adverse impact .

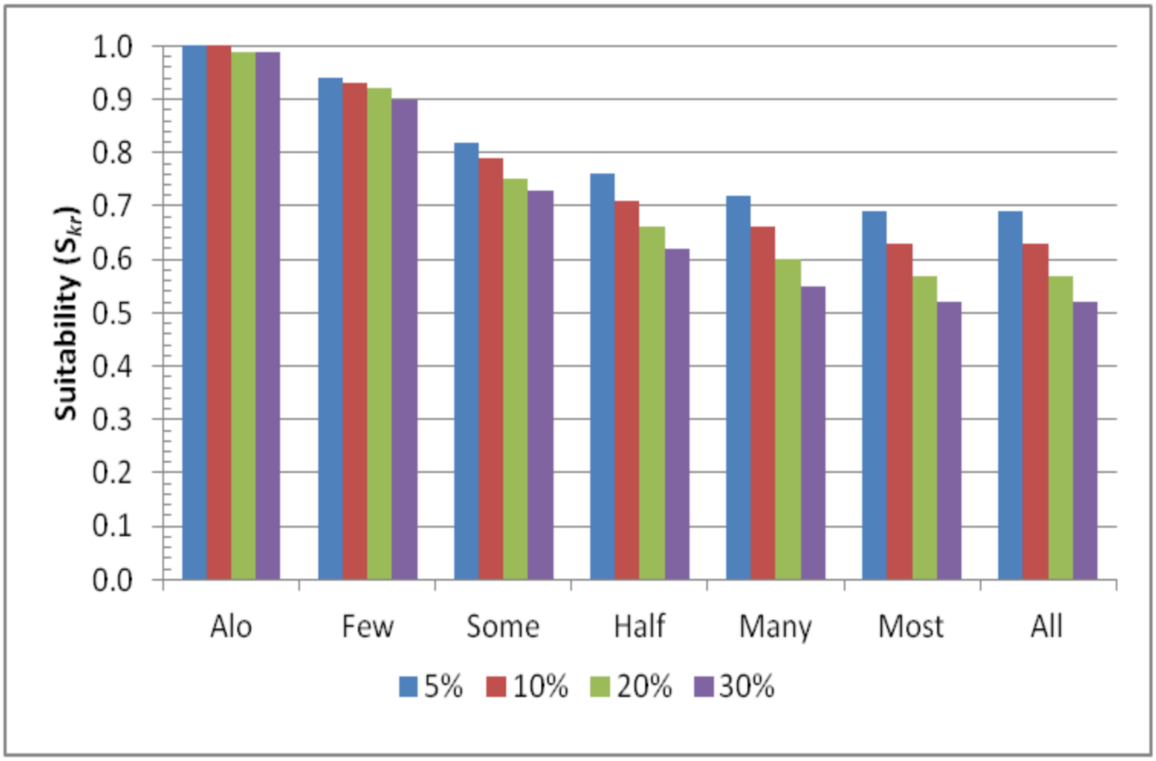


Figure 4.4 Suitability with respect to percent area targets and conservation scenarios.

Other results from applying various percent area targets (i.e. 5%, 10%, 20%, and 30%), showed four trends. First, the adverse impacts on socioeconomic activities and biodiversity gain showed positive linear increase with increasing percent area targets (Figure 4.5). Biodiversity gain was relatively lower when compared to adverse fishing impacts than when compared to other marine uses. Adverse impacts on fishing were higher than with other marine uses. Second, the average suitability across percent area targets decreased with increasing percent area targets (Figure 4.6).

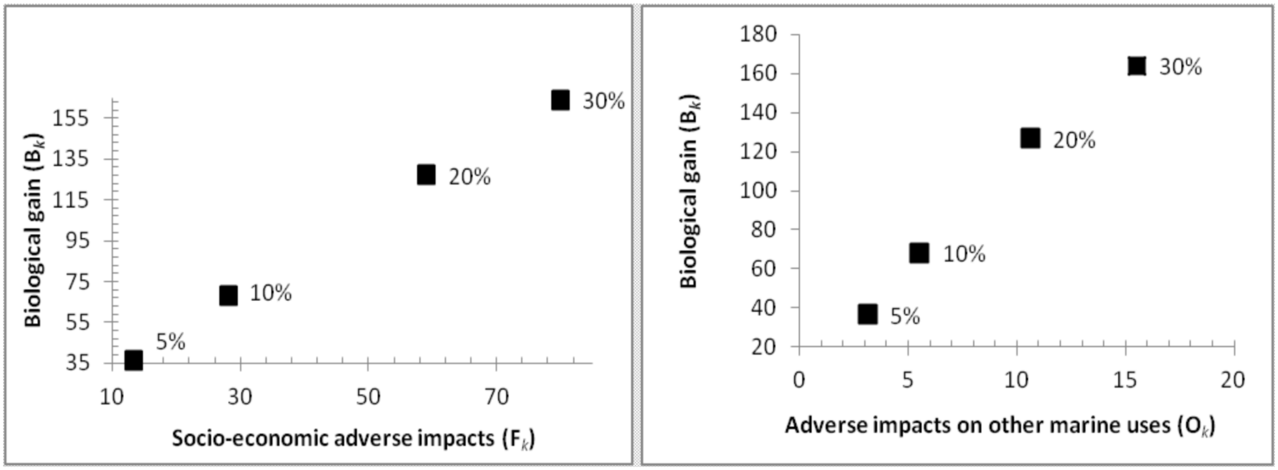


Figure 4.5 Biodiversity gain versus adverse impacts on fishing activities (left) and other marine uses (right) across various percent area targets.

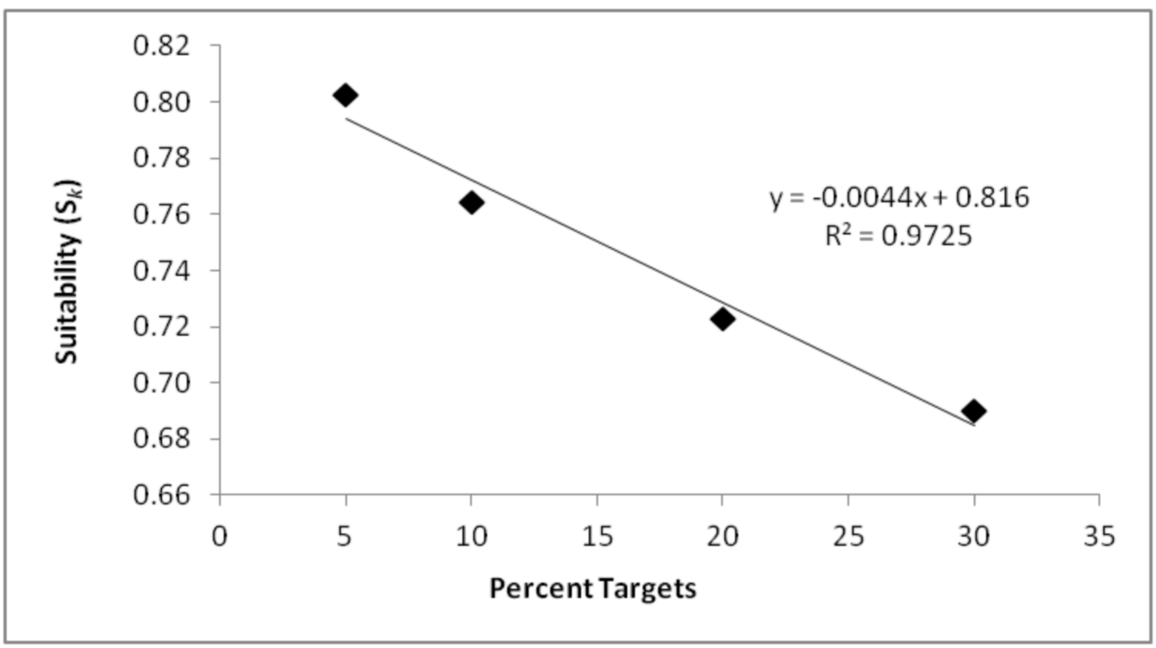


Figure 4.6 Average suitability with respect to percent area targets.

Third, the cost-effectiveness of conservation scenarios versus fishing activities showed a decrease with increasing percent area targets (Figure 4.7). This relationship is not as strong as with other marine uses. It should be noted that with percent area targets ≤ 20%, cost-effectiveness tends to be relatively within a small range, but it significantly dropped at the 30% target (Figure 4.7).

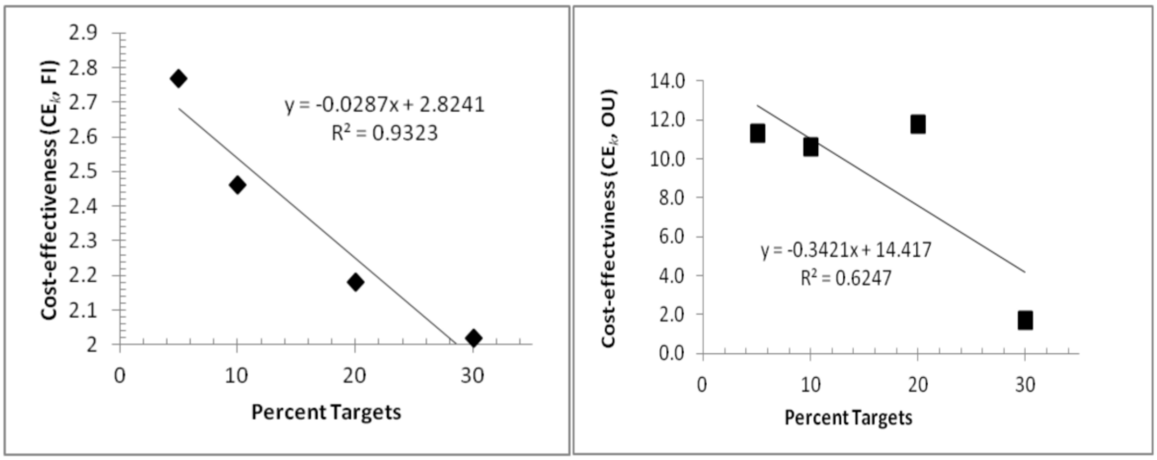


Figure 4.7 Cost-effectiveness of various percent area targets with respect to fishing impacts (left) and other marine uses (right).

4.4.2.2 CATEGORIES OF CONSERVATION SCENARIOS

The seven conservation scenarios provided by STF-OWA can be presented as three main groups: (1) the risk-taking scenarios ("ALO," "Some," and "Few"), (2) the neutral scenario ("Half"), and (3) the risk-averse scenarios ("All," "Most," and "Many").

THE RISK-TAKING SCENARIOS

When applying a 20% target, the two risk-taking scenarios (i.e. "ALO," "Few") tend to show PUs with high suitability scores in the northern region of the study area (i.e. the Labrador shelf and the southeast of the Newfoundland shelf) (Figure 4.8). The "Some" scenario, however, showed PUs with high suitability scores mostly on the shelf edge.

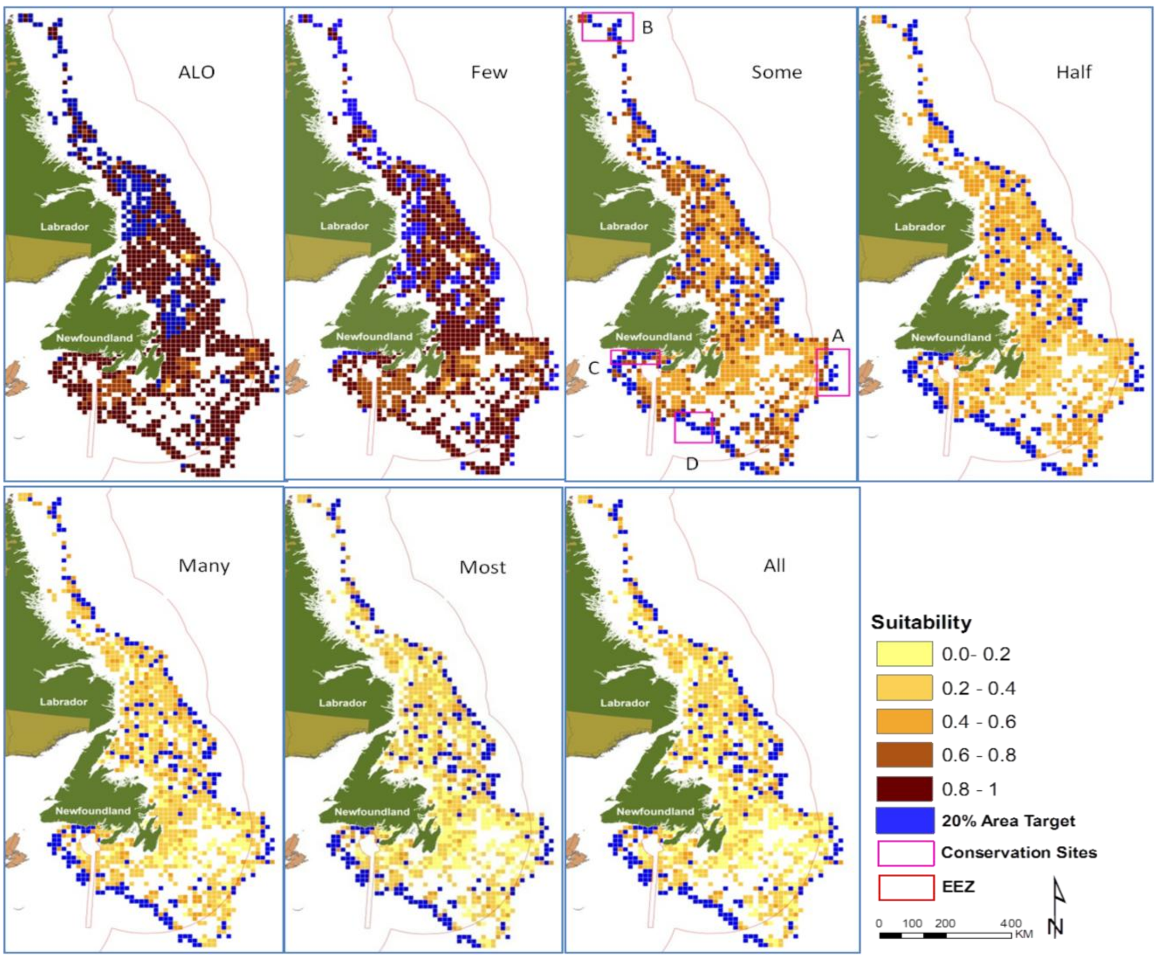


Figure 4.8 Seven conservation scenarios overlaid with 20% target and the four conservation sites. The four ROIs, labeled on the "Some" map, are: A - Flemish Pass (FP), B - Northern Labrador (NLab), C - South Newfoundland (inshore) (SNfld), and D - Southwest Grand Banks (SWGB).

When considering biodiversity gains with respect to socioeconomic adverse impacts, the "ALO" scenario tends to have the least number of socioeconomic impacts - that is, least adverse impacts on fishing and other marine uses and the least biodiversity gain across seven scenarios (Figure 4.9).

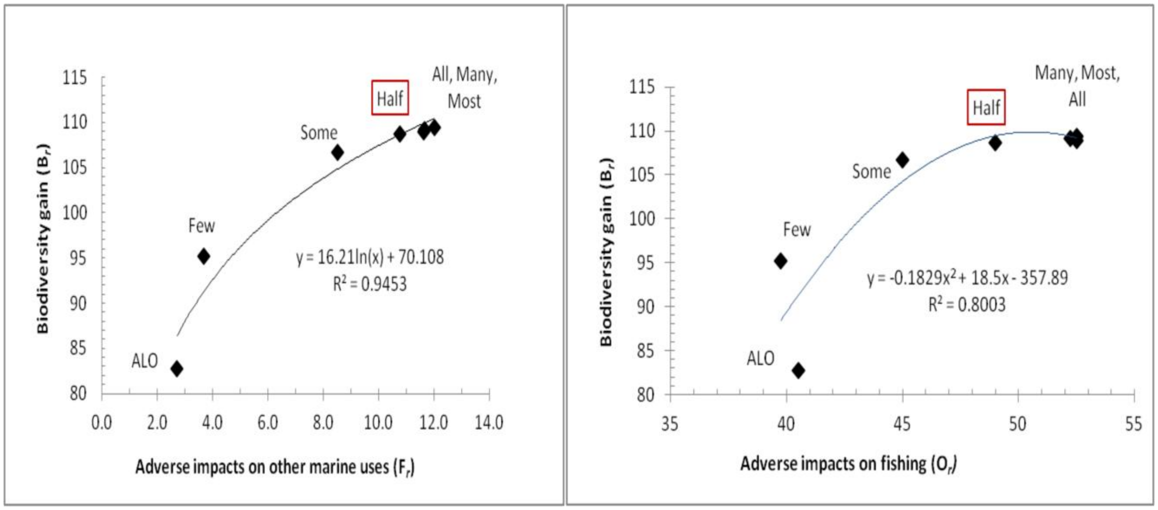


Figure 4.9 Biodiversity gain across seven scenarios with respect to negative impacts on fishing activities (left) and other marine uses (right)

For these reasons, "ALO" can be seen as a *cheap* conservation scenario from the economic and biological perspectives. Results showed that "Few" is relatively the most *cost-effective* scenario (i.e. offers the highest biodiversity gain for every unit of adverse impacts on socioeconomic activities) across seven scenarios (Figure 4.10)

The "Some" scenario also tends to be relatively cost-effective, being more cost-effective for 30% area targets than the "Few" scenario. Also, the biodiversity gain of the "Some" scenario shows a remarkable leap compared to the "Few" scenario (Figure 4.9). This, though, makes the "Some" scenario look like the "Half" and the more expensive scenarios discussed below (Figure 4.8).

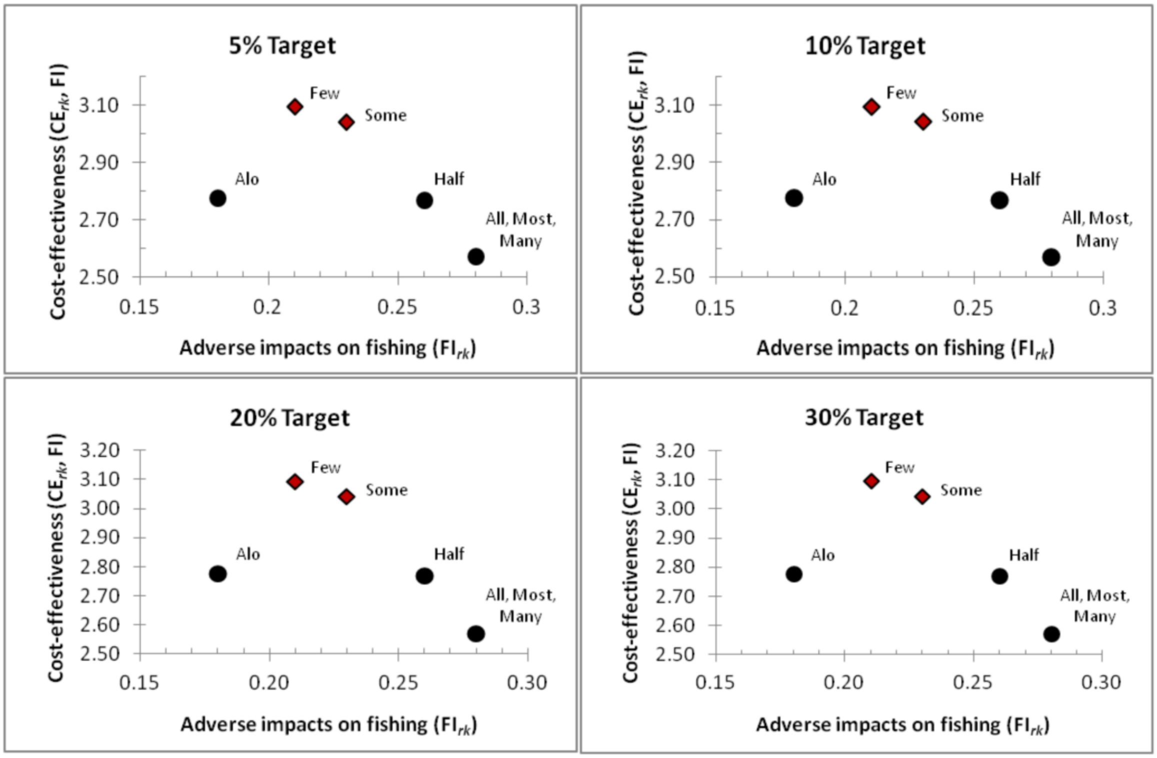


Figure 4.10 Relative cost-effectiveness of the seven conservation scenarios using various percent area targets (5%, 10%, 20%, 30%)

THE RISK-AVERSE SCENARIOS

When applying the 20% target, the three risk-averse conservation scenarios show that the PUs along the shelf edge and the northeastern Newfoundland shelf (i.e. around NAFO zone 3K) have relatively higher suitability scores than any other portion of the study area. The risk-averse scenarios showed similar biodiversity gain and a pattern of high suitable PUs as did the "Some" and "Half" scenarios (Figure 4.9). The difference is that the risk-averse scenarios have relatively higher fishing adverse impacts than "Half"; hence the risk-averse scenarios are relatively expensive scenarios.

THE NEUTRAL "HALF" SCENARIO

The "Half" scenario is called neutral as it trades off all criteria equally proportional to the criteria's capacity to compensate. This scenario also represents in full the pro-biodiversity conservation priorities from the workshop. Hence, it is noticeable that this scenario resembles the scenarios showing high biodiversity gain such as the "Some" and risk-averse scenarios (Figure 4.8).

4.4.3 ANALYSES OF REGIONS OF INTEREST

Four ROIs were identified and analyzed in more detail for their level of suitability and cost-effectiveness: the south coast of Newfoundland (SNfld), the Northern Labrador (NLab), the Southwest Grand Banks (SWGB), and the Flemish Pass (FP) (Figure 4.8). Figure 4.11 shows the cost-effectiveness and suitability scores of the four ROIs. The four ROIs are generally less cost-effective relative to fishing than for other marine uses. The NLab has the highest suitability score, followed by FP, SWGB, and SNfld (Figure 4.11A). Additionally, plotting ROIs against their relative biodiversity gain and relative cost-effectiveness (Figure 4.11B,C) shows how suitability scores explain conservation tradeoffs.

For example, SWGB ranks third (i.e. relatively less suitable for conservation) despite its high potential for biodiversity gain. SWGB is the least cost-effective ROI for both fishing and other marine uses, having a potential for the highest adverse impacts on fishing and other marine uses. The NLab received the highest suitability score, ranking second in terms of biodiversity gain, but having the least adverse impact on other marine uses and relatively low adverse impact on fishing.

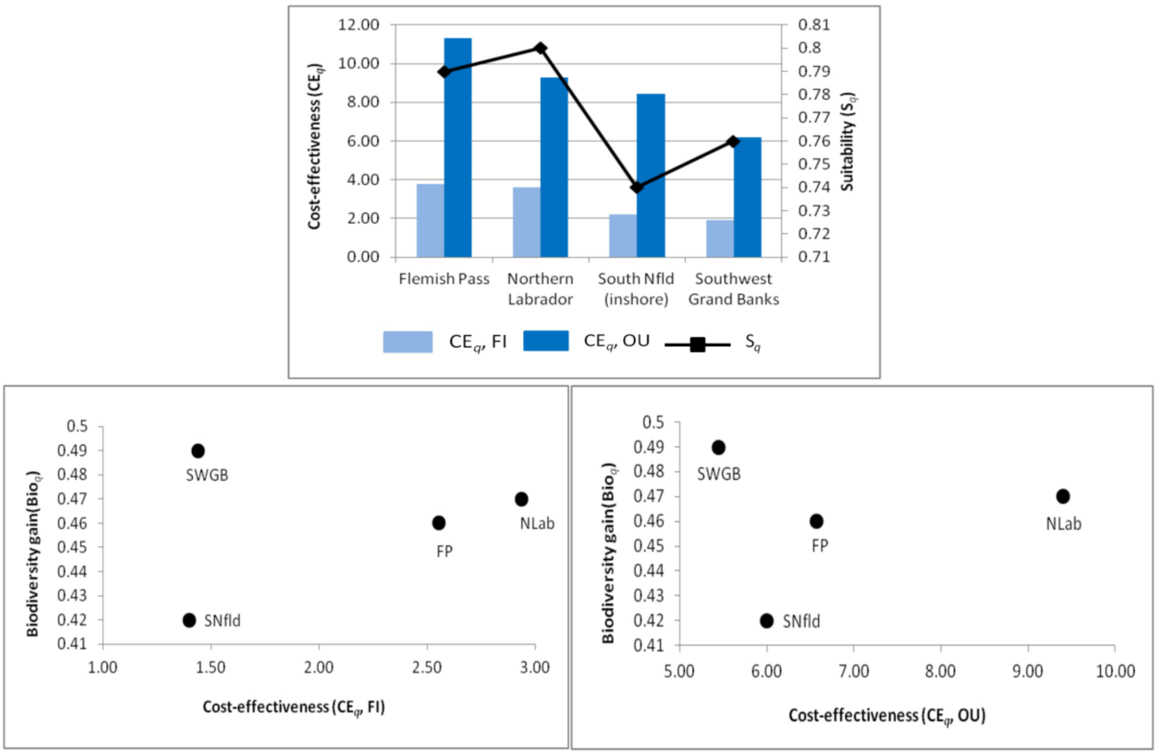


Figure 4.11 Suitability scores and the relative cost-effectiveness of the four ROIs (top). Plots represent biodiversity gain of ROIs versus the cost effectiveness for fishing (bottom left), and for other uses (bottom right)

With the STF-OWA model, low-level criteria (e.g. disaggregated six fishing attributes) can be visualized in comparison with the suitability scores. Results show that the suitability scores captured the priorities agreed upon at the workshop. It should be noted that the workshop participants weighted the combined social-based fishing attributes (i.e. employment, fishing business, fisher groups based on richness, and evenness) as being more important than the monetary value of fishing (i.e. fishing revenue). As a result, the NLab and FP, with relatively low social-based adverse impacts, received higher suitability scores (Figure 4.12). In contrast, SWGB and SNfld have relatively higher social-based fishing activities resulting to their low suitability scores.

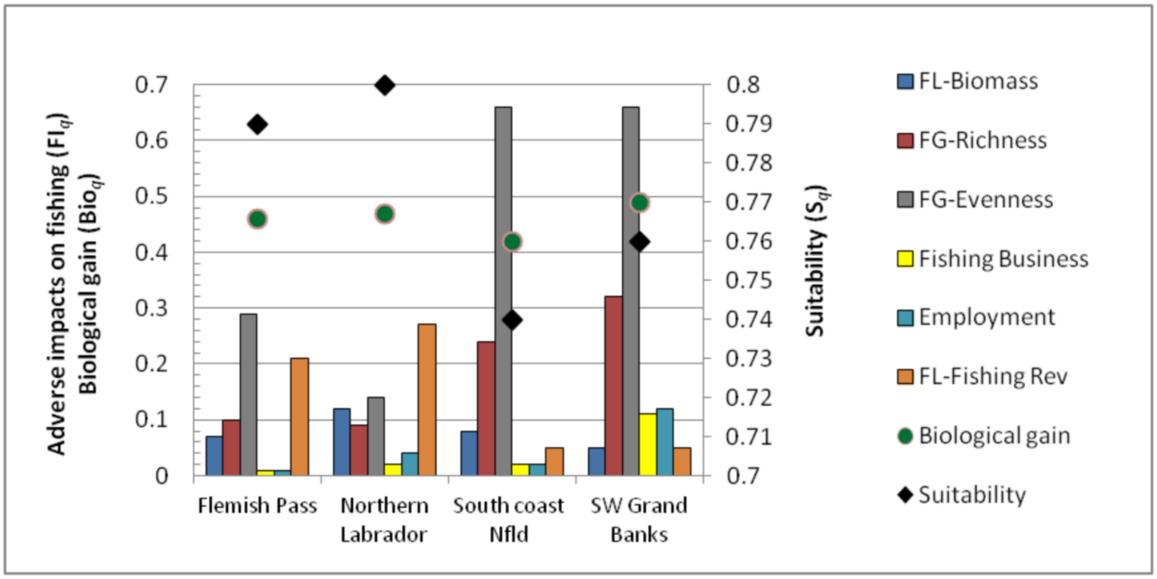


Figure 4.12 Four ROIs characterized for their suitability, biological gain, and the six fishing attributes

Finally, the STF-OWA scenarios can help make decision-makers aware of socioeconomic groups with potentially competing interests in the study region. In the NL bioregion, a greater number of fishers work in small to medium scale fisheries (e.g. inshore and near-shore fisheries), while a smaller portion participate in a large scale fishery (e.g. offshore shrimp fishery). The large-scale fishery is mostly captured in fishing revenue, while the small to medium scale fisheries are mostly captured in social-based fishing attributes.

Workshop participants gave higher importance (or weight) to fishing business, employment, FG-Richness, and FG-Evenness, giving ultimately a higher importance to areas that benefit high numbers of fishers (users). SNfld and SWGB are ROIs that are used by a variety of small to medium scale users (fishers). As a result, these areas came out as PUs with high adverse impacts on fishing (Figure 4.12). In contrast, the FP and the NLab were deemed the most suitable for conservation as they will adversely impact fewer users, but will have higher adverse impacts on fishing revenue. Clearly, with conservation priorities placing less importance on fishing monetary revenue, FP and SWGB came out with higher suitability. These results show that if monetary value were given a higher priority compared to social-based fishing attributes, the suitability ranking of the NLab and FP ROIs would likely change. Similarly, weighting monetary value higher than social-based fishing attributes would shift the adverse impacts on socioeconomic activities the small and medium scale fisheries.

4.4.4 WORKSHOP PARTICIPANTS FEEDBACK RESULTS

Out of the 15 participants, 13 provided feedbacks regarding the utility of the method STF-OWA. In terms of the usefulness of unpacking the conservation goal into detailed attributes, 91% agreed that the STF is useful in making criteria spatially explicit. Understanding the details of the concept of OWA was challenging: 8% found it difficult, 35% somewhat difficult and 59% of the participants were indifferent. Nevertheless, 83% of the participants found using weights useful (i.e. universal and order weights) in expressing conservation priorities. In terms of the utility of OWA in conservation planning 17% found it very useful, 33% found it useful, 17% found it somewhat useful, and the remaining 33% of participants were indifferent.

4.5 DISCUSSIONS

Many conservation planners have suggested that a conservation policy should aim to achieve both biological and social objectives (Christie 2004, White et al. 2006). In particular, competing social objectives need disaggregation as this could be important in the long-term success of a conservation policy (Adams et al. 2010). Most systematic conservation methods, however, have been focused on integrating competing biological objectives, with less consideration of competing social objectives (Ban & Klein 2009). Accordingly, this study explored and tested a systematic conservation planning method whereby the competing social interests are integrated, dependent on the available data, along with the competing biological criteria.

4.5.1 CONSERVATION PRIORITIES USING UNIVERSAL AND ORDERED WEIGHTS

After structuring all competing interests using spatially explicit criteria based on the STF, we demonstrated that using the universal and ordered weights can help integrate competing interests in a conservation planning method. This study shows that by doing so, both biological and social conservation priorities can be made spatially explicit in conservation scenarios. First, this was made possible by allowing stakeholders to discuss and assign conservation priorities among and between the given sets of criteria by assigning them the level of importance. For example, it was made explicit that the high suitability scores of PUs along the shelf edge result from the high level of importance associated with the attributes including richness, endemism, and rarity of groundfish and sensitivity of corals. This high importance of the shelf edge was reinforced when the biodiversity objective, in Tier 1, was weighted as more important than the aggregated two socioeconomic activities, namely fishing and other marine uses.

Aside from the relative importance of criteria weights, conservation priorities were integrated using the concept of tradeoff implemented through the ordered weights. Typically, a tradeoff-based model assumes that the criteria in question are exchangeable or compensable. Holland (2002, p 17) notes that with the concept of exchangeability, "tradeoff refers to the idea that, in any choice between a range of options, there is always a dimension of value in terms of which the options that we face may be compared as more, less, or equally to be preferred" [*sic*]. The tradeoff, in this sense, means that compensation is possible –that is, a loss in a criterion can be offset by another criterion (Jiang & Eastman 2000).

However when comparability of criteria is difficult to make, which can occur due to the incomparability of values (i.e. what the stakeholders care about), decision is beyond mere tradeoff (Jentoft & Kooiman 2005). In conservation planning, this decision is referred to as a hard choice, also called hard decision or HD in this thesis

(Schmid 2002, McShane et al. 2009). It is "hard", as any decision will necessarily disadvantage a criterion and its associated stakeholder group. In our case study, the no-tradeoff rules in STF-OWA were able to identify this type of decisions. A novel feature of the STF-OWA is its ability to show scenarios based on tradeoffs and HDs. Below, in Section 4.5.2, we discuss that by combining tradeoffs and HDs in a single method such as the STF-OWA, it is possible to explain what win-win and HD options could mean in a space-based evaluation of priority areas.

4.5.2 THE SEVEN CONSERVATION SCENARIOS MADE WIN-WIN, HD, AND OTHER OPTIONS SPATIALLY EXPLICIT

Our case study shows that the STF-OWA, as an alternative method, offered seven different possible scenarios, including the scenario generated by the commonly used WLC. Informed by the three types of stakeholders' attitudes (i.e. risk-taking, risk-averse, and neutral), planners can leverage biodiversity gain and socioeconomic adverse impacts in a spatially explicit manner.

The risk-taking scenarios implement tradeoff among highly performing criteria (i.e. "Few" and "Some") and no-tradeoff ("ALO") for the single best performing criterion in each PU. From a site prioritization perspective, these scenarios are risk-taking, as the performing criterion can come from either the biological or socioeconomic objective. Results showed that these scenarios tend to select the low socioeconomic adverse impact areas that also tend to have relatively low biodiversity gain. Hence, from a site prioritization perspective, the risk-taking scenarios could mean *risking biodiversity loss.*

The risk-averse conservation scenarios implement tradeoff among the low performing criteria ("Most", "Many") and no-tradeoff for the lowest performing criterion ("All") in each PU. From a site prioritization perspective, the risk-averse scenarios are *restrictive*, as they identify the highest possible conservation gain by keeping the socioeconomic adverse impact to a minimum. Technically, in the STF-OWA these are relatively the "win-win" scenarios. Our results showed, however, that risk-averse conservation PUs in the NL shelf and slope bioregions were patchy and minimal. When risk-averse scenarios are set to meet the > 5% area conservation target, it provided the relatively expensive PUs such as those along the NL bioregion's shelf edge. This region scored high both for socioeconomic activities and biodiversity, presenting a potentially high conflict area. In this respect, this region is likely to involve HDs if considered for conservation purposes.

The "Half" scenario implements an equal tradeoff or equal compensation among and between criteria (or objectives). Based on the NL case study, the "Half" scenario tends to lean toward the expensive risk-averse scenarios. The reason is that workshop participants assigned higher importance to biodiversity protection, which can lead to sacrificing socioeconomic activities when they overlap.

Moreover, in understanding loss and gain, the STF-OWA shows that that the risk-taking scenarios could also be viewed as "flexible" scenarios, accommodating flexibility in allowing and choosing among socioeconomic impacts in conservation scenarios. This allowed identifying two categories of risk-taking scenarios: (1) the "ALO" scenario, a relatively cheap option for which the Labrador shelf is an important region for conservation, and (2) the "Few" and "Some" scenarios, being relatively more cost-effective, which identified the Labrador shelf and some of the shelf edge of Newfoundland as suitable for conservation.

4.5.3 THE PROBLEM OF SPATIAL CONFLICT MADE SPATIALLY EXPLICIT

The STF-OWA scenarios show that varying the levels of tradeoffs (or lack thereof) between conservation objectives can help planners and policy makers understand the nature and spatial locations of the tradeoffs required when aiming for both high biodiversity protection and low adverse socioeconomic impacts. It should be noted that the STF-OWA trades off biodiversity with socioeconomic criteria with increasing "restrictiveness" or an attitude of being risk-averse as scenarios move from "ALO" to "All" (i.e. "ALO", "Few", "Some", "Half", "Many", "Most", and "All"). With the increasing restrictiveness, scenarios increasingly aim at achieving both objectives of maximizing biodiversity and minimizing socioeconomic adverse impacts.

The case study showed that when increasing restrictiveness, PUs get decreasing suitability scores, as achieving both objectives becomes more challenging. This challenge is not only evident across scenarios, but also across increasing percent area targets as shown by the evidence for decreasing cost-effectiveness and increasing adverse socioeconomic impacts. Scenarios requiring a higher percent area target will have to integrate PUs with lower suitability scores. The STF-OWA, in this case, reveals that achieving high biodiversity gain and higher percent area targets requires dealing with significant adverse impacts on socioeconomic activities.

Additionally, risking biodiversity loss as indicated by the risk-taking scenarios from the case study suggests two related inferences. First, areas with low biodiversity are associated with low socioeconomic activities, supporting previous studies suggesting that there is a high overlap between biodiversity and human activities (e.g. Salm and Clark 2000, Roberts et al. 2002, Sobel and Dahlgren 2004). Also, it confirms findings from recent studies suggesting that selecting low-adverse impact areas can be picking "low-hanging fruit", resulting in residual conservation areas (Devillers et al. 2014).

These trends across scenarios and percent area targets confirm the problem of conflicts resulting from spatial overlap between high biodiversity and high socioeconomic activities in the study area (see Section 2.6). It is worth noting that these overlaps are determined by predefined attributes rather than by modeling dynamic attributes such as spillover effects. This overlap is potentially severe as is evident in the low suitability score of the "All" scenario, as well as indicated by the affinity of the "Half" scenario with the expensive nature of the "All" scenario. This overlap, however, is higher and more consistent with fishing criteria than with other marine uses. The reason is that fishing activities are more widespread than other marine uses that tend to be confined to specific areas, such as commercial marine traffic and oil and gas activities. This suggests that identifying high-biodiversity areas for conservation purposes in the study area requires sacrificing a significant amount of socioeconomic activity, particularly in the fishing sector. Therefore, a decision to increase the biological goal and percent area target for conservation may depend on the capacity or willingness of stakeholders to sacrifice part of what they currently enjoy.

The pieces of empirical evidence described above support the problem of spatial conflict (i.e. spatial overlap between biodiversity and human activities) discussed in Section 2.6. They also support previous studies that showed that failing to recognize the limitations of resources and social issues can lead to very optimistic, conflicting, and confused objectives (Bailey & Jentoft 1990, Jones 2006, Christie & White 2007). Hence, a careful analysis of conservation tradeoffs should consider a variety of limits, as these limits ultimately set the achievability of a goal (Calabresi 1991).

The problem of spatial conflict is likely to increase as human extractive activities extend to deeper waters that host highly vulnerable species (Roberts 2002). If this trend continues, achieving high biodiversity gain on a global level is likely to involve increasingly expensive options. Hence, it may not be surprising that even an "efficient" solution generated by an optimal model tends to be expensive, as reported by Di Minin and Moilanen (2012). Nonetheless, this expensive nature of optimal solutions is not given the attention it deserves, not being sufficiently spatially explicit.

4.5.4 PRIORITIES MADE SPATIALLY EXPLICIT ACROSS REGIONS OF INTEREST

The STF-OWA uses a hierarchical representation of criteria (i.e. spatial tier framework) and a fuzzy scoring system (i.e. scores ranging from 0-1) to calculate the suitability scores of each PU. This quantification is an alternative to a binary scoring system that uses 0 or 1, commonly applied in optimal models for site prioritization. One of the limits of the binary system is that a conservation site can only be scored as being either included or excluded in a set of conservation priorities. Also, an optimal solution is theoretically not divisible into units. This implies that a solution (set of priority conservation areas) must be implemented all at once, which is a challenge in practice (Meir et al. 2004, Visconti et al. 2010).

In this case study, we demonstrated that the STF-OWA allows for identifying potential ROIs, visualizing priorities based on low-level criteria (e.g. attributes), and showing the impacts of conservation to potentially competing socioeconomic groups. By visualizing the six disaggregated fishing criteria, we can spatially and explicitly identify the potentially competing socioeconomic groups. Protecting the Northern Labrador (NLab) ROI, in the northern portion of the study area, would impact the large scale fishing industry, particularly the shrimp fishery. In contrast, the Southwest Grand Banks (SWGB) would impact small to medium scale fisheries.

These findings show that the STF-OWA "win-win" option may not be actually "win-win" for all stakeholders, highlighting one of the benefits of a more detailed analysis of impacts on socioeconomic criteria. This study implies that the STF-OWA output may have the capability to inform the moral and politically-based question: "conservation for whom?" This question is fundamental in all conservation policies; so is making the choice itself (Bailey & Jentoft 1990, Schmid 2002, Lackey 2006). Finding an answer to this question has proven to be difficult, especially if the ambiguities surrounding the planning process are not clarified (Oracion et al. 2005). However, the option to conceal or ignore this question can utterly lead to a decision failure (Nutt 2002), and it can compromise the long-term success of a conservation policy (Adams et al. 2010).

4.5.5 DECISION OPTIONS AND CONSERVATION DIRECTIONS

The STF-OWA provides conservation planners with alternative conservation scenarios where conservation gain is maximized and adverse socioeconomic impacts are minimized, based on a combined concept of tradeoff and HD models. When biodiversity conservation simultaneously and directly benefits people’s livelihood, win-win is possible and should be encouraged. However, such a scenario is not common in conservation (Salafsky 2011).

In this study, we call "win-win" an option where high biodiversity gain and a low impact on socioeconomic activities can be achieved simultaneously, acknowledging that everyone may not win. We should note that this definition of "win-win" anchors in loss and gain with reference to a status quo. It also differs from another common notion of "win-win" aiming for a "better" situation, by adopting strategies such as alternative livelihood, eco-friendly agricultural techniques, and compensation packages. These sorts of "win-win" efforts are instead viewed, in this thesis, as the means to addressing the consequences of HDs to mitigate the adverse impacts of conservation actions.

In the NL shelf and slope bioregion case study a win-win option was insufficient to meet conservation targets beyond 5%. HDs compromising a conservation objective (possibly to a significant extent) in exchange for gaining the other preferred conservation objective are likely the norm. HDs do not mean protecting biodiversity *only* or protecting socioeconomic activities *only*. Instead we showed that an "either-or" type of decision (an HD), as implemented by no-tradeoff rules in OWA (i.e. "ALO" and "All"), allowed understanding that these two preferred competing objectives cannot be ideally achieved to the same degree in most PUs.

In this chapter, we discussed how site prioritization can be viewed in terms of biological gain and socioeconomic adverse impacts. Such a framework can be illustrated using four quadrants (subdivisions) of decision options, where the biological gain and the socioeconomic adverse impacts can either be low or high (Figure 4.13). It should be noted that each decision option is relative to the others and can change depending on the biological and socioeconomic context in question.

The first quadrant (top-left of Figure 4.13) is the "win-win" option as defined in this thesis, providing high biological gain at a low socioeconomic cost. While this option still impacts existing socioeconomic activities, it attempts to achieve the least difficult compromise. In our case study, only about 5% of the PUs could be classified as "win-win"; some of these PUs were widely dispersed in location and thus are not appropriate in practice for conservation purposes (Figure 4.8). The second quadrant (top-right) identifies "hard conservation choices", providing high biological gain at a high socioeconomic cost. Our case study suggests that this option should often be used to complement a minimal number of "win-win" areas in order to provide sufficient conservation areas (e.g. reaching targets such as the 10% Aichi target).

The third quadrant (bottom-right) identifies "no-win" decision option, providing low biological gain with high socioeconomic adverse impact. Such an option is likely the least desired option in conservation planning exercises, bringing little conservation benefit at a high cost. Finally the fourth quadrant (bottom-left) identifies the "low-hanging fruit" decision option, providing lower biological gains at lower socioeconomic costs.

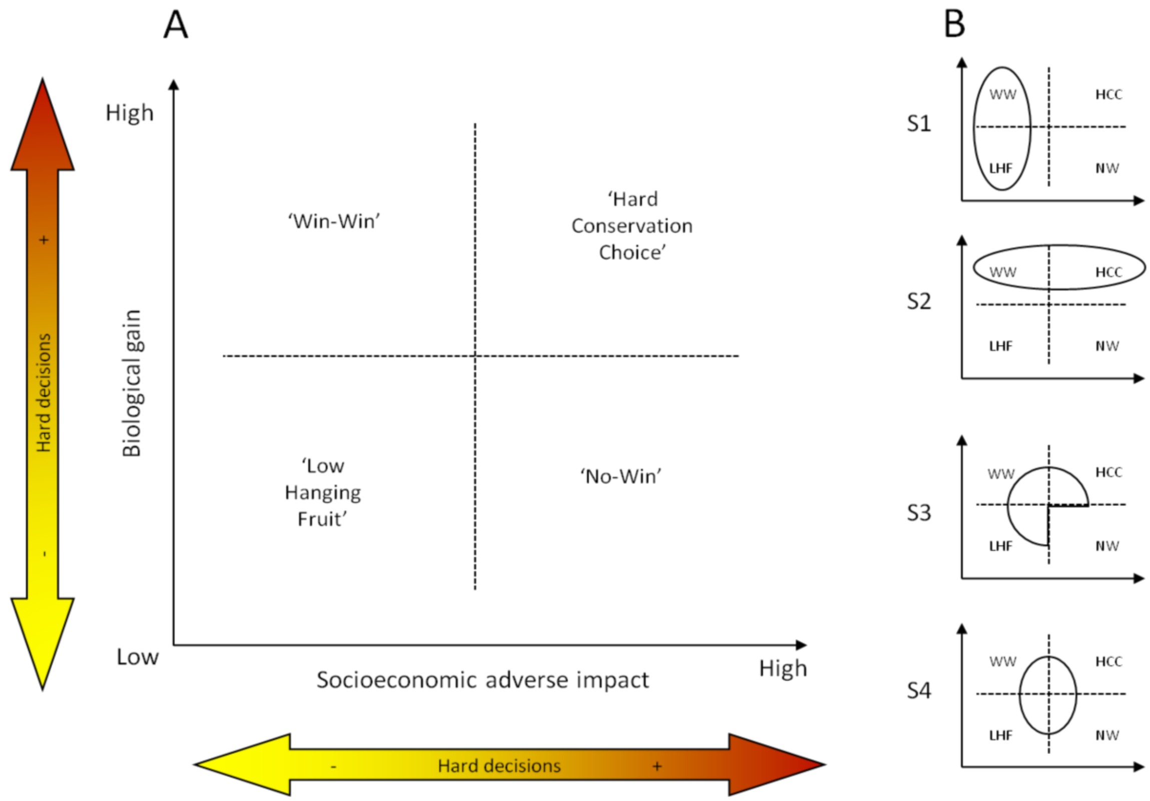


Figure 4.13 (A) Four decision options in site prioritization and their relative level of difficulty and (B) four conservation directions based on STF-OWA scenarios

Although these low hanging options tend to be selected in practical conservation planning exercises, as they often provide cheaper and larger areas than the rare "win-win" areas, they may not actually achieve the intended biodiversity protection, identified by Devillers et al. (2014) as residual MPAs.

In practice, conservation decisions will tend to combine different decision options that are hard on either socioeconomic or biological objectives or hard on both of these competing objectives. Figure 4.13B illustrates some examples of such a combination. *S1*, combining "win-win" and "low-hanging fruit" options, tends to be the preferred option in "real-world" conservation planning as it generally minimizes socioeconomic adverse impacts for variable conservation gain. While this approach is easier to implement due to higher stakeholder acceptance, it only offers a limited contribution to conservation objectives, risking biodiversity loss. *S2* ("win-win" and the "hard conservation choice" option) provides the highest biological gain but entails a socioeconomic adverse impact that is often not acceptable politically as it requires closing high-conflict areas for conservation. *S3* ("win-win," "hard conservation choice," and "low-hanging fruit") offers a more diverse range of options but can limit conservation benefits or have a higher socioeconomic impact. *S4* (that combines all four options) should be avoided as the "no-win" options are relatively the least appropriate choice to reach all of the objectives.

Identifying these conservation options and directions can help planners and policy makers explore explicitly the conservation gain with respect to its negative socioeconomic impact. This is a relevant process in investigating a bigger conservation question, "how much we can afford to pay" that will ultimately determine an achievable conservation target.

4.6 CONCLUSIONS

Conservation planning methods range from *ad-hoc* and expert approaches to very formal SCP methods based on optimization algorithms. While both types have respective strengths, they also have weaknesses that have been discussed in the literature. This study tested an approach called STF-OWA that aims at providing more rigor than expert-based planning, but more flexibility than conventional SCP methods in the way it can handle socioeconomic data. OWA has been used in urban decision-making and terrestrial conservation planning and this study presents the first use of OWA in marine conservation planning. Unlike conventional SCP methods, the STF-OWA can vary levels of tradeoffs (or lack thereof) among and between criteria in a spatially explicit manner. The use of the STF-OWA in a case study provided quantitative evidence that a "win-win" option is not always possible. Instead, it shows that varying levels of HDs are often necessary to meet a given conservation target.

We demonstrated that a continuum of conservation scenarios, from "risk-taking" to "risk-averse" can provide alternative scenarios that can help make biodiversity gain and adverse socioeconomic impact spatially explicit.

The STF-OWA scenarios demonstrated that maintaining low socioeconomic impacts may not be an effective approach to conserving marine biodiversity. First, comparatively, it can lead to risking biodiversity loss even at an area target below the minimum 10%. Second, it can identify "win-win" areas (i.e. high biodiversity gain with a low impact on socioeconomic activities), recognizing that, based on the case study, those areas are not sufficient to meet targets like the 10% Aichi target. We also found that these "win-win" areas can adversely impact certain groups of stakeholders more than others, highlighting the need to consider the distribution of socioeconomic impacts among sectors. Finally, the ideal objective of reaching high biological gain with insignificant adverse socioeconomic impact is rarely achievable.

Like many other countries in the world, Canada supports and adopts the global strategic plan for biodiversity that could help protect at least 10% of its waters before 2020, based on the Aichi biodiversity target. Our study shows that achieving this objective in the NL bioregion would cause considerable adverse impact to the fishing sector, involving HDs. Such HDs mainly result from spatial overlap between biodiversity and socioeconomic activities. This trend is not unique to NL waters and conflicts are likely to intensify globally as a result of increasing demand for marine-based resources.

Making "hard conservation choices" is difficult to justify both economically and socially for governments, leading to a global trend of picking "low-hanging fruit" where some variable conservation gain can be achieved at a very low socioeconomic cost. The NL region faces competing HDs, such as favoring large-scale fisheries over small-scale ones (i.e. minimizing foregone revenues over jobs), while small-scale fishers may be critical in supporting the small communities around the study area. The equitable management of protected areas, referred to in the Aichi Biodiversity Target 11, requires understanding such tradeoffs.

While the idea of HDs is not new in conservation planning, they have not been explored using this spatially explicit method. Success stories and best practices in protected areas planning were often not based on "win-win" decision options, but by successfully working with stakeholders who may not support the policy, making restrictions on certain socioeconomic activities, and providing long-term funding (Fernandez et al. 2005, Christie and White 2007, Cadiou et al. 2008, Gleason et al.

2012). The use of incentives and similar approaches (i.e. economic, interpretative, knowledge, legal, and participative) to get a balance when governing a policy such as the MPAs (e.g. Christie and White 2007, Jones et al. 2011, Knight et al. 2011) were essentially strategies designed to address the consequences of HDs. These incentives can, however, become more effective if they are explicitly tailored to addressing the adverse impacts of key HDs.

Finally, studies (e.g. Marshall et al. 2007, Sutton & Tobin 2012) about the resilience of stakeholders show that stakeholders are not equal in terms of their ability to embrace the negative impacts of conservation actions. This finding may help answer questions such as "conservation for whom," a question that often engenders HDs. In short, by explicitly identifying HDs, it is hoped that planners and policy makers can realize the importance of a strategic and proactive response to HDs when implementing MPAs. This view, however, is often de-emphasized under the more promising notion of "win-win".

4.7 LITERATURE CITED

Adams, V., B. Pressey, and R. Naidoo. 2010. Opportunity costs: Who really pays for conservation? Biological Conservation **143**:439–448.

Ban, N. and C. Klein. 2009. Spatial socioeconomic data as a cost in systematic marine conservation planning. Conservation Letters **2**:206-215.

Bailey, C. and S. Jentoft. 1990. Hard choices in fisheries development. Marine Policy **14:**333-344.

Boroushaki, S. and J. Malczewski. 2008. Implementing an extension of the analytical hierarchy process using ordered weighted averaging operators with fuzzy quantifiers in ArcGIS. Computers and Geosciences **34**:399-410.

Brodie, W. 2005. A description of the autumn multispecies surveys in SA2+ divisions 3KLMNO from 1995-2004. NAFO SCR Document 05/8. Northwest Atlantic Fisheries Organization.

Cadiou, G., et al. 2009. The management of artisanal fishing within the marine protected area of the Port-Cros National Park (northwest Mediterranean Sea): A success story? ICES Journal of Marine Science: **66**:41-49.

Christie, P. 2004. Marine protected areas as biological successes and social failures in Southeast Asia. American Fisheries Society Symposium **42**:155–164.

Christie, P. and A. White. 2007. Best practices for improved governance of coral reef marine protected areas. Coral Reefs **26**:1047–1056.

COP X (10th Meeting of the Conference of Parties). 2010. COP X/11 (Science-policy interface on biodiversity, ecosystem services and human well-being and consideration of outcome of the intergovernmental meetings). Nagoya, Japan. Available from http://www.cbd.int/decision/cop/default.shtml?id=12277 (accessed December, 2014).

Crowder, L. and E. Norse. 2005. The greatest threat: Fisheries. Pages 183-184 in A. Elliot and L. Crowder, editors. Marine Conservation Biology: The science of maintaining the seas biodiversity. Island Press, Washington.

Devillers, R., R. Pressey, A. Grech, J. Kittinger, G. Edgar, T. Ward, and R. Watson. 2014. Reinventing residual reserves in the sea: Are we favouring ease of establishment over need for protection? Aquatic Conservation: Marine and Freshwater ecosystems.

DFO. 2009. Development of a framework and principles for the biogeographic classification of Canadian marine areas. Science Advisory Report 2009/056. DFO Canadian Science Advisory Secretariat, Ottawa.

Di Minin, E. and A. Moilanen. 2012. Empirical evidence for reduced protection levels across biodiversity features from target-based conservation planning. Biological Conservation **153**:187-191.

Edinger, E., K. Baker, R. Devillers and V. Wareham. 2007. Coldwater corals off Newfoundland and Labrador: Distribution and fisheries impacts. WWF-Canada, Toronto.

Emrouznejad, A. M. Marra. 2014. Ordered weighted averaging operators 1988–2014: A Citation-Based Literature Survey. International Journal of Intelligent Systems **29**:994-1014.

Ferguson, J. 1990. The anti-politics machine: 'Development', depoliticization and bureaucratic power in Lesotho. University Minnesota Press, Minnesota.

Fisher, B. 2012. Conservation and livelihoods: Identifying trade-offs and win-wins. Frontiers in Ecology and the Environment **10**:343-343.

Fernandes, L., et al. 2005. Establishing representative no-take areas in the Great Barrier Reef: Large-scale implementation of theory on marine protected areas. Conservation Biology **19**:1733–1744.

Gleason, M., McCreary, M. Miller-Henson, J. Ugoretz, E. Fox, M. Merrifield, W. McClintock, P. Serpa, and K. Hoffman. 2010. Science-based and stakeholder-driven marine protected area network planning: A successful case study from north central California. Ocean & Coastal Management **53**:52–68.

Goulet, P., M. Walsh, A. Cuff, R. Devillers, and E. Edinger. 2010. Data analysis towards GIS-based ecoregion delineation in the NL shelf, 1995-2007. Final reports to Fisheries and Oceans Canada, St. John's, NL. St. John's, NL, Canada.

Halpern, B., K. McLeod, A. Rosenberg, and L. Crowder. 2008. Managing for cumulative impacts in ecosystem-based management through ocean zoning. Ocean & Coastal Management **51**:203-211.

Holland, A. 2002. Are Choices tradeoffs? Pages 17-34 in D. Bromley and J. Paavola, editors. Economics, ethics, and environmental policy: Contested choices. Blackwell Publishing, Oxford, UK.

Hulme, D., and M. Murphree. 2002. African wildlife and livelihoods: The promise and performance of community conservation. Canadian Journal of African Studies **36**:384-386.

Jameson, S., M. Tupper, and J. Ridler. 2002. The three screen doors: Can marine protected areas be effective. Marine Pollution Bulletin **44**:1177-1183.

Jiang, H. and J. Eastman. 2000. Application of fuzzy measures in multi-criteria evaluation in GIS. International Journal of Geographical Information Science **14**:173-184.

Jones, P. 2006. Collective action problems pose by no-take zones. Marine Policy **30**:143–156.

Jones, P., W. Qiu, and E. De Santo. 2011. Governing marine protected areas: Getting the balance right. United Nations Environment Program, Nairobi.

Kaimowitz, D. and D. Sheil. 2007. Conserving what and for whom? Why conservation should help meet basic human needs in the tropics. Biotropica **39**:567–574.

Keeney, R. 1992. Value focused thinking. Harvard University Press, Cambridge, Massachusetts.

Keeney, R. and H. Raiffa. 1976. Decisions with multiple objectives: Preferences and value tradeoffs. Wiley, New York.

Kiss, A. 1990. Living with wildlife: Wildlife resource management with local participation in Africa. World Bank Technical Paper, 217pp.

Knight, A., R. Cowling, A. Boshoff, S. Wilson, and S. Pierce. 2011. Walking in STEP: Lessons for linking spatial prioritizations to implementation strategies. Biological Conservation **144:**202-211.

Kooiman, J. and S. Jentoft. 2005. Hard choices and values. Pages 285-302 in J. Kooiman, M. Bavinck, S. Jentoft and R. Pullin, editors. Fish for life: Interactive governance for fisheries. Amsterdam University Press, Amsterdam.

Kukkala, A. and A. Moilanen. 2013. Core concepts of spatial prioritization in systematic conservation planning. Biological Reviews **88**:443-464.

IUCN 2002. Beyond Rhetoric: Putting Conservation to Work for the Poor. Gland, Switzerland.

Lackey, R. 2006. Axioms of ecological policy. Renewable Resources Journal **24**:13-16.

Leader-Williams, N., W. Adams, and R. Smith. 2010. Trade-offs in conservation: Deciding what to save. Wiley-Blackwell, Oxford, UK.

Malczewski, J. 1999. GIS and multicriteria decision analysis. John and Wiley Sons, Inc., USA.

Malczewski, J. 2006. Integrating multicriteria analysis and geographic information systems: The ordered weighted averaging (OWA) approach. International Journal of Environmental Technology and Management **6**:7-19.

Margules, C. and S. Sarkar. 2007. Systematic Conservation Planning. University Press, Cambridge.

Marshall, N., D. Fenton, P. Marshall, S. Sutton. 2007. How resource dependency can influence social resilience within a primary resource industry. Rural Sociology **72**:359–390.

McShane, T., et al. 2011. Hard choices: Making trade-offs between biodiversity conservation and human well-being. Biological Conservation **144**:966-972.

MBI (Monaco Blue Initiative). 2014. Monaco blue initiative from Yeosu to Rio+20. Secretariat, MBI. Available from http://www.biomarine.org/wpcontent/uploads/2012/03/MBI-From-Yeosu-to-Rio+20.pdf (accessed May 2015).

Margules, C. and R. Pressey. 2000. Systematic conservation planning. Nature **405**:203-253.

Maclean, J. 1992. Developing-country aquaculture and harmful algal blooms. Pages 252-284 in R. Pullin, H. Rosenthal and J. Maclean, editors. Environment and aquaculture in developing countries. ICLARM Conference Proceedings, Makati, Philippines.

Meir, E., S. Andelman, and H. Possingham. 2004. Does conservation planning matter in a dynamic and uncertain world? Ecology Letters **7**:615-622.

Meng, Y., J. Malczewski, and S. Boroushaki. 2011. A GIS-Based multicriteria decision analysis approach for mapping accessibility patterns of housing development sites: A case study in Canmore, Alberta. Journal of Geographic Information System **3:**50-61.

Moilanen, A. 2008. Generalized complementarity and mapping of the concepts of systematic conservation planning. Conservation Biology **22**:1655-1658.

Moilanen A., K. Wilson, and H. Possingham. 2009. Spatial Conservation Prioritization: Quantitative methods and computational tools. Oxford University Press, New York.

Newell, A. and H. Simon. 1972. Human problem solving theory. Prentice Hall, Inc., New Jersey.

Nutt, P. 2002. Why decisions fail: Avoiding the blunders and traps that lead to debacles. Berrett-Koehler Publishers, Inc., San Francisco.

Oracion, E., M. Miller, and P. Christie. 2005. Marine protected areas for whom? Fisheries, tourism, and solidarity in a Philippine community. Ocean and Coastal Management **48**:393-410.

Pereira, H., et al. 2010. Scenarios for global biodiversity in the 21st century. Science **330**:1496-1501.

Pressey, R., and A. Nicholls. 1989. Efficiency in conservation: Scoring versus Iterative approaches, Biological Conservation **50:**199-218.

Roberts C., et al. 2002. Marine biodiversity hotspots and conservation priorities for tropical reefs. Science **295**:1280-1284.

Roberts, C. 2002. Deep impact: The rising toll of fishing in the deep sea. Trends in Ecology and Evolution **17**:242-245.

Roberts, C. and J. Hawkins. 1999. Extinction risk in the sea. Trends in Ecology and Evolution **14**:241-246.

Saaty, T. 1980. The analytic hierarchy process: Planning, priority setting, resource allocation. McGraw Hill, New York, New York.

Salafsky, N. 2011. Integrating development with conservation: a means to a conservation end, or a mean end to conservation? Biological Conservation **144**:973-978.

Salm, R., J. Clark, and E. Siirila. 2000. Marine and Coastal Protected Areas: A guide for planners and managers. 3rd edition. IUCN, Gland Switzerland.

Schmid, A. 2002. All environmental policy instruments require a moral choice as to whose interests count. Pages 133-147 in D. Bromley and J. Paavola, editors. Economics, ethics, and environmental policy: Contested choices. Blackwell Publishing, Oxford, UK.

Singleton, S. 2009. Native people and planning for marine protected areas: How "stakeholder" processes fail to address conflicts in complex, real-world environments. Coastal Management **37**:421-440.

Sobel J., C. Dahlgren. 2004. Marine reserves: A guide to science design and use. Island Press, Washington DC.

Seip, K. and F. Wenstop. 2006. A primer on environmental decision-making: An integrative quantitative approach. Springer, The Netherlands.

Starr, M. and M. Zeleny. 1977. MCDM: State and future of the arts. Pages 5-9 in M. K. Starr, and M. Zeleny, editors. Multiple criteria decision making. Amsterdam, North-Holland.

Stewart, R. and H. Possingham. 2005. Efficiency, costs and trade-offs in marine reserve system design. Environmental Modeling and Assessment **10:**203-213.

Sutton, S. and R. Tobin. 2012. Social resilience and commercial fishers' responses to management changes in the Great Barrier Reef Marine Park. Ecology and Society **17**:1-10.

Svancara, L., R. Brannon, J. Scott, C. Groves, R. Noss, and R. Pressey. 2005. Policy-driven versus evidence-based conservation: A review of political targets and biological needs. Bioscience **55**:985-995.

UNEP (United Nations Environment Program). 2004. Conference of the parties to the convention on biological diversity. 7th Meeting. Kuala Lumpur, Malaysia.

Valente, R. and C. Vettorazzi. 2008. Definition of priority areas for forest conservation through the ordered weighted averaging method. Forest Ecology and Management **256**:1408-1417.

Visconti, P., R. Pressey, D. Segan, B. Wintle. 2010. Conservation planning with dynamic threats: The role of spatial design and priority setting for species' persistence. Biological Conservation **143:**756–767.

Weeks, R., G. Russ, A. Bucol, and A. Alcala. 2010. Incorporating local tenure in the systematic design of marine protected area networks. Conservation letters **3:**445-453.

White A., P. Alino, and A. Meneses. 2006. Creating and managing marine protected areas in the Philippines. Fisheries improved for Sustainable Harvest Project, Coastal Conservation and Education Foundation Inc, University of the Philippines Marine Science Institute, Philippines.

Williams, J., C. ReVelle, and S. Levin .2004. Using mathematical optimization models to design nature reserves. Frontiers in Ecology and the Environment **2**:98-105.

Worm, B. et al. 2006. Impacts of biodiversity loss on ocean ecosystem services. Science **314:**787-790.

Yager, R. 1988. On ordered weighted averaging aggregation operators in multicriteria decisionmaking. Systems, Man and Cybernetics, IEEE Transactions on Systems, Man, and Cybernetics **18**:183-190.

Yager, R. 1996. Quantifier guided aggregation using OWA operators. International Journal of Intelligent Systems **11**:49-73.

Yager, R. 1997. On the Inclusion of Importances in OWA Aggregation. Pages 41-59 in R. Yager and J. Kacprzyk, editors. The Ordered Weighted Averaging Operators: Theory and applications. Kluwer Academic Publishers, Boston.

Yates, F. 2003. Decision management: How to assure better decisions in your company. 1st Edition, John Wiley & Sons, Inc. California.

Zadeh, L. 1983. A Computational Approach to Fuzzy Quantifiers in Natural Languages. Computers and Mathematics with Applications **9**:149-184.

CHAPTER 5 CONCLUSIONS

Conservation planning typically involves integrating various and often incompatible priorities. Scholars have suggested for an integration and disaggregation of diverse socioeconomic interests in conservation planning to enhance the success of MPAs implementation and possibly design a more equitable MPA (Adams et al. 2010, Gurney et al. 2014). Most work in designing conservation planning methods and tools has focused on optimizing conservation gain for a minimal cost, and has not received the same level of attention in integrating the socioeconomic data (Ban & Klein 2009). Inspired by the decision and planning theories, this thesis presents a new method called the STF-OWA (Spatial Tier Framework – Ordered Weighted Averaging) for integrating diverse biological and socioeconomic interests within a systematic conservation planning process. The application of this method was illustrated and tested using an extensive dataset for the Newfoundland and Labrador region of Canada.

5.1 SUMMARY OF FINDINGS

This thesis aimed at answering four research questions mentioned in the introduction. The first question was: "What are the practical and theoretical bases for encouraging different types of methods in conservation planning, and on what grounds do multi-criteria methods deserve exploration in spatial planning?" To answer this question, we presented in Chapter 2 a review of existing methods in conservation planning, highlighting methods, strengths and weaknesses in practice and in theory.

First, the review emphasized the multi-dimensionality (i.e. biological, socioeconomic, governance) of conservation. While several methods can be used to represent and measure these multiple dimensions, no single method is capable of combining them into a meaningful single measure. This fact is not trivial as it has resulted in different ways of setting goals for identifying protected areas. These measurements can be categorized into two groups as identified in the review, namely, the objective, rule-based, scientific approach and the subjective, value-laden, social-based approach.

Second, we anchored in a theoretical perspective of bounded rationality, because decision-makers will not find all possible solutions. Another related theory is associated with the dilemmas of conservation planning which asserts that pinpointing "the problem" is difficult and which, in turn, results in a planning dilemma of identifying which method is most suitable to address the problem. The need to understand conservation losses and gains in spatial planning, especially with increasing evidence that losses are expected in order to achieve conservation gains, led to the necessity to explore a multi-criteria method. Finally, this chapter illustrated the importance of making tradeoffs and hard decisions (HDs) explicit, and argued for novel thinking in conservation planning.

Chapter 3 presented the datasets and data processing methods. Two of the challenges associated with adding socioeconomic interests in planning include the increasing need for data and the technical expertise to process large datasets. Subsequently, the study asked three related research sub-questions with special focus on making socioeconomic interests explicit. The first sub-question was: "Is it possible to identify competing groups from the available datasets?" To help answer it, spatial datasets of biological and socioeconomic activities for the study area were compiled and analyzed to identify competing interests, all of which were reported in Chapter 4. This data compilation exercise resulted in one of the most comprehensive marine use datasets for the study area. A second sub-question was: "Can the detailed data on fishing activities help assess competing socioeconomic interests?" The fishing dataset provided detailed records, allowing this study to examine potential competing interests within the fishing sector. Third, to derive meaningful spatially explicit information, this study asked the sub-question: "What are alternative methods for generating spatial socioeconomic criteria and biological data in marine spatial conservation planning?" This question became necessary because there are limited methods to process the variety of attributes needed for the study. Hence, several new methods were introduced and presented in Chapter 3, including methods for generating attributes for regional rare and endemic species, gear conflict, fishing gear impact, distribution of different groups of fishers, fishing business, and fishing employment.

Finally, in Chapter 4, this study developed the STF-OWA to help answer the research question: "How can tradeoffs among socioeconomic competing interests be integrated systematically and as explicitly as the biological competing interests?" The hierarchical arrangement of the 25 attributes shown in Figure 4.2 was the first step towards integrating structurally and quantitatively socioeconomic attributes with biological ones. Through the use of the STF-OWA method, this study investigated several additional questions. First: "Will the STF-OWA allow for making various levels of tradeoffs and HDs spatially explicit?" This study demonstrated seven conservation scenarios, five of which show five levels of tradeoffs and two scenarios that show lack of tradeoffs. From these seven scenarios, the use of the STF-OWA demonstrated how the relative toughness of conservation decisions can be made spatially explicit through three main decision-makers' attitudes, namely: accepting risk against biodiversity loss, accepting risks against socioeconomic loss, or the attitude that seeks to balance these risks.

The second question was: "Will the STF-OWA allow for constraining and comparing regions of interest (ROIs) using a conservation scenario?" In real-world planning, not all planning units (PUs) that may be identified in the solution set can be immediately implemented. This is important from the standpoint of prioritizing groups of adjacent PUs within a scenario. Since all PUs were scored between 0 and 1, groups of PUs or potential ROIs were queried and analyzed separately. This was made possible through the individual ID attached to each PU. Isolating these ROIs allowed comparison of their relative cost-effectiveness, and this can be used as a basis for prioritizing groups of PUs within the solution set.

The third question was: "Will the STF-OWA allow for visualizing stakeholders' priorities concerning competing interests?" The study showed that the STF-OWA helps to visualize stakeholders' priorities. As an example, we plotted and compared the scores for the six individual fishing attributes against the suitability and biodiversity gain scores for the four ROIs. Visualizing these three measurements for the ROIs simultaneously allowed this study to see how priorities, through universal weights, can be of importance concerning the suitability of PUs for conservation. For instance, a change in priority from fishing revenue to non-fishing revenues would also involve switching priority sites, due to the dissimilar spatial patterns between the monetary and non-monetary fishing attributes.

It should be noted that areas of the SW Grand Banks at depths between 800 and 2000 m were designated in 2007 through the Northwest Atlantic Fisheries Organization or NAFO's effort toward coral protection closure. Unfortunately, these depth limits do not afford protection to the rocky habitat that supports the gorgonian corals (Gilkinson & Edinger 2009). This thesis, by visualizing the six individual fishing attributes along with biodiversity gain, demonstrated evidence that the SW Grand Banks, shallower than 800 meters, is in fact high in biodiversity but poses significant adverse impacts to the fishing sector. Among the four conservation sites compared, SW Grand Banks has the highest number of fisher groups, a range of business interests, provides the highest employment, and is fairly accessible to different fisher groups, although it provides relatively low fishing revenues. The diversity of stakeholders and economic activities increases the risks of conflict in the area if it were to be closed for protection purposes.

Finally, this study answered the question: "Will the STF-OWA make the competing socioeconomic groups aware of impacts?" According to Adams et al. (2010), the success of MPAs is influenced by its integration of socioeconomic groups. Therefore, this study included an analysis of three potentially major competing socioeconomic groups, namely, marine transportation, oil and gas, and fishing. The STF-OWA shows that with a target of 20% or less of the area, the marine transportation and oil and gas sectors may not significantly compete with MPAs.

The fishing sector, however, due to its widespread presence in the study region, is likely to compete severely with an MPA target beyond 5% of the study area. Close examination was required to look at potentially competing groups within the fishing sector. Based on fishing revenue (landed values of fisheries) and FG-Richness (richness distribution of fisher groups), two competing groups stood out. In the NL bioregion, the attribute for fishing revenue shows areas relevant to the large-scale fishing industry, while the non-monetary fishing attributes show areas important to small- and medium-scale fisheries. These two attributes imply that the priority given to them by stakeholders will have a clear impact on who is going to lose or win. This finding agrees with an ethical discourse suggesting that most difficult decisions, such as conservation policy, tend to be value-laden (Nutt 2002, Schmid 2002, Kooiman & Jentoft 2005).

5.2 TRADEOFFS AS VIEWED IN THE STF-OWA DECISION-SUPPORT METHOD

The proposed STF-OWA decision-support method is based on the concept of tradeoffs where losses and gains are weighed. By "weighing", this thesis means viewing loss and gain as deviation from the current wealth, as previously defined by Kahneman and Tversky (1979).

Viewing tradeoff through loss and gain has two relevant meanings in PA planning. First, Kahneman and Tversky (1979) suggested that stakeholders are expected to react more strongly to a loss than a gain. Lackey (2006) confirms this in one of his policy axioms that "potential losers are usually more assertive and vocal than potential winners are." This axiom leads to the next relevant meaning: that loss should be carefully treated, and should be viewed with equal importance if not more importance than gain. Thus, the STF-OWA can view the importance of tradeoffs not only among biological interests but also among socioeconomic interests. By doing so, STF-OWA does not intend to provide procedures that could implement existing SCP concept such as efficiency, adequacy, and complementarity. Instead, STF-OWA explored the relative toughness of conservation decisions discussed in this thesis under the concept of hard decisions (HDs).

Figure 5.1 summarizes how tradeoff is situated in the STF-OWA method. There are at least three dimensions that interact with one another concerning MPA tradeoffs. These include multiple interests and interest groups, stakeholder value-laden priorities, and geographical space. In MPA planning, both multiple interests and groups of stakeholders are involved (Brown et al. 2001, Oracion et al. 2005, Adams et al. 2010, Gleason et al. 2010). Stakeholders include conservation groups and different socioeconomic groups whose interests lean toward exploitation of resources (e.g. industry such as oil and gas, fishers). Subsequently, these multiple interests and groups require a planning method that accommodates multi-dimensional criteria (Gillman et al. 2011, Thorpe et al. 2011), and simultaneously identifies different groups of stakeholders (Naidoo et al. 2006, Adams et al. 2010). The STF-OWA addresses these multiple interests by representing them with spatially explicit attributes based on the available datasets.

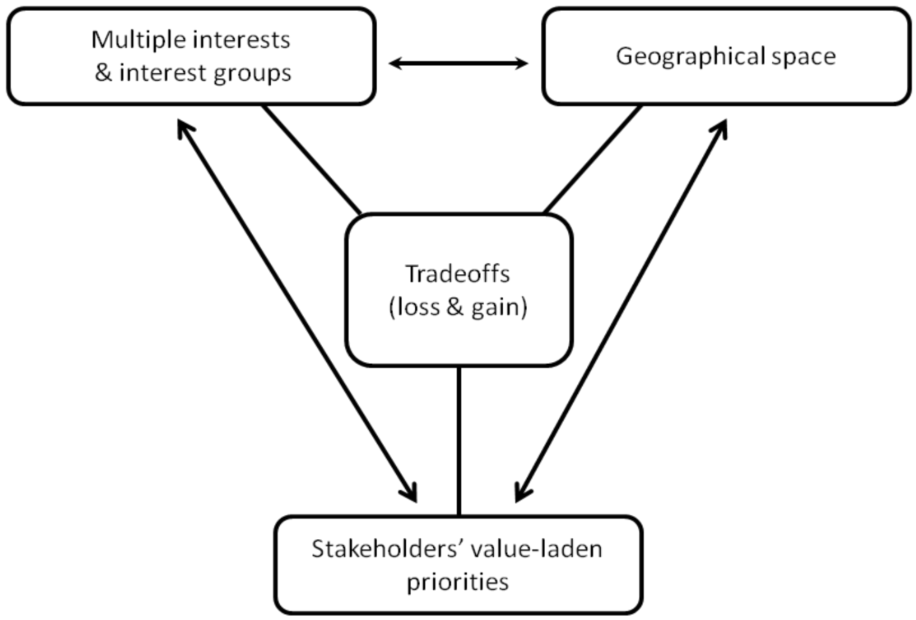


Figure 5.1 Multi-dimensional aspects of conservation tradeoffs as viewed in STF-OWA. The arrows show the relationships between dimensions.

When the multiple interests are not achievable simultaneously, prioritization is necessary, and agreements on priorities may not be possible without human value-laden priorities (Soulé 1985, Shafer 1987, Marris 2007, Wilhere 2007). Eliciting conservation priorities can be achieved through several means. This study used several methods such as direct weighting, analytic hierarchy process (AHP), consensus, and group discussions. The STF-OWA uses two types of weights, namely universal weights and ordered weights, to express conservation priorities. Universal weights are weights of importance that stakeholders can assign to each criterion. The ordered weights can identify various levels of tradeoffs among and between the weighted criteria in a spatially explicit manner. Finally, the multiple interests and various interest groups usually interact and compete in a similar geographical space, making PA planning inherently spatial (Norse 2005, Jones 2006, Bess & Rallapudi 2007, Crowder & Norse 2008, Halpern et al. 2010). The STF-OWA builds on spatial analysis using GIS -based data. Spatial analyses explored in this study include identifying the regions of interest (ROIs) and percent area targets in a solution scenario. Also analyzed spatially are the units of gain and loss as well as the cost-effectiveness of solution scenarios, percent area targets, and ROIs. These space-based analyses are some of the unexplored capabilities of combining OWA and the concept of goal hierarchy for marine conservation assessment.

Finally, it is important to note how the STF-OWA treats the interaction between the three dimensions. It provides a window of opportunity for stakeholders to discuss a set of priorities for any type of participative (Fisher & Ury 1981, Salm et al. 2000, Pomeroy et al. 2005) or multi-disciplinary type of decision-making or planning (Degnbol et al. 2006).

5.3 RESEARCH CONTRIBUTIONS AND HIGHLIGHTS

A major strength of the STF-OWA is that it accommodates alternative means of setting priorities, similar to the argumentative process (Rittel & Webber 1973), principled negotiation (Fisher & Ury 1981), and any form of open discussion (Hirsch et al. 2010) that includes context-dependent priorities and preferences. Embracing this type of process opens conservation planning to a less familiar territory of dealing with limits and HDs as a way to achieve conservation goals. With regards to this novel thinking in planning, the following section will highlight the major contributions of this research for spatial conservation planning.

- *Conservation planning methods need to be more diversified.*

The mainstream conservation planning research, over the last three decades, has largely focused on methods that are optimal, objective, and rule-based. These methods are powerful at identifying areas that achieve biological objectives in a solution set, but they do not pay similar attention to the diversity of socioeconomic objectives, resulting in rare implementations of those optimal sets of solutions. In their place, other methods that can better capture the diversity of socioeconomic objectives, such as expert-driven processes, are often used to help reach consensus amongst stakeholders. These, however, lack rigor to meet conservation objectives, and may not support the creation of effective networks of protected areas. Other approaches, such as the multi-criteria method, attempt to better balance biological with socioeconomic objectives, offering compromises between biological benefit and socioeconomic loss, and losses and gains between socioeconomic sectors.

Unfortunately, there is no single method that addresses all positive advantages associated with each of these methods. This problem is expected, as suggested in this thesis, due to the competing dimensions of protected areas (PAs), the "wickedness" of the PA problem, and the lack of ability of decision-makers and planners to identify all possible solutions to the PA problem (see Section 2.5 for related discussions). Consequently, recognizing the strengths and weaknesses of various approaches may provide a positive avenue to dealing with these limitations.

- *Integrating finer details about socioeconomic losses and biological gains in the decision-support method helps to make conservation tradeoffs and HDs more explicit.*

Conservation models based on biological/ecological optimization alone are effective tools to identify the best places to protect. With the integration of a single socioeconomic constraint to generate optimal areas, conservation models provide interesting tradeoff results important for planning purposes. Still, emerging research suggests that it would be more desirable to consider a variety of socioeconomic constraints, particularly when difficult questions, such as "conservation for whom?", are raised or when tradeoffs or HDs need to be identified among socioeconomic groups. This study shows that conservation tradeoffs and HDs can be made explicit when various types of socioeconomic losses are integrated in the decision-support method. The model shows that the amount of gain depends not only on the amount of loss but also the *type* of loss upon which everyone can agree, which is a more realistic picture of the planning process on the ground -- especially with increasing spatial conflict between conservation gains and losses.

- *Balancing losses and gains in favor of conservation is likely about making HDs rather than about making win-win decisions.*

Positive impacts of conservation planning decisions for *all* stakeholders are ideal. For this reason, conservation projects have promoted a win-win perspective for getting support from funders, communities, and other stakeholders. To date, however, real win-win situations are rare. This study provides empirical evidence that often minimal success can be expected, resulting from an increasing spatial conflict between conservation losses and gains. The study, however, also showed that some areas can achieve high conservation gain while keeping a minimum adverse impact on socioeconomic activities; but everyone may not win. Unlike the traditional win-win, the study shows that higher conservation gain is possible when HDs -- where certain interests will have to be sacrificed, possibly to a significant extent -- are consciously made. This finding is not trivial, as it reinforces the breadth of limits that can determine the achievability of goals (Calabresi 1991). Put another way, a tower builder would typically think of more than one limiting factor, such as available time, materials, expertise, technology, finances, and physical environment.

▪ *Tradeoff as a form of exchange alone can fail to make HDs spatially explicit in SCP models.*

Holland (2002, p 17) argues against the notion that "all choice is basically a form of exchange." When an exchange is made, it is expected that the best deal is chosen and there is hardly any ground for anguish or other deeply felt concerns. For this reason, Holland (2002, p 25) states that a tradeoff model "fails utterly to explain the toughness of tough decisions."

This study provides empirical evidence supporting Holland's argument. When exchange between objectives was not allowed, through the no-tradeoff rule, the STF-OWA scenarios demonstrated two types of tough conservation decisions: namely, tough decisions for and against biodiversity protection, and tough decisions for and against protecting socioeconomic activities. More importantly, when no-tradeoff scenarios were compared with tradeoff scenarios, the STF-OWA scenarios show the relative toughness of conservation decisions. As proposed in Chapter 2, making tradeoffs and HDs is important in conservation planning. It may require new questions and strategies. A decision-maker and planner who recognizes an HD is likely to build a cushion against potential shocks, but when he sees tradeoffs he might simply expect everyone to agree at some point. The latter can disregard the value-based dilemmas that often result from HDs (Nutt 2002, Schmid 2002).

5.4 LIMITATIONS

The STF-OWA, like any method, has strengths and limitations. The workshop participants found the method used by the OWA hard to understand. In particular, some found it difficult to comprehend the multiple levels of priority in the form of weights that the model requires. One reason for this is the lack of interaction of workshop participants with the tool itself. During the workshop, participants did not get a chance to explore different sets of priority weights that could have helped them understand how such weights can be translated into conservation scenarios.

The current design of the tool used in the study did not allow such an automated process. However doing that would make the STF-OWA much more useful in planning processes. Also, stakeholders' preferences are expected to be representative of all interest groups. In this study, while representatives of all interest groups were invited, representatives of the fishing industry were not able to attend the workshop. A different set of weights can be expected from a different group of stakeholders, resulting in a different spatial configuration of suitable conservation areas.

From a scientific perspective, one limitation associated with this type of method is the inherent subjectivity of the process. Objectivity and rigorously defined scientific principles such as complementarity and efficiency are hard to track and integrate with the method. Finally, accommodating stakeholders' priorities, in terms of social values, assumes that stakeholders are well informed about their preferences regarding the available options (Mitchell & Carson 1989, Theobald et al. 2009), or that they have developed their preferences based on certain choice heuristics rules (Tversky & Kahneman 1974). In the real world of planning, preferences are surrounded by inconsistencies and thus, identifying them presents dilemmas or potential conflicts (Rittel & Webber 1972, March 1994).

From a theoretical viewpoint, providing evidence that HDs exist requires embracing tougher questions in PA planning. For instance, challenging questions can arise when some groups of stakeholders perceive that they are required to sacrifice more than others. Such questions include: "who should win or lose?", "what criteria are useful in identifying potential winners and losers?", "how should losers and winners be engaged in the planning process?", or "is there a way to identify and regulate conservation benefits in order to compensate those who have lost more?" Considering these questions in modern planning will require broader – or perhaps new -- directions in terms of research and policy-making.

Planning without due consideration of the consequences of HDs can make the MPA more vulnerable to contestation and rejection; this could easily compromise the long-term goals of PAs (Fiske 1992, Singleton 2009). These issues raise the question, however, as to whether considering the consequences of HDs would necessarily lead to social acceptance of an MPA plan. While there may be no easy answer to this question, embracing the tough consequences of HDs would certainly mean taking into account difficult realities at the planning table. For this reason, on a philosophical basis, embracing HDs is not the solution *per se*. Perhaps it is reasonable to consider this notion as one of planners' attempts to understand complex realities, as opposed to the mainstream approach of simplification (Rittel & Webber 1973).

While the notion of HDs reflects complex realities, it might also require a fundamental principle in order to be effective. According to Connor (2002), when everyone cannot possibly win, the principle of co-existence or self-sacrifice is important, where stakeholders accept that they may not end up with a big portion of what they currently enjoy. Also, the concept of HDs may require fundamental values such as openness to agreement and mutual gain among stakeholders. Interestingly, these principles and values would have to be equally a limitation of the HD concept, as they entail a less straightforward planning method than quantitative operations.

5.5 FUTURE OPPORTUNITIES

Findings from testing the STF-OWA decision-support method show that this method offers a new perspective in investigating the role of tradeoffs and HDs in PA planning. This method was able to illustrate the nature of HDs and to show that quantifying them is possible to some extent. One of the immediate directions of the STF-OWA method is its application into a visualization and decision-support tool, a platform that would require significant development and was beyond the scope of this thesis. The conservation scenarios, quantitative analyses, and visualizations in this study could be made readily understandable if implemented into an interactive tool where stakeholders could explore different sets of conservation priorities. Offering a software tool could also help simplify the concepts that come with the method, a point raised by workshop participants.

The STF-OWA is not specifically designed to meet the CAR (comprehensiveness, adequacy, and representativeness) principles to which a systematic type of planning often operates. Future work can look into combining these principles along with the concept of hard decisions. For example, it is worth investigating the possibility of combining the strength of Marxan and STF-OWA into a single decision-support tool. Sensitivity analyses could also be conducted with

STF-OWA to explore how the approach behaves with different input data and weights. This technique could help stakeholders understand the impact of various sets of conservation priorities about the spatial configuration of conservation areas.

Another potential future of the STF-OWA is the addition of another layer of information where the seven scenarios can be implemented into a single scenario. Combining all seven sets of scenarios into one scenario is an interesting concept supported by the OWA method. Dubbed "spatial OWA," this concept was initially discussed in urban planning, and Makropoulos & Butler (2006) proposed that with a third index layer, it might be possible to combine the seven scenarios into one comprehensive scenario. However, there is currently no developed GIS-based OWA system and tool, nor any other form of technique that integrates this third index layer. Based on our proposed STF-OWA decision-support method, an integration of relational or conditional parameters by considering simultaneously the type, score, and weights of criteria in each planning unit could be another way to generate a single solution. This alternative approach is conceptually simple and can easily be facilitated in participatory-based planning. However, testing this approach would require developing a decision-support tool, and this study considered it outside the research goals. It can also be useful in determining thresholds of biological or socioeconomic scores that can be used as technical aids to facilitate a principled negotiation process as suggested by Fisher and Ury (1981).

This study used the best available data. It is important to note, however, that some datasets have inherent limits, and some of them could be difficult to address.

Nevertheless, they needed to be recognized and could be of interest for future research in understanding the impacts of these limitations that are beyond the scope of this study:: (1) for MSS, seabird and logbook, each data point represents only the start location of a survey trip, ignoring the heading and distance of data collection in which a sample is recorded;; (2) the accuracy of fishing vessel trip information depends on whether the information was recorded truthfully by fishers; (3) the quantitative representation of dynamic fishing activities cannot be represented fully in a static model; (4) not all relevant decision criteria were represented in the analyses, due to data unavailability at the time of data collation; and (5) some criteria are temporally or spatially more comprehensive than others. Hence, a confidence map, which provides a more systematic grade of consistency and quality of disparate datasets, is another layer that deserves attention in future testing of the method.

Finally, Chapter 3 discusses how each GIS-based map was generated so that readers can get an understanding of some of the caveats associated with each attribute. Often, addressing data limitations typically requires a significant amount of time, hence was considered to be outside the scope of this study. Some of these limitations include the following. First, criteria maps are calculated based on averages or sums of available data, which ignores temporal or seasonal variability. Second, while this study was able to show the relative densities of oil and gas activities in the study area, this spatial analysis is not sufficient to identify priority areas with respect to oil spill risks in the region. Third, it should be noted that the groundfish richness, biomass, and evenness did not include invertebrates, as the latter were mapped separately. Integrating the invertebrates into these measures should be explored in future mapping efforts. Fourth, while standardization of datasets allowed mathematical combination of attributes that were initially measured in different units, it requires further research and understanding.

5.6 RECOMMENDATIONS FOR MPA PLANNING

Transparency, a fundamental view of good governance, requires that choices are made explicit and available for negotiation. To make a policy decision transparent, this study recommends a planning process to provide equivalent effort in understanding the two sides of tradeoff (i.e. loss and gain). By making conservation losses and gains explicit, this study shows that it is possible to explore a range of policy decisions (i.e. easy to hard), offering alternatives to decision-makers and a better understanding of how changes in values can impact decision objectives. This type of information is imperative in a democratic setting, to make governing efforts transparent.

Monetary losses are important considerations in planning; however, taking into account non-monetary losses is also critical. Considering various sets of information within a fishing dataset, we can conclude that fishing is not all about monetary gain. In fact, this thesis shows that the financial aspect of fishing could be less important than the non-monetary dimension of fisheries. The fishing dataset, as demonstrated in this study, can provide information as to how non-monetary losses associated with fishing could affect small- to medium-scale fisheries and possibly the viability of small coastal communities bordering the central and southern region of the study area. In the NL bioregion, this is an interesting political perspective that needs further investigation.

Evidence-based decision-making typically requires data, and the STF-OWA method is expected to perform better with richer data. It is important to note that this study used Newfoundland and Labrador that has relatively rich data, however, our findings shows that the available data are still limited in understanding the ocean environment. Regarding the limitations associated with the availability of data and the inherent limits associated with available data, we recommend planners to discuss the limits and utility of existing datasets with stakeholders. This action would help stakeholders understand the limits of data and modeling techniques and hopefully help them come up with an agreeable and attainable set of data and information to inform their decisions.

In terms of processing and representing information, this thesis recommends the following. First, for future research, different sizes for planning units and tessellations could be explored. Second, this study looked primarily at the spatial component of biodiversity gains and socioeconomic costs. Adding the temporal dimension into this type of research is likely to provide an additional perspective to informing conservation decisions. For example, this study looked at the spatial component of the overlap between fix gears (i.e. gear conflict layer). It is possible that these gears may occur at the same place, but could occur at a different time. Third, we recommend investigating the different groupings of attributes as STF-OWA has the flexibility to do so. For example, this case study looked at the attributes for fishing and other marine uses as two separate groups. Other ways of clustering the attributes are also possible that may offer additional information. Fourth, the attributes presented here,, such as fishing revenue, are one of the potential calculations relevant to representing landed catch. Hence, this study recommends representing attributes in ways appropriate to the spatial planning goals and context. Finally, the results of the STF-OWA analysis for the case study show different regions of interests, potentially different, that can be further researched following the concept of representation.

Finally, from a research perspective, this study recommends examinations of how HDs can be approached in planning, particularly to understand their practical strengths and limitations. Applying the HD concept might also require use of new research questions such as those indicated in Section 5.4. It might also require broad understanding of the role of social justice and equitability in the MPA planning process. While this concept is discussed less often in conservation planning (Vatn 2002), it is embedded in the equitable management of resources promoted in the global biodiversity goal specified in Aichi Target 11.

5.7 LITERATURE CITED

Adams, V., B. Pressey, and R. Naidoo. 2010. Opportunity costs: Who really pays for conservation? Biological Conservation **143**:439–448.

Ban, N. and C. Klein. 2009. Spatial socioeconomic data as a cost in systematic marine conservation planning. Conservation Letters **2**:206-215.

Bess, R. and R. Rallapudi. 2007. Spatial conflicts in New Zealand fisheries: The rights of fishers and protection of the marine environment. Marine Policy **31**:719-729.

Brown, K., W. Adger, E. Tompkins, P. Bacon, D. Shim, and K. Young. 2001. Trade-off analysis for marine protected area management. Ecological Economics **37**:417–434.

Calabresi, G. 1991. The Pointlessness of pareto: Carrying Coase further. The Yale Law Journal **100**:1211-1237.

Crowder, L. and E. Norse. 2008. Essential ecological insights for marine ecosystem-based management and marine spatial planning. Marine Policy **32**:772-778.

Connor, M. 2002. Social cost and sustainability. Pages 181-202 in D. Bromley and J. Paavola, editors. Economics, ethics, and environmental policy: Contested choices. Blackwell Publishing, Oxford.

Degnbol, P., H. Gislason, S. Hanna, S. Jentoft, J. Nielsen, S. Sverdrup-Jensen and D. Clyde Wilson. 2006. Painting the floor with a hammer: Technical fixes in fisheries management. Marine Policy **30**:534-543.

Fisher, R. and R. Ury. 1981. Getting to Yes: Negotiating agreement without giving in. Houghton Mifflin Company, Boston.

Fiske, S. 1992. Sociocultural aspects of establishing marine protected areas. Ocean & Coastal Management **18**:25-46.

Gilman, E., D. Dunn, A. Read, K. Hyrenbach, and R. Warner. 2011. Designing criteria suites to identify discrete and networked sites of high value across manifestations of biodiversity. Biodiversity and Conservation **20**:3363-3383.

Gleason, M., S. McCreary, M. Miller-Henson, J. Ugoretz, E. Fox, M. Merrifield, W. McClintock, P. Serpa, and K. Hoffman. 2010. Science-based and stakeholder-driven marine protected area network planning: A successful case study from north central California. Ocean & Coastal Management **53**:52-68.

Gilkinson, K. and E. Edinger. 2009. The ecology of deep-sea corals of Newfoundland and Labrador waters: Biogeography, life history, biogeochemistry, and relation to fishes. Canadian Technical Report of Fisheries and Aquatic Sciences No. 2830, 136 pp.

Gurney, G., R. Pressey, C. Ban, J. Álvarez-Romero, S. Jupiter, and V. Adams. 2015. Efficient and equitable design of marine protected areas in Fiji through inclusion of stakeholder-specific objectives in conservation planning. *Conservation Biology*.

Halpern, B., S. Lester, and K. McLeod. 2010. Placing marine protected areas onto the ecosystem-based management seascape. Proceedings of the National Academy of Sciences **107**:18312-18317.

Hirsch, P., W. Adams, J. Brosius, A. Zia, N. Bariola and J. Dammert. 2011. Acknowledging conservation trade-offs and embracing complexity. Conservation Biology **25**:259-264.

Holland, A. 2002. Are Choices tradeoffs? Pages 17-34 in D. Bromley and J. Paavola, editors. Economics, ethics, and environmental policy: Contested choices. Blackwell Publishing, Oxford.

Jones, P. 2006. Collective action problems pose by no-take zones. Marine Policy **30**:143–156

Kahneman, D. and A. Tversky .1979. Prospect theory: An analysis of decision under risk. Econometrica: Journal of the Econometric Society **47**:263-291.

Kooiman, J. and S. Jentoft. 2005. Hard choices and values. Pages 292-293 in J. Kooiman, M. Bavinck, S. Jentoft and R. Pullin, editors. Fish for life: Interactive governance for fisheries. Amsterdam University Press, Amsterdam.

Lackey, R. 2006. Axioms of ecological policy. Fisheries **31**:286-290. [Reprinted Renewable Resources Journal **24**:13-16].

Makropoulos, C. and D. Butler .2006. Spatial ordered weighted averaging: Incorporating spatially variable attitude towards risk in spatial multi-criteria decision-making. Environmental Modelling & Software **21**:69-84.

March, J. 1994. A primer on decision-making: How decisions happen. The Free Press, New York.

Marris, E. 2007. Conservation priorities: What to let go. Nature **450**:152-155.

McShane, T., et al. 2011. Hard choices: Making trade-offs between biodiversity conservation and human well-being. Biological Conservation **144**:966-972.

Mitchell, R. and R. Carson. 1989. Using surveys to value public goods: The contingent valuation method. Resources for the future, Washington D.C.

Naidoo, R., A. Balmford, P. Ferraro, S. Polasky, T. Ricketts, and M. Rouget. 2006. Integrating economic costs into conservation planning. Trends in Ecology and Evolution **2**:681-687.

Norse, E. 2005. Ending the range wars on the last frontier: Zoning the sea. Pages 422-443 in E. Norse and L. Crowder, editors. Marine conservation biology: The science of maintaining the seas biodiversity. Island Press, Washington.

Nutt, P. 2002. Why decisions fail: Avoiding the blunders and traps that lead to debacles. Berrett-Koehler Publishers, Inc., San Francisco.

Oracion, E., M. Miller, and P. Christie. 2005. Marine protected areas for whom? Fisheries, tourism, and solidarity in Philippine community. Ocean & Coastal Management **48**:393-410.

Pomeroy, R., L. Watson, J. Parks, and G. Cid. 2005. How is your MPA doing? A methodology for evaluating the management effectiveness of marine protected areas. Ocean & Coastal Management **48**:485-502.

Rittel, H. and M. Webber. 1973. Dilemmas in a general theory of planning. Policy Sciences **4:**155-169.

Roberts, C., et al. 2002. Marine biodiversity hotspots and conservation priorities for tropical reefs. Science **295**:1280-1284.

Salm, R., J. Clark, and E. Siirila. 2000. Marine and Coastal Protected Areas: A guide for Planners and Managers. 3rd edition. IUCN, Gland Switzerland.

Shaffer, M. 1987. Minimum viable populations: Coping with uncertainty. Pages 69–86 in M. E. Soule, editor. Viable populations for conservation. Cambridge University Press, New York.

Schmid, A. 2002. All environmental policy instruments require a moral choice as to whose interests count. Pages 133-147 in D. Bromley and J. Paavola, editors. Economics, ethics, and environmental policy: Contested choices. Blackwell Publishing, Oxford.

Singleton, S. 2009. Native people and planning for marine protected areas: How "stakeholder" processes fail to address conflicts in complex, real-world environment. Coastal Management **37**:421-440.

Soulé, M.1985. What is conservation biology? BioScience **35**:727-734.

Theobald, D., N. Hobbs, T. Bearly, J. Zack, T. Shenk, W. Riebsame. 2000. Incorporating biological information in local land-use decision making: Designing a system for conservation planning. Landscape Ecology **15**: 35–45.

Thorpe, A., M. Bavinck, and S. Coulthard. 2011. Tracking the debate around marine protected areas: Key issues and the BEG framework. Environmental Management **47**:546-563.

Tversky, A. and D. Kahneman. 1974. Judgment under uncertainty: Heuristics and biases. Science **185**:1124-1131.

Vatn, A. 2002. Efficient or fair: Ethical paradoxes in environmental policy. Pages 148-163 in D. Bromley and J. Paavola, editors. Economics, ethics, and environmental policy: Contested choices. Blackwell Publishing, Oxford, UK.

Wilhere, G. 2007. The how-much-is-enough myth. Conservation Biology **22:** 514–517.

Appendices

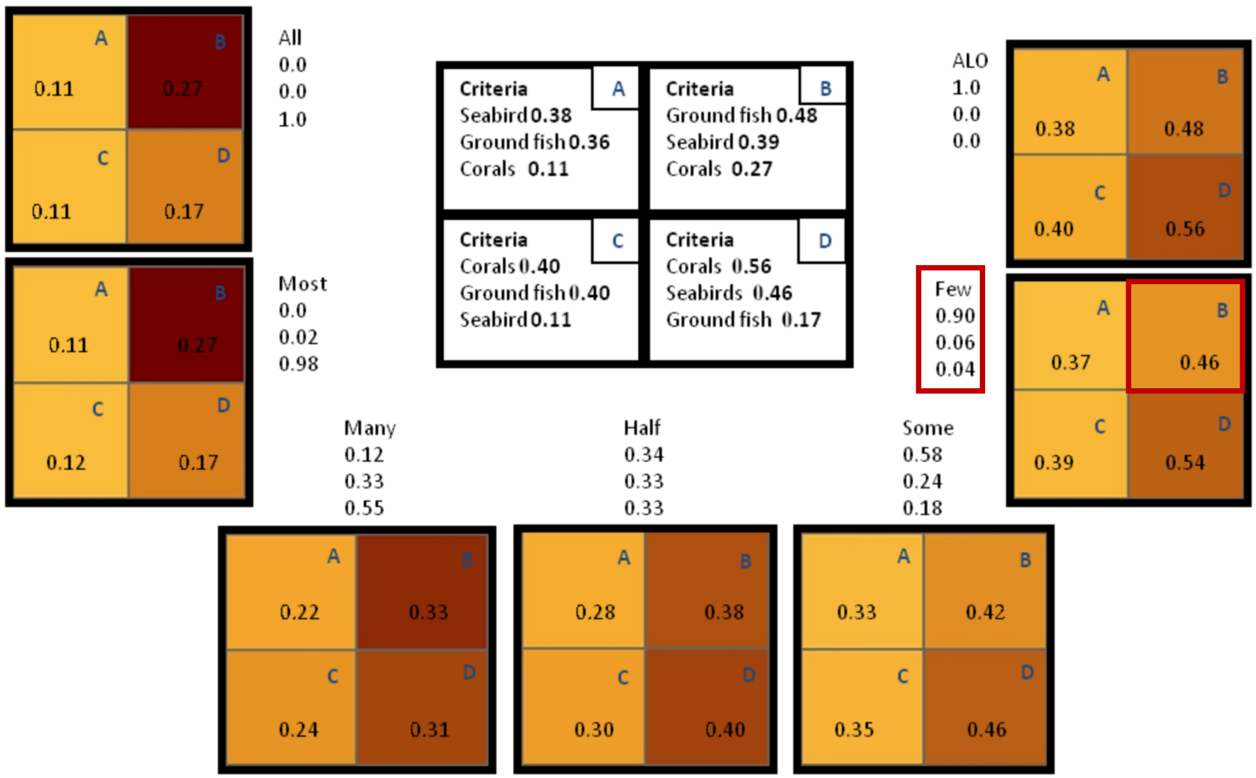
Appendix 1

OWA Calculation at PU level

How to calculate the OWA score for planning unit B, PU_B, of "Few" scenario?

(1) Obtain Z_{ij} by arranging scores of criteria j in PU_B from highest to lowest (i.e., 0.48, 0.39, 0.27)

(2) Multiply the Zij by a set of ordered weights (V_j) for "Few" scenario:
0.48 * 0.90 = 0.43
0.39 * 0.06 = 0.02
0.27 * 0.04 = 0.01
PU_B = 0.46



Notes: PUs are identified on the upper right corner, using letters A-D. Each PU contains a corresponding score for each of the three given criteria j (i.e. seabirds, groundfish, and corals). The ordered weight is given as a set of numbers for each scenario (e.g. "Few": 0.90, 0.06, 0.04). See Equation 4.4 and Equation 4.5 for calculating the ordered weights and additional details in Meng et al. 2011 p 53. The OWA score for each PU is indicated by a decimal number (e.g. 0.46 for PU_B of "Few" scenario).

Appendix 2

Categorization of Data

Three Categories of spatial criteria as obtained from Geographic Data

Criteria (Objective)	Criteria (Sub-objective)	Description
Biological Diversity	Seabirds, Groundfish, Corals, Sponges, Exploited invertebrates	Include measures of diversity (i.e. richness, evenness) and abundance (i.e. density or biomass) of the 5 marine taxa including groundfish rare species and species with different level of endangerment.
Adverse Impacts on Fishing Activities	Landed catch (amount and value), fishing business distribution, employment, diversity and equality of fishers within fisher groups	Fishing benefits denote foregone benefits such as: number of crew per boat (proxy for employment), distribution of different fisher groups in planning units (measured in terms of richness and evenness), distribution of fishing licenses (proxy for fishing business), aggregate landed catch (measured in monetary value, $ and weight, kg).
Impacts on other marine uses	Oil and gas, gear conflict, marine traffic, gear impact	Other marine uses represent other important activities: density of oil and gas activities, marine traffic, and gear impacts. Also, gear conflict based on effort overlap.

Appendix 3

Workshop Protocol

GIS-based STF-OWA approach to conservation planning using Newfoundland and Labrador data

November 29, 2012

Venue: SN 2000 at 9-2PM

I. Background and context

Expanding the coverage of marine protected areas continues to be a global goal in order to reach the 10% target recommended by the Convention on Biological Diversity (CBD) and the 20-30% target set by the 5th IUCN World Parks Congress. The Government of Canada has made an important commitment to identify and implement networks of marine protected areas by 2020. In response, Canada has been moving forward toward achieving its commitment by identifying ecologically and biologically significant areas (EBSAs) and by identifying and establishing MPAs. In the province of Newfoundland and Labrador, there are two MPAs that have already been identified and established. In addition, there have been past and recent efforts in the province to identify EBSAS which are expected to help in the regional planning process toward identifying network of marine protected areas (MPA).

While there are tools and approaches that are already available in order to identify marine protected areas, it is clear from current literature and practice that there is still a lot of room for exploring a regional planning approach. In particular, there is a need to develop participatory approaches that provide explicit tradeoffs. As conservation planning is a great challenge that requires concerted efforts and contributions from various interest groups, balancing these multiple interests plays a critical role in coming up with an agreeable conservation tradeoff decision. Hence, our study embarks on a participatory planning approach that uses multi-criteria analysis and geographic information system (GIS). This tandem of quantitative techniques and spatial tool provides an avenue to represent multiple interests of stakeholders and to carry out explicit tradeoff analysis. It is hoped that this study will offer additional perspective in conservation planning approach.

Multi-criteria decision analysis (MCDA) is a family of techniques that offers decision makers with several methods to evaluate areas of interest. MCDA has been used in fields such as urban planning (Joerin et al. 2009), land-based conservation (Strager & Rosenberg 2006), and environmental management (Linkov et al. 2006). So far, one of the common MCDA techniques used in conservation planning is weighted linear combination (WLC). It is based on a concept of weighted average where decision makers can allocate weights or "relative importance" to each decision criteria. There are three weaknesses associated with WLC, such as the following: (1) it does not offer real threshold that determine areas that should be included and excluded in decision-making; (2) contrary to the common expectation, WLC and other popular GIS overlay methods (e.g. Boolean, such as the *logical AND* and *logical OR*) do not yield similar results; and (3) WLC requires standardization of attributes that is typically based on simple linear transformation where up to this date, the rationality of this method lacks clarity. Hence, an OWA technique using fuzzy measures has been proposed in order to address these limitations (Jiang and Eastman 2000). It is explained that in using OWA, the continuity and uncertainty associated to the degree of membership of criteria in the decision set become explicit, which in turn provides strong logic for standardizing attributes (Jiang and Eastman 2000). Illustration is provided in the following section in order to explain the OWA concept using set problem.

II. Order weighted averaging (OWA)

OWA is relatively new and has not been applied in marine conservation and spatial planning. Theoretically OWA allows for explicit tradeoffs between various decision criteria, as well as offering decision-makers options to vary and control levels of tradeoffs. Hence, it can also generate alternatives based on WLC calculation. For these reasons, the utility of this technique in marine conservation planning, which accommodates data-driven approaches and incorporates stakeholder judgments, is worth exploring.

The following tables provide a simple comparison between OWA and WLC calculation, following three types of datasets, referred to here as "attributes." In a common MCDA technique, WLC is calculated by directly multiplying the value of each attribute (A) by the assigned weight (W) (Table 1). WLC total is the sum of A*W. Note that sum of W equals 1. In OWA, there are two types of weights that need to be determined such as the universal weight and the OWA set of weights, called **order weights**. In order to calculate OWA two things must be carried out. First, the attribute scores must be re-arranged from highest to lowest. In annex 1, this re-arrangement of attributes must be done in each block or planning unit (PU) of the study area. Second, seven sets of order weights need to be calculated. In Table 2, the order weights are already provided, in red text. OWA is calculated by getting the sum of the products of order weights and rearranged attribute scores. In Table 2, there are three sets of order weights provided such as "All," "Half, "and "At Least One or ALO." The **OWA** score varies with the given set of order weights. Table 2, explains the calculation. Annex 1 provides more detail as to how OWA can be calculated to determine the score of each

Table 1: WLC

Attribute (A)	Weight (W)	A*W
$X_1 = 0.8$	0.5	0.40
$X_{2,} = 0.4$	0.2	0.08
$X_3 = 0.7$	0.3	0.21
WLC total		**0.69**

Table 2: OWA

Attribute (A)	Weight (W)	A*	W*	Sets of ORDERED WEIGHTS		
				ALL	HALF	ALO
$X_1 = 0.8$	0.5	0.8	0.5	0.0	0.5	1.0
$X_{2,} = 0.4$	0.2	0.7	0.3	0.0	0.3	0.0
$X_3 = 0.7$	0.3	0.4	0.2	0.1	0.2	0.0
OWA				**0.4**	**0.69**	**0.8**

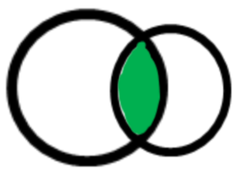
A* and W* are the re -arranged values of A and W, respectively, from highest to lowest.

OWAtotal = ∑ [Order weights*Rearranged A]. **Order weights** refer to the red numbers in the table

In Table 1, we can see that the WLC total is the same as the OWA value under sets of order weights called "Half." This illustrates that OWA can also generate calculation based on WLC calculation. In the following illustration, the OWA concept is explained like a set problem. It will be noted that the "set of order weights" in the following illustration and in table 2 is labeled with certain language quantifier, such as "All," "Half," and "ALO." These language quantifiers provide guidance in understanding the degree of membership of attributes in the decision set or planning unit score. For example, we can think of "All" as, "all criteria or attributes is met in the decision set or planning unit score", and for "at least one" as "at least one of the criteria or attributes is met."

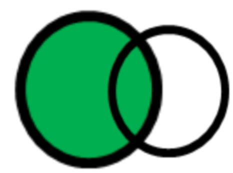
ILLUSTRATION

The following illustrates the OWA concept using two attributes. Each of the attributes is represented by circle. The area outlined in green represents the membership of each circle in the decision set. In this illustration, consider three set of order weights namely, "All," "Half," and "ALO."



Takes value common to both attributes. In this situation, the decision set takes the lowest value between the attributes. In this case, the decision maker is not willing to tradeoff the lowest score. Thus, it represents a very restrictive strategy where decision maker wants the minimum score in the decision set. Language quantifier: **ALL**

Take half of the values from each attribute. In this situation, the decision set takes values equally from each attributes. In this case, the decision maker wants a equal tradeoff between attributes. Thus, it represents a neutral strategy where decision maker is indifferent. Language quantifier: **HALF**

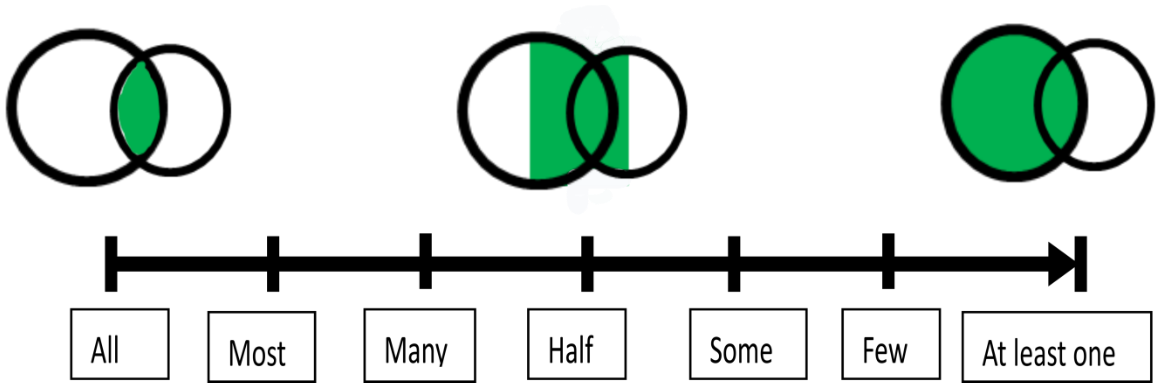


Union of two attributes. In this union, the decision set takes whatever is the highest value between the attributes. In this case, the decision maker is not willing to tradeoff the highest score. Thus, it represents an extremely unrestrictive strategy where decision maker wants the maximum score in the decision set.

Language quantifier: **ALO**

Note: The size of the circle represents the value (or score of an attribute). The tradeoff score is represented by green shade.

These sets of ordered weight can be identified in a continuum from "All" to "At least one," with "Half" in the middle point. For the purpose of this workshop, **seven sets** will be considered as per below. The language quantifiers, "All" to "Half," represent the continuum of extreme restrictiveness to neutrality while the linguistic terms, "Most" and "Many" represent very restrictive and restrictive strategy. On the other hand, the language quantifier "Half" to "ALO," represent the continuum of neutrality to extreme unrestrictiveness. Thus the linguistic terms, "Some" and "Few" represent unrestrictive and very unrestrictive strategy.



Essentially, OWA offers more options to vary levels of tradeoffs between criteria as datasets are aggregated into composite maps. In comparison to WLC, decision makers are given only one option in aggregating data and that is to do a full tradeoff between criteria. OWA on the other hand allows for other levels of tradeoffs such as "Most", "Many", "Some", and "Few". The workshop participants will explore the utility of this weighting flexibility of MCDA-OWA. For example, do we apply "Few," "Many" or "Some" in aggregating various attributes (density, richness, status and evenness) in order to generate a composite seabird map?

III. Workshop purpose

As part of the exploration of this MCDA-OWA system, we employ existing data from the eastern marine region of Newfoundland and Labrador as a test area (see Figure 2 below). The main objective of the workshop is to obtain your feedback about the applicability and the user-friendliness of MCDA-OWA in conservation planning, particularly, in making conservation tradeoffs explicit to the decision-maker.

IV. Testing of MCDA-OWA

In order to test the utility of MCDA-OWA, various sets of data have been compiled to represent three key decision criteria in conservation planning, i.e. biodiversity, socioeconomic benefits and other issues/uses. A total of 26 GIS information layers have been generated, with the following subtotals: 16, 7, and 3 for biological, socioeconomic and other uses/issues respectively. Each of these GIS information layer was calculated for each block (20x20 km) or planning units to subdivide the test area, Newfoundland and Labrador marine region. Biological diversity information was represented by diversity indices such as density using biomass or count, richness and evenness of several marine taxa. Socioeconomic information was represented by major human uses in the province. These include activities related to oil and gas and fishing industries that were acquired through Canada-Newfoundland and Labrador Offshore Petroleum Board (C-NLOPB) and DFO logbook, respectively. Oil and gas information layer was represented by several activities such as exploration, production, and significant discovery licenses. Fishing information was represented using the dollar value and biomass based on landed catch records. In addition, 21 fisher groups were identified by combining major fisheries and boat length. Based on these 21 fisher groups, social justice (number of fisher groups per planning unit) and social equity (equality or distribution of fishers among fishers groups per planning unit) were calculated. Please refer to Annex 2 for further description of the datasets generated for this test run.

As for other marine uses/issues, three information layers were generated such as commercial transportation (density of commercial vessel tracks), gear impacts (density based on severity ranking of fishing gear impacts on ecosystem) and gear conflicts (spatial overlap between fix and mobile gears). Table 3 shows the list of attributes for each of the three decision criteria discussed above while Figure 1 shows the structure on how data will be aggregated based on the proposed spatial tier framework.

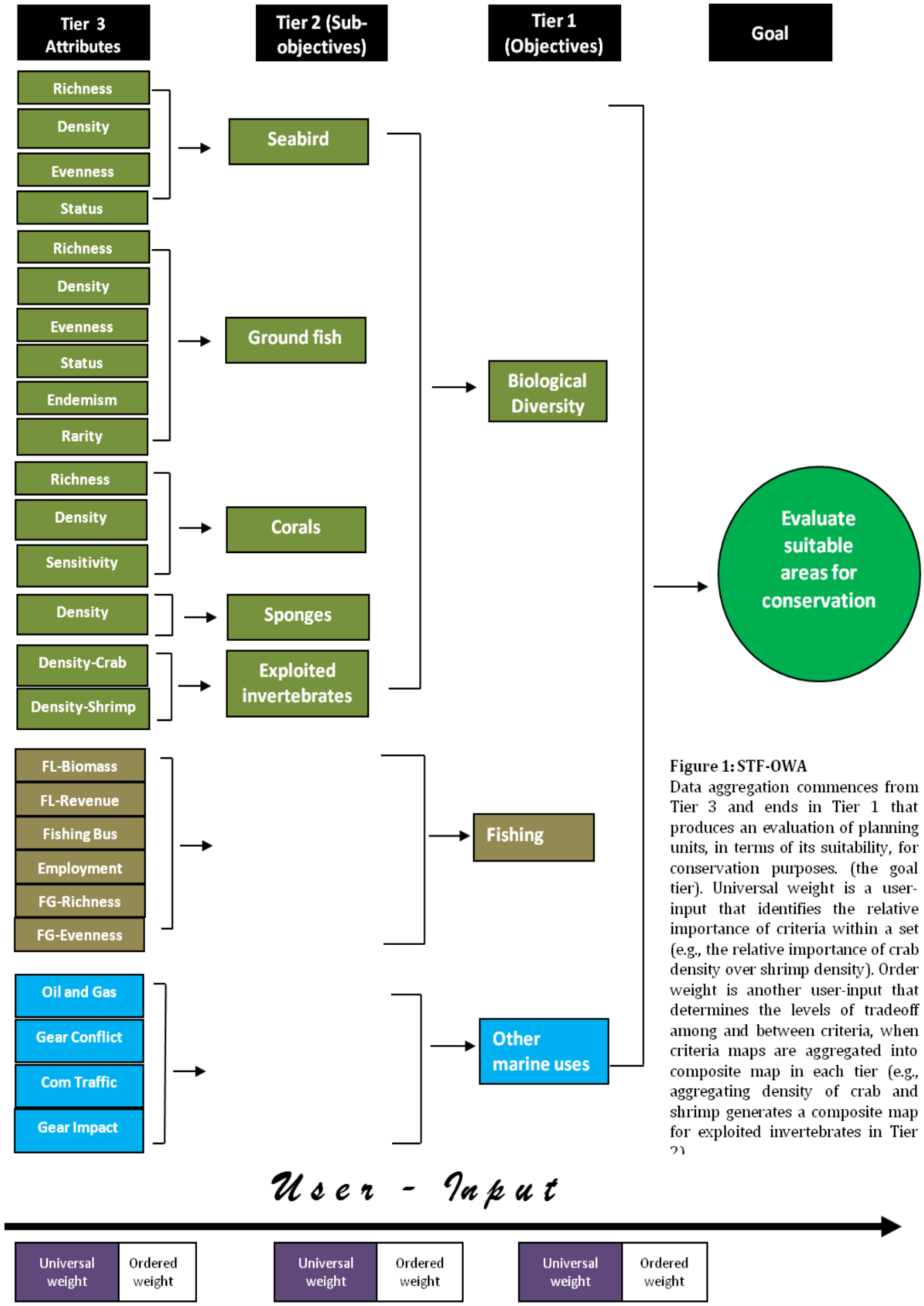


Figure 1: STF-OWA
Data aggregation commences from Tier 3 and ends in Tier 1 that produces an evaluation of planning units, in terms of its suitability, for conservation purposes. (the goal tier). Universal weight is a user-input that identifies the relative importance of criteria within a set (e.g., the relative importance of crab density over shrimp density). Order weight is another user-input that determines the levels of tradeoff among and between criteria, when criteria maps are aggregated into composite map in each tier (e.g., aggregating density of crab and shrimp generates a composite map for exploited invertebrates in Tier 2)

1. Spatial Tier Framework (STF), universal weight and order weights.

Figure 1 shows the STF, a structure where spatial data can be systematically organized, in a hierarchical manner, and MCDA-OWA can be implemented in a step-wise fashion. It is hope that with this framework, the tradeoff between competing objectives of conservation planning can be made more spatially explicit to the participants. STF has four columns referred to as attributes (tier 3), sub-objectives (tier 2), objectives (tier 1) and goal tier. First column is the list of **attributes** which can be thought of as "indicators" of the items in tier 1 (e.g. richness, density and evenness are indicators of seabird diversity). The second column refers to conservation planning sub-objective (e.g. seabirds, groundfish and coral diversity are specific conservation planning sub-objectives that can be considered in achieving biological diversity, a conservation planning objective indicated in the third column or tier 2. Lastly, is the fourth column or tier 3, which can be considered as the conservation planning goal (e.g. find the most suitable areas for management). This goal is achieved by considering the tradeoffs between the three criteria (or conservation objectives) in tier 3.

Universal weight is essentially the weights assigned to each set of attributes (in tier 1), sub-objectives (in tier 2) and objectives (in tier 3) in order to assess the level of importance of each attribute or criteria. The universal weights are assigned and can be based on value judgments, proportional ranking and ratios. On the other hand, order weights are not assigned to the attributes or criteria according to stakeholders' ranking but rather according to each attributes or criterion's position relative to each other in the planning unit. In short, universal weights apply to specific attribute or criteria while order weights apply to the ranked (re-arranged criteria score from highest to lowest) criteria after the application of the universal weights. In traditional WLC, universal weights determines how criteria tradeoff relative to each other. However, the level of tradeoff is not adjustable and full tradeoff is always assumed between all criteria. In contrast, OWA provides more leverage as to how criteria may compensate (or tradeoff) with one another as order weights are maximized, minimized, or allowed to tradeoff equally. In conservation planning, we are testing the utility of this weighting flexibility of OWA in making conservation tradeoffs more explicit. In the workshop, there are seven sets of order weights that will be considered as a way of aggregating spatial data and making conservation tradeoff decisions more explicit to the decision-maker. If you have time, please take a look at Annex1, it illustrates and explains the seven sets of order weights. In the workshop, it is important that participants understood what these seven order weights mean.

2. Order of Workshop Activities

The workshop will start with a short presentation. It will be followed by the facilitation of weighting process where the workshop facilitator will seek participants' inputs on universal weights and order weights of the decision criteria for conservation planning. In this section, we expect the participants to explore the potential utility of the weighting flexibility of OWA in making conservation tradeoff decisions. As mentioned in previous section, there will be seven scenarios (two scenarios that do not allow tradeoff and four scenarios that allow different levels of tradeoff) that the participant will explore in each tier. We would like to know if these varying levels of tradeoff options help conservation tradeoffs more explicit to the decision makers.

Finally after MDA-OWA exercise, the participants will be requested to discuss and evaluate the utility of MCDA-OWA in conservation planning. Figure 2 shows the snapshot of the workshop flow.

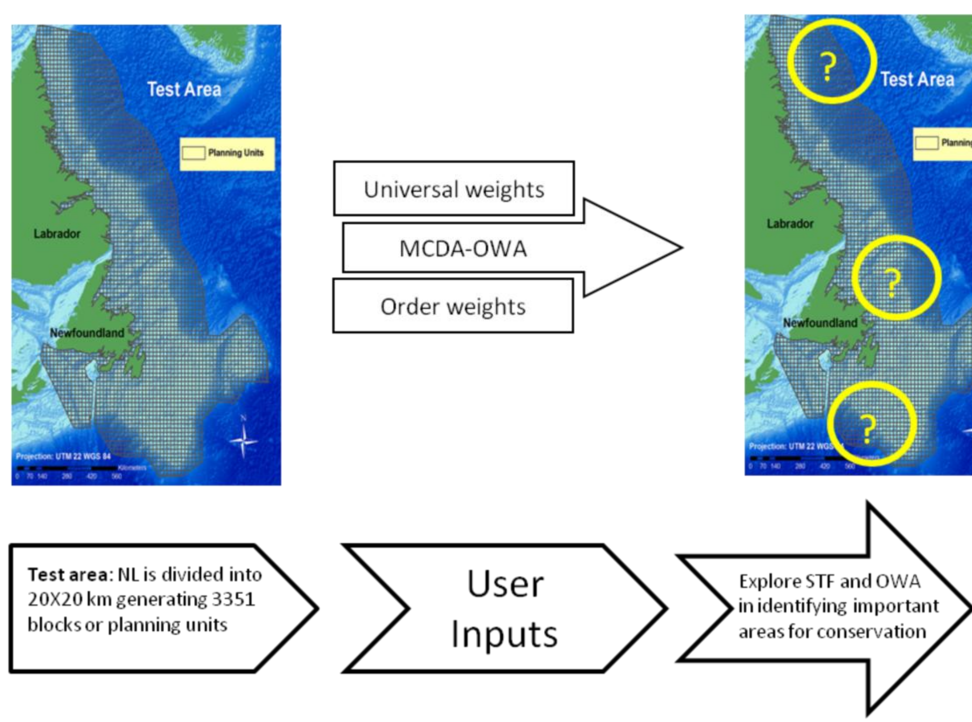


Figure 2 The workshop is divided into three sections where participants (1) get familiar with the data, (2) discuss conservation priorities and determine them using universal and order weights, and (3) provide feedback.

Table 3 List of attributes that will be used for the test.

Goal	Tier 1	Tier 2	Tier 3 (Attributes)
Suitable areas for conservation	**Biological diversity**		
		Seabirds	Richness
			Density using count/biomass
			Evenness
			Status
		Groundfish - all	Richness
			Density using count/biomass
			Evenness
			Rarity
			Endemism
			Status
		Corals	Richness
			Density using count / biomass
			Sensitivity
		Sponges	Density using biomass
		Exploited invertebrates	Shrimp
			Crab
	Fishing	Aggregated foregone benefits of fishing	Biomass
			Dollar value
		Fisher groups	Evenness of fisher groups
			Richness distribution of fisher groups
		Enterprise (or fishing business) and Employment	Individual Owners or licensee
			Number of Crew Members
		Oil and gas	Density of oil and gas activities
	Other marine uses	Gear Conflicts	Effort overlap between fix and mobile gears (including shrimp and crab gears)
		Gear impacts	Density of gear impact on ecosystem
		Transportation	Line density of commercial vessels

Appendix 4

Workshop Evaluation Questionnaire

1. Do you have previous/current experience or interest in conservation planning? If yes, please answer below, otherwise proceed to item 2.
   (a) What was/is your role/interest ____
   (b) Number of years of involvement ____
2. Have you had any experience with other tools or system in conservation planning? If yes, please compare it to MCDA-OWA in terms of weakness and strengths (use the back page).
3. What is your level of familiarity with Newfoundland and Labrador marine region? Choose from a scale of 1-5 (1, least familiar and 5, very familiar) ___.

Spatial Tier Framework [Yes/No]

4. Is the spatial tier framework useful in organizing spatial data? ___
5. Did it help you in examining your tradeoff decision? ___

Multi-criteria decision analysis using order weighted averaging [Yes/No]

6. Did the seven scenario maps help you decide in making tradeoff decision? ___
7. Is it necessary to have a scenario map other than the moderate scenario? ___
8. Is universal weighting useful in data and stakeholder-driven approach to planning? ___
9. Does it make a difference if we can choose one of the seven scenarios per planning unit? ___

Please choose from a scale of 1-5

10. Is the concept of tradeoff using MCDA-OWA easy or difficult to understand?

1	2	3	4	5
Easy				difficult

11. How do you find the user-friendliness of the whole MCDA-OWA exercises?

1	2	3	4	5
Not Friendly				Very Friendly

12. How do you find the utility of MCDA-OWA to conservation planning?

1	2	3	4	5
Not Useful				Very Useful

General Comment

____

____