CHARACTERIZING THE SOCIAL-ECOLOGICAL IMPORTANCE OF COASTAL MARINE LOCATIONS: INTEGRATIVE CHALLENGES, INSIGHTS AND SOLUTIONS FROM THE PACIFIC NORTH COAST OF BRITISH COLUMBIA

by

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Abstract

Human exploitation of earth's ecosystems has impacted the flow of ecological services, many with complex links to human health and well-being. The need to understand and plan for these impacts in an integrative manner is today an imperative. Yet, their integration into the planning process has been largely unsuccessful. In Canada, the Canadian Environmental Assessment Act (CEAA) was established to achieve this integration. Yet, despite decades of effort there has been limited progress in practice. Thus, the aim of this research was to contribute new knowledge and insights to the challenge of integrating a broad range of social and ecological concerns into the environmental planning and management process, focussing on pragmatic solutions.

A scoping review of the literature revealed key underlying issues affecting integration. These were discussed and contextualized to the CEAA mandated Environmental Assessment (EA) process, and a number of recommendations made for improved integration. The integration challenge was then examined within a spatial context. Two approaches to integrated spatial analyses were investigated. The first approach focussed on available marine spatial social, ecological, economic and protection legislation data; analyzing the data both singly to detect statistically significant clustering of high value or high incidence data (hotspots) and collectively to detect areas of agreement (overlaps). The analyses provided a perspective on the spatial distribution of marine social-ecological-economic hotspots. The integration was, however, challenged by the characteristics of the underlying data including differing approaches to data collection and units of measure. The second approach to integrated spatial analysis was based on expert spatial knowledge of the social-ecological system, and was termed expert informed geographic information systems (xGIS). Important social-ecological spaces were similarly detected using xGIS. It was found that xGIS allowed for a broader range of values to be considered, and the results were more readily integrated. The final considerations of this research addressed the question of application. Modeled environmental impacts (oil spills at sea) were established as a backdrop to integrated analysis. It was argued that a quantitative analysis of results could provide only limited pragmatic insights. A collaborative process of engagement among the actors to interpret results was proposed.

TABLE OF CONTENTS

List of Tables

List of Figures

List of Acronyms

Bbl – Barrels / Bbbl – Billion Barrels
BC – British Columbia
BCMCA – BC Marine Conservation Analysis
BCMEC – BC Marine Ecosystem Classification
BGCP – Biogeochemical Provinces of the Ocean
CBD – Convention on Biodiversity
CDN – Canadian
CEA – Canadian Environmental Assessment Agency
CEAA – Canadian Environmental Assessment Act
COP9C – Conference of the Parties
CV – Contingent Valuation
DFO – Fisheries and Oceans Canada
DWT – Deadweight Tonnage
EA – Environmental Assessment
EARP – Environmental Assessment Review Process
EBSA – Ecologically and Biologically Significant Area
EEZ – Exclusive Economic Zone
EIA – Environmental Impact Assessment
ENGP – Enbridge Northern Gateway Project
EPA – Environmental Protection Agency
ERA – Ecological Risk Assessment
FAO – UN Food and Agricultural Organization
GIS – Geographic Information Systems
GNOME – General NOAA Operational Modeling Environment
GOODS – Global Open Oceans and Deep Seabed Biogeographic Classification
HHRA – Human Health Risk Assessment
HIA – Health Impact Assessment
IA – Important Area
IAIA – International Association for Impact Assessment
IMP – U.S. Health Impact Project
LEK – Local Ecological Knowledge
LME – Large Marine Ecosystems
LOMA – Large Ocean Management Areas
MA – Millennium Ecosystem Assessment
MaPP – Marine Planning Partnership
MEOW – Marine Ecosystems of the World
MOE – Ministry of Environment
MOF – Ministry of Forests
NEB – National Energy Board
NEPA – National Environmental Policy Act
NOAA – National Oceanic and Atmospheric Administration
OSRA – US Oil Spill Risk Analysis
PPGIS – Public Participatory GIS
PFMA – Pacific Fisheries Management Area
PLA – Participatory Learning and Action
PNCIMA – Pacific North Coast Integrated Management Area
r – Pearson Product Moment Correlation Coefficient
SD – Standard Deviation
SIA – Social Impact Assessment
TEEB – The Economics of Ecosystems and Biodiversity
TEK – Traditional Ecological Knowledge
TUS – Traditional Use Study
UN – United Nations
UNDP – United Nations Development Programme
UNEP – United Nations Environment Programme
WHO – World Health Organization
WTA – Willing to Accept
WTP – Willing to Pay
xGIS – Expert-informed GIS

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"That one indeed is a [human being] who, today, dedicateth himself to the service of the entire human race... that ariseth to promote the best interests of the peoples and kindreds of the earth"

Bahá'u'lláh, 1817-1892

Chapter One. Introduction and Overview

In the two and a half centuries following the dawn of the industrial revolution, human activities such as fishing, grazing, logging, mining, cultivation and many others have exploited virtually every ecosystem on the planet (Millennium Ecosystem Assessment 2005) (MA). The resulting impact is affecting both the supply of the target resources, as well as a range of ecosystem services which provide important ecological, economic and socio-cultural benefits to people (MA 2005).

A growing body of evidence is helping to demonstrate the linkages that exist between human and ecosystem health (Parkes et al. 2010, Charron 2012), including a multi-country study by the World Bank (2007) which found that maintenance and access to ecosystem services was consistently associated with better health and economic outcomes. These linkages are also the central focus of the MA (2005) which characterizes ecosystem services as fundamental for human development and argues that human well-being (including freedom from preventable disease) is fundamentally dependent on the provisioning, cultural and regulating services of ecosystems. In light of this understanding, the MA (2005) puts forward the warning that human well-being is vitally dependent on improving our management of Earth's ecosystems. This may serve as a stark reminder of the efforts now needed to advance the practice of environmental management with the goal of balancing economic objectives with those of protecting human and ecosystem health (see UN 2015).

Today, many Canadian communities are facing growing pressures from both the private sector and government to support development opportunities on both land and marine-based ecosystems. The North Coast of British Columbia (BC) is one such example, currently being reviewed for over 20 major natural resource export and energy sector projects valued at over $200 billion (for an overview of projects currently under consideration see Carleton Ray and McCormick-Ray 2013, District of Kitimat 2015, Prince Rupert and Port Edward Economic Development Corporation 2015). In concert with the advancement of these projects, there appears to also be a parallel recognition among the 'actors' involved (i.e. the public, proponents and regulators) that a broad range of impact considerations, including health and well-being, are relevant to the discourse. Though not systematically documented, this recognition is implicit in the public relations documentation being distributed. The following examples help to illustrate:

> *"BG Canada believes LNG development...can help deliver economic and social prosperity for the people of Northern BC"* (BG Canada 2015);

> *"Our commitment towards contributing to the well-being of peoples and nations wherever we operate"* (Petronas 2015).

> *"No project will be approved unless it is safe for Canadians and safe for the environment"* -Joe Oliver, minister of natural resources (McCarthy et al. 2013).

It would appear, therefore, that the volition for better management is present. Yet in practice, in the Canadian context, the process is challenged as agencies of the Canadian government work largely in isolation, separately mandated by their respective legislations to manage specific components of the ecosystem, with arguably minimal intersectoral collaboration between them. See Table B-1 for a listing of Canadian federal Acts pertaining to the management of species groups and other resources of Canadian oceans and Table B-2 for a listing of the government agencies mandated to manage each Act and, by extension, each resource. The outcome of poor integration across these efforts is a failure to capture the necessary complexity of interactions and considerations; human health and well-being among those.

In Canada, the environmental assessment (EA) process, governed by the Canadian Environmental Assessment Act (CEAA 2012), has the primary mandate to balance the many considerations involved, including potential ecological, socio-economic, health and cultural impacts (Section 5 [s5] Government of Canada 2012). Given the breadth of factors included in the CEAA, the EA process may be viewed as the first significant opportunity for an integrated analysis of impacts.

The EA framework has, however, fallen short of achieving this mandate, tending to focus heavily on biophysical impacts and largely ignoring other considerations, especially those of human health (Steinemann 2000, Yap 2003, Health Canada 2004, Noble and Bronson 2005, Morgan 2011, Wright 2011). The net result is that the EA process, rather than serving as a mechanism of integration, is often the trigger for significant environmental conflicts, with many EA decisions ultimately challenged in the judicial arena (e.g. the proposed Enbridge Northern Gateway Project, see BC Nature 2014, Coates 2014, Laanela 2014a, Moore 2014).

1.1 Problem statement and research questions

The North Coast of BC is under consideration for a number of energy sector and natural resource export projects. The projects pose many potentially significant and complex economic, ecological, social and health impacts. These risks of impact have triggered opposition from municipal and First Nation's governments and the public. A number of processes have been proposed to manage and plan for these impacts in an integrative manner. Examples include the EA process, the Pacific North Coast Integrated Management Area (PNCIMA) initiative (PNCIMA 2011) (now cancelled), and the Marine Planning Partnership (MaPP) initiative (MaPP 2015). However, these processes are themselves challenged to achieve integrated analyses. The challenges involved range from limited understandings of the nature of impact (be they economic, ecological and/or social), the extent, severity and irreversibility of impact, the spatial distribution of impact, and the pragmatic challenge of integrating and applying analyses into an environmental planning and management framework.

Therefore, the overarching aim of this research was to contribute new knowledge and insights to the challenge of integrating a broad range of social and ecological concerns into the environmental planning and management process, focussing on pragmatic solutions that could be readily applied. The inherent complexity of this aim is acknowledged, yet it is also underscored by the urgent need for advancing our management of Earth's ecosystems as a means of vitally protecting human well-being (MA 2005).

To address this aim, the following research questions were posed:

1. What are the key issues preventing the integration of human health considerations into environmental frameworks such as the EA process? Are there opportunities within the current CEAA legislation to improve such integration?

2. What insights can an integrated analysis of existing marine spatial economic, ecological, social and protection legislation data offer with respect to the spatial distribution of important spaces in the ecosystem?

3. What insights can a scoping tool based on local expert knowledge offer with respect to the completeness and accuracy of spatially detecting and measuring important marine social-ecological spaces, and integrating those measures to describe the complete social-ecological importance of those locations?

4. How can the learning from and analyses of the questions above be applied in the context of a modelled scenario of environmental impact (i.e. a modelled oil spill at sea) and contribute to a discussion of integrated effects?

1.2 Defining scope: Selecting a suitable study area

Selecting an appropriate scale of analysis was an important initial consideration in the design of much of this research. Though large study areas may be desirable as a means of maximizing data capture, selection will often be limited by resource constraints and practical considerations. Another consideration to setting the boundaries of study is that of alignment with the focus of research. If the focus is purely ecological, then the biophysical extents of the ecosystems may be the appropriate boundary. If the focus is the social system, then lines of organization, demography, knowledge, economy, communications (including paths of movement), language and culture, and others (Parkes et al. 2010) may be sought to delineate a study region. In the context of this research, the focus was social and ecological. Thus, the selected study area should be both socially and ecologically relevant. Selecting an appropriate extent is important to the outcomes of research. Parkes et al. (2008) write:

> "Traditionally our understanding and management of human health has been organized spatially on the basis of human constructs such as municipalities, counties, health authorities, and provinces or states. While these boundaries do influence environmental and resource management, they often overlook and override the structure and function of ecosystems, and create a disjuncture between the objects of management and biophysical processes (p3)".

In the terrestrial context, Parkes et al. (2008) offer the watershed as an example of an appropriate unit of management, calling attention to its recurrent identification as the scale of focus for political and economic activity (e.g. MA 2005). Watersheds have also been identified and prioritized as appropriate spatial units around which to organize and manage natural resources and human health. As such, in certain applications, watersheds may be perceived and treated as social-ecological systems (Parkes et al. 2008).

The watershed concept is not, however, readily applicable to marine systems. Instead, marine systems are differentiated from one another based on factors such as physical, chemical, biological and socio-political characteristics. Various ocean classification systems are in use worldwide and are considered central to the establishment of good management practices. The question is which, if any, might serve as a useful unit of analysis for this research?

A number of common large ocean classification systems are examined in Appendix A, including the Biogeochemical Provinces of the Ocean (BGCP), biographic classifications of the CBD Conference of the Parties (COP9), Large Marine Ecosystems (LME), Large Ocean Management Areas (LOMAs), Global Open Oceans and Deep Seabed Biogeographic Classification (GOODS) and the Marine Ecosystems of the World (MEOW). Among these, the BGCP, COP9, LME and GOODS classifications are delineated based entirely on biophysical criteria. The MEOW classification is based on both biophysical and uncertain 'practical utility considerations'. The LOMA classification is, however, ecologically and socially integrated (i.e. based on integrated management plans including biophysical and socio-political considerations). The PNCIMA is one of Canada's 5 LOMAs and is frequently used for marine use planning purposes by federal, provincial and First Nations governments (e.g. the PNCIMA marine use planning process, PNCIMA, 2011). Thus, it is a familiar unit that is broadly recognized by the public and other sectors as a social-ecological system.

To address the issue of scale, the PNCIMA required further subdivision. A number of smaller subdivisions of the large ocean units were also examined in Appendix A, including simple grids, the British Columbia Marine Ecosystem Classification (BCMEC), Ecologically and Biologically Significant Areas (EBSAs), Pacific Fisheries Management Areas (PFMA), and the MaPP classification. Among these, the BCMEC, EBSA and disturbance/scope for growth sub-divisions are entirely biophysical in nature. The PFMA and grids are based on uncertain or no criteria. The MaPP delineation, however, is an integrated approach that is compatible with the PNCIMA and is delineated at a scale that is appropriate for localized planning.

Given the social-ecological nature of this research, the selected study area should be both socially and ecologically relevant. The PNCIMA was delineated with these criteria and was, therefore, selected as a suitable larger study unit. In instances when finer scale analyses were required, the MaPP subdivisions (i.e. the northeast portion of the PNCIMA in the case of this research) provided the best study area (an area of approximately 25,000 km^2).

As a measure of social-ecological relevance, the boundaries of the MaPP may be compared to a number of relevant social and ecological boundaries. For example, ecologically, the EBSAs (Figure 1-1c) provide a useful backdrop. Socially, lines of First Nations culture and language were considered an important consideration (see Figure 1-1a). Lines of travel, as evident in the vessel density map (Figure 1-1b) were also considered an indication of social and economic ties. An overlay of these data sources (Figure 1-1d) demonstrated the general validity of the MaPP boundary as a suitable social-ecological study area. For example, the MaPP boundary is relatively well-aligned with the

Smalgyax language/cultural boundary. It also encapsulates ecologically and biologically significant area (EBSA) units 10, 11, 14, 15 and the majority of 1 without interruption (see Figure A10 for a detailed map of the EBSAs), as well as, major north-south patterns of travel.

Appendix A (Figure A-12) is a representation of the PNCIMA and MaPP study areas in the context of the political boundaries and settlements of the region. The MaPP study area includes parts of the of Skeena-Queen Charlottes Regional District and the Kitimat-Stikine Regional District including the municipalities of Prince Rupert, Kitimat and Port Edward, the settlements of Oona River and Dodge Cove, and the First Nations communities of Kitimat Village (Haisla), Hartley Bay (Gitga'ata), Kitkatla (Gitxaala), Lax Kw'alaams and Metatakla (see Figure 1-1 and Figure A-12). The human population of the study area is estimated at approximately 25,000 (BC Stats 2014).

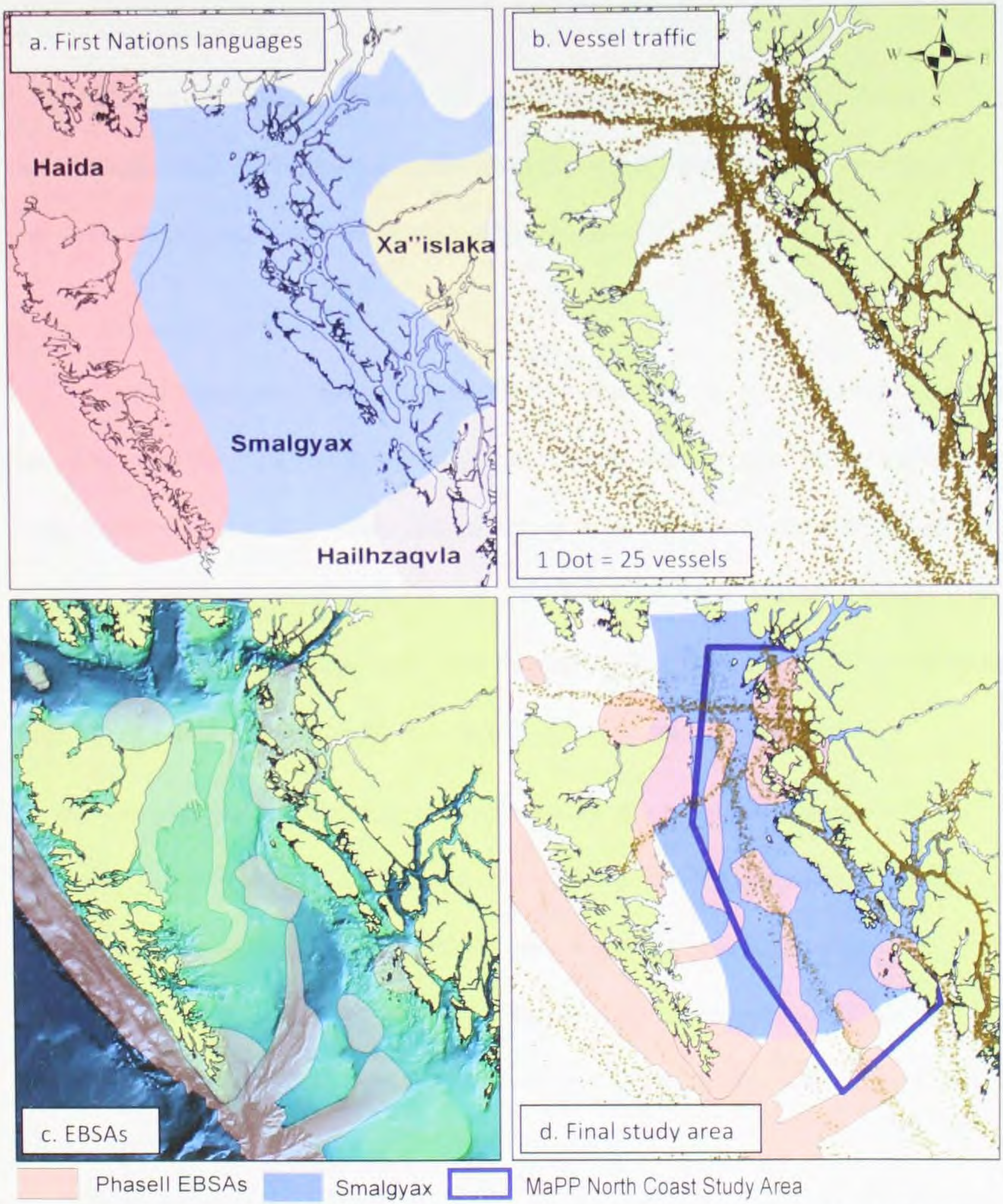


Figure 1-1. Spatial social and ecological datasets considered in the selection of a study area including, (a) Lines of First Nations language and culture, (b) Patterns of marine vessel traffic (2007 data), (c) Phase II Ecologically and Biologically Significant Areas (EBSA) with underlying bathymetric data, (d) the Marine Planning Partnership (MaPP) boundary referenced the three datasets and has been selected as a suitable boundary for the final study area in this research. See Appendix A for details of the social-ecological features found within this boundary.

1.3 Research design

Creswell (2009) describes a three step process to research design: (1) an acknowledgement of the research philosophy espoused; (2) defining the approach to research; and (3) selecting the methods of investigation.

1.3.1 Philosophical framing

The *research philosophy* espoused by an investigator is an important precursor to research design as it ultimately guides the approach to investigation. Others use terms such as *paradigms* (Lincoln and Guba 2000), *broadly conceived research methodologies* (Neuman 2000), *world views* (Creswell 2009), or *views of scholarship* (Boyer 1990). Creswell (2009) advises that individuals preparing for research should first make explicit the beliefs they espouse as a means of explaining their approach to the research problem.

The philosophy underpinning this research is grounded in the **post-positivist** perspective of producing quantified and repeatable results; the **pragmatic** perspective of moving theory to application; the **participatory** view that people (public knowledge) are a key asset in research; and the belief that **integration** across related sectors and disciplines is vital to advancement. These research philosophies are discussed below and contextualized to this research.

Post-positivist investigation, the successor to positivism which implied the unrealistic notion of absolute truth, is often referred to as the scientific method or empirical science. It strives to reduce problems to small discrete research questions that can be tested (reductionism) and seeks to find cause and effect relations (deterministic) through careful observation and measurement, resulting in the collection of quantitative data (Creswell 2009). Post-positivism aligns with a form of scholarship that Boyer (1990) terms the scholarship of discovery - a perspective ideally oriented to contribute to the stock of human knowledge. Discovery research is described as the acquisition of knowledge for its own sake, the freedom of inquiry to follow an investigation wherever it may lead (Boyer 1990). Much of the work conducted in the environmental field tends to align with the quantitative character of post-positivism and discovery research (see Chapter 3). In this research, quantitative analyses (i.e. quantifying the objects of investigation and conducting repeatable and statistically defensible analyses) were central to research questions 2 and 3 (see section 1.1).

Pragmatism involves the application of research for the purpose of seeking practical solutions. It aligns with what Boyer (1990) termed the *scholarship of application*; an approach to inquiry grounded in engagement and service, and the seeking of responsible applications of knowledge to the problems of individuals and institutions. Boyer (1990) contends that new intellectual understandings can arise out of the very act of application, where theory and practice vitally interact, and one renews the other. This view of scholarship is particularly needed in a world context defined by huge and almost intractable problems. It is in this arena where scholarship must "prove its worth not on its own terms, but by the service to the nation and the world" (Hanlin, in Boyer 1990). All of the stated objectives of this research are fundamentally pragmatic in nature; seeking to contribute to the many practical challenges currently faced in the field of practice. To this end, this research has aimed to develop conceptually and technically pragmatic solutions.

Participatory views of research recognize the role of public participation in the research process. This research is focused on issues of environmental impact to communities, with the public regarded as a central focus of impact. The relevance of people to the issues considered necessitates a certain degree of public participation in the research design.

The scholarship of integration, as described by Boyer (1990), was a key element of all four research questions above. The pursuit of integration led to a significant portion of this research to be dedicated to building connections across original disciplinary works in order to bring new meaning and perspectives to them. Discipline-specific works and data were collected and integrated as a means of bearing new insights on the research questions posed. In this regard, this research straddled the boundaries where the fields of ecology, health, economics and legislation converge. Boyer (1990) contends that as traditional disciplinary categories prove increasingly confining, integrative works are proving increasingly important in creating new topologies of knowledge that are responsive to new intellectual questions and to pressing human problems.

1.3.2 Methodology

Investigators must also decide on what Creswell and Plano Vlark (2007) refer to as an 'approach to inquiry', also referred to as a 'research methodology' (Mertens 1998, Rajasekar et al. 2013). Rajasekar et al. (2013) describe research methodology as the systematic approach that is selected to solve a problem. Methodological selection is influenced by the research philosophy of the investigator (e.g. post-positivist investigators will often choose quantitative approaches such as surveys and experimental designs).

As noted above, work conducted in the environmental field (e.g. in the EA process) is often characterized by a tendency towards quantitative analysis. This work is also challenged by complex social-ecological interactions which span a broad range of disciplines. The objectives of such work are largely pragmatic in nature (i.e. in need of practical management solutions) and tend to make efforts to be inclusive of the affected public at the center. These characteristics necessitate a methodology that is quantitative, integrative across disciplinary boundaries, pragmatic and inclusive of public participation.

According to Bammer (2005) research design focussed on integration and implementation is an effective way to tackle complex societal issues and problems. Pohl and Hadorn (2008) argue for the value of transdisciplinary collaboration among the social and scientific spheres, especially in the context of complex real-world problems. They contend that in such instances, transdisciplinary approaches can help foster links between knowledge groups, where each is transformed through its interaction with the specific problem during the research process, allowing for the development of new knowledge and practices.

Two methodologies with suitable characteristics were selected for this research; each with an associated body of literature. These included **ecohealth** and public participatory geographic information systems (**PPGIS**) (described in detail in Chapter 2 below).

- The **ecohealth approach** was deemed a valuable methodological framework within which the overall aim of this research could be housed (i.e. that of integrating a broader range of social-ecological considerations into the planning and management process). The positioning of human health and well-being as a central theme of inquiry, its systems-based and participatory character, and its systematic approach to inquiry, together with a body of relevant literature, provided the basis for addressing research questions 1 and 3.
- The focus of **PPGIS** on the collection, quantification and integration of spatial public knowledge as an approach to detecting important places in the ecosystem, together with its relatively well-developed methods to facilitate the application of its methodology, provided a useful framework to address research question 3.

The interaction between the methodologies of ecohealth and PPGIS resulted in the development of an expert informed geographic Information systems (xGIS) tool which is integrative, includes public participation, is both quantitative and spatial, and served as the basis for addressing research questions 3 and 4.

1.3.3 Methods

The methods of research are the procedures and algorithms used in research, the instruments used to collect data and the techniques to analyze and interpret them (Rajasekar et al. 2013). In this research each of the main chapters to follow employed specific methods. For example, a scoping review of the literature was conducted in Chapter 3, spatial statistics were applied in Chapters 4 and 5. These methods are described in detail in each chapter below.

1.3.4 Assumptions

Some core assumptions characterizing this research are as follows:

The natural world is a social-ecological system and is therefore aptly described by attributes such as complexity, relationships, change (both adaptive change and dynamic flux) and social phenomena (including cultural, political and economic). Research and study of social and ecological phenomenon in this view of the world are best suited to *systems approaches* which necessitate the employment of different skill sets and expertise and the incorporation of multiple stakeholders perspectives (Waltner-Toews 2011).

There are cross-linking relationships between human and ecosystem health. Despite the difficulties of measuring cause-effect relationships between ecosystem services and human health and well-being (Birley 2002, Noble and Bronson 2005, Briggs 2008, Braveman et al. 2011), the cross-linking relationship is assumed. This assumption is reinforced by a broad range of contextual works; some focused on measuring the direct benefits of ecosystems to human health, while others have demonstrated how adverse impacts to the former can be a root cause of adverse conditions in the latter (Gilles and Lebel 2001, Corvalan et al. 2005, MA 2005, Molnar et al. 2009).

People as social-ecological experts. People who live and interact extensively with their social-ecological environment tend to also form strong connections and relationships with the components of that system in what is commonly referred to as a 'sense of place' (Tuan 1974, Relph 1976, Tuan 1977). In doing so, they become 'experts' of their local environments (Brown et al. 2004), its ecological services (Brown and Reed 2011), the locations of biologically (Brown et al. 2004, Alessa et al. 2008), socio-culturally (Brown 2005) and economically (Brown and Pullar 2011) important locations, and the importance of such locations to their health and well-being. In this research, the term 'expert' is used to refer to individuals who, through personal and/or professional experience, have gained an extensive spatial perspective of the social-ecological system. These are a diverse group of individuals with respect to their roles in the community (from decision-makers to labourers), as well as their education, age, gender and economic classes.

1.4 Dissertation outline

This thesis comprises four separate manuscripts, each addressing one of the research questions posed in Section 1.1. These manuscripts have been published or are in review for publication in peer-reviewed journals.

Chapter 2

The objective of this chapter is to review some of the core theoretical concepts supporting the design of this study and the arguments presented in the chapters to follow.

Chapter 3

A scoping review of the literature was conducted to identify the most pressing issues pertaining to the application of health impact assessment (HIA) and the integration of health concerns into the Environmental Assessment (EA) process in Canada and internationally. The issues identified were also contextualized to the status quo practice of EA in Canada and the Canadian Environmental Assessment Act (CEAA 2012). Recommendations were proposed as a starting point for improved integration. The case was made for a new era of Canadian leadership and innovation at the interface of health and environmental assessment.

Chapter 4

The objective of this chapter was to develop an approach to integrate social, economic, ecological and legislated data in order to identify important marine spaces in the study area. Several categories of available economic (commercial fish harvest), ecological (ecologically and biologically significant areas), social (local ecological knowledge) and legislated marine protection legislation data were analyzed using spatial statistics in geographic information systems (GIS). An integrated analysis of the combined data produced a final map of important marine economic-ecological-social spaces. The research demonstrated that despite the challenges of integrated analysis, the proposed approach does produce a useful and comprehensive overview of important marine spaces for coastal and marine management processes.

Chapter 5

The objective of this chapter was to develop, apply and critically assess a tool to identify important social-ecological hotspots in the marine environment. The xGIS tool was applied within the study area to capture the knowledge of local experts from a range of backgrounds with respect to a series of 12 social-ecological value attributes, such as biodiversity, cultural and economic values. A series of spatial statistical analyses were performed to locate and quantify the relative social-ecological importance of marine spaces and the results were ultimately summarized in a single hotspot map of the entire study area. This study demonstrated the utility of xGIS as a useful tool for stakeholders and environmental managers engaged in the planning and management of marine resources at the local and regional levels.

Chapter 6

The objective of this chapter was to examine the findings of Chapters 3, 4 and 5 with respect to pragmatic application to a scenario involving environmental impact (i.e. modelled oil spills at sea). Three oil spills were modelled based on proposed oil tanker transportation routes using the General NOAA Operational Modeling Environment (GNOME™ 1.3.9); a standard spill–trajectory model supporting the NOAA standard for 'best guess' trajectories and 'minimum regrets'. The spills were spatially analyzed against the important social, ecological and economic areas identified in Chapters 4 and 5. Spatial overlaps were identified, serving as the premise of planning and management analyses. It was argued, however, that an integrated interpretation of the results within a collaborative context is required in order to determine social and ecological thresholds. An integrated process was proposed involving collaboration among expert focus groups in a staged approach to considering complexity.

Chapter Two. Background and literature review

2.1 Introduction

The overarching aim of this research was to build on scholarly learning across the disciplines in order to contribute new knowledge, insights and pragmatic solutions to the challenges of integrating ecological and social (including economics and health) considerations into the environmental planning and management process. The methodological frameworks of ecohealth and PPGIS were selected as the approach to investigation and a number of objectives were established to address the overarching aim including:

- determining the underlying issues related to the integration of health considerations into the environmental framework and to contextualize the issues to the Canadian EA process and associated legislation;
- developing an approach to integrating spatial social, ecological, economic and marine protection legislation data with the goal of better understanding the spatial distribution of ecologically, economically and socially valuable locations;
- developing and testing a spatial tool capable of integrating expert knowledge to create a better understanding of the spatial distribution of important social-ecological locations the study area.
- applying the learning above to a hypothetical scenario of impact (a modelled oil spill at sea) to facilitate a discussion related to the integrated effects of impact.

To explore this aim and objectives a number of core concepts need be defined and contextualized to this research. Thus, the objective of this chapter is to conduct a scholarly review of the literature to present a summary of the core concepts related to the themes discussed above.

2.2 Core concepts

The ecosystems of earth are facing increasing impacts, either by direct exploitation of their natural resources or through the indirect influence of fluxing global processes such as climate change, changes to global air and water quality and the like (see MA 2005). The impacts sustained by ecosystems are multi-faceted, involving numerous changes to biophysical processes and species compositions, measured by detailed and complex data and analyses. To make such information accessible and relevant to managers, policy-makers and the public, these ecosystem changes are often expressed in terms of changes to ecosystem goods and services (**ecosystem services**). Ecosystem service can be central to the health and well-being of people (see section 2.2.1) and changes to those services are often regarded as a measure of **ecosystem health** (see section 2.2.2).

Human dependence on ecosystem services is an important impetus to global and grassroots efforts to help improve ecosystem management. Both anecdotal information and evidence-based studies contribute compelling arguments for the **cross-linking relationship between human and ecosystem health** (section 2.2.4). Yet, measuring cause-effect relationships between them is a considerable challenge.

Underpinning this challenge is that of establishing a notion of what comprises human health. Various fields of scholarship focus on different understandings of health including those focussed on **physiological health**, **well-being** and the **determinants of health** (see section 2.2.3). Moreover, varying perspectives found within each of these fields creates additional confusion. For example, the determinants of health focus primarily on social concerns, with minimal reference to contributions from natural ecosystems. This has led to the development of additional frameworks in order to more wholly represent this relationship. One such framework emerges from the field of PPGIS and is focussed on **landscape value attributes** (section 2.2.5). Landscape value attributes have the benefit of ready measurement and allow inferences serving as a proxy for the environmental determinants of health (section 2.2.6). Landscape value data tend to rely heavily on local **expert knowledge** (section 2.2.7). This is in stark contrast to the more traditional **monetary approaches** (section 2.2.8) used to measure the values people place on ecosystem services.

These core concepts are applied, in the context of this research, to several common assessment processes including the **EA** and **HIA** (section 2.2.9). The approach used to investigate these assessment processes is a combination of elements from two methodologies: **ecohealth** and **PPGIS** (section 2.2.10). These concepts are systematically discussed in the sections to follow.

2.2.1 Ecosystem services

The concept of 'ecosystem services' was originally introduced by Westman (1977), Ehrlich and Ehrlich (1981) and others, as a means of soliciting action for nature-based conservation work (Gómez-Baggethun et al. 2010). The milestone work of Costanza et al. (1997), focusing on the monetary valuation of ecosystem services, was paramount to mainstreaming the concept of ecosystem services in both science and policy-making. More recently, the conceptual framework linking ecosystem services to human well-being, presented in the Millennium Ecosystem Assessment (MA 2005) - a four-year study sponsored by the United Nations involving more than 1300 scientists worldwide, brought further recognition and policy relevance to the concept.

There are innumerable ecosystem services that might be conceived. Various works have summarized those services in a range of categories. Table 2-1 presents an integrated list of the main categories and sub-categories of services described in the literature, including the (MA 2005), Daily et al. (2009), Costanza et al. (1997) and The Economics of Ecosystems and Biodiversity (TEEB) (de Groot et al. 2010).

Despite the range of services provided by ecosystems - serving as the basis for sustaining ALL forms of biota - the term 'ecosystem services' is most often regarded as the "benefits that people obtain from ecosystems" (MA 2005) and "the direct and indirect contributions of ecosystems to human wellbeing" (TEEB 2015). Some ecosystem services are directly used by humans (i.e. have consumptive use values) - for example, sea-life harvested for consumption. Others are used but not consumed (i.e. have non-consumptive use values) -example, the cultural values of ecosystems, the climate regulating services of ecosystems, etc. While others are not used but are still valued (i.e. have non-use values) -example, the values that humans place on ecosystems for their existence (see Hadley et al. 2011 for a description of use and non-use values).

Given the central focus of this research on human health and well-being, the most appropriate definitions of 'ecosystem services' were considered those provided by the MA (2005) and TEEB (2015), which place human health and well-being as a central consideration. Moreover, as described in Chapter 3, these definitions are also appropriate in the field of practice, where risks of adverse impacts to ecosystem spaces valued by people are central to public discontent and a range of natural resources management conflicts. Lastly, these definitions are also recognized in Canadian federal legislation (e.g. the CEAA 2012) which establishes the goal of sustainable development [s4.1h] setting the protection of human health [s5] as a central focus (discussed further in section 2.2.9.1 below).

Table 2-1. An integrated list of the main categories and sub-categories of ecosystem services described in the literature.

Main Category	Sub Category
Provisioning Services	Food (e.g. crops, livestock, fisheries, aquaculture, wild foods, etc)
	Water (for drinking and irrigation)
	Raw Materials (e.g. fiber, timber, fuel wood, fodder, fertilizer, etc)
	Genetic resources (for crop-improvement and medicinal purposes)
	Medicinal resources (e.g. biochemical products, medicines, pharmaceuticals, etc)
	Ornamental resources (for artisan work, decorative plants, etc)
Regulating Services	Air quality regulation (e.g. capturing dust, chemicals, etc)
	Climate regulation (e.g. C-sequestration, influence of vegetation on rainfall, etc)
	Moderation of extreme events (e.g. storm protection and flood prevention)
	Regulation of water flows (e.g. natural drainage, irrigation and drought prevention)
	Waste treatment and water purification
	Erosion prevention
	Maintenance of soil fertility (including soil formation)
	Pollination
	Biological control (e.g. seed dispersal, pest and disease control)
Cultural & Amenity Services	Aesthetic values
	Opportunities for recreation & tourism
	Inspiration for culture, art and design
	Spiritual experience
	Information for cognitive development
Indirect (Habitat) Services	Maintenance of life cycles of migratory species (incl. nursery service)
	Maintenance of genetic diversity (especially in gene pool protection)

Based on MA (2005), Daily et al. (2009), Costanza et al. (1997) and de Groot et al. (2010). Adapted from de Groot et al. (2010).

2.2.2 Ecosystem health

Defining 'ecosystem health' is a key underpinning to how ecosystems are managed, and the notion of ecosystem health has been increasingly regarded as the means of clarifying, evaluating and implementing ecological policy in the 21^{st} century (Lackey 2003). However, owing to the inherent complexity of ecosystems, there is no broadly accepted operational definition of ecosystem health (Rapport et al. 1998, Xu and Tao 2000). Instead, a range of definitions are inferred by way of the characteristics attributed to healthy ecosystems. For example, healthy ecosystems are considered those that are 'well-functioning' (Belsky 1995) in terms of their ability to self-organize (Haskell et al. 1992, Rapport et al. 1998, Costanza 2012), display vigor, resilience (Haskell et al. 1992, Rapport et al. 1998, Costanza 2012) and resistance (TEEB 2015), can achieve homeostasis and maintain stability (Karr 1986, Page 1992, Ulanowicz 1992) within normal ranges of biodiversity (TEEB 2015) and stages of succession and climax (Ulanowicz 1992), with the internal characteristics of relatedness, hierarchy, creativity and fragility (Norton 1992).

Rapport et al. (2001) and Costanza (2012) contend that ecosystem health should be defined by an integration of both biological factors, as described above (i.e. objective concepts), as well as on the capacity of the ecosystem to achieve reasonable and sustainable human (normative) goals - for example, the role of ecosystem services for provisions of food, fiber, potable water, clean air and waste assimilating/recycling. Integrating these two notions of ecosystem health results in the development of a broad conceptual framework characterized by the provisioning capabilities of ecosystems to sustain not only biological functions but also human communities, including economic opportunities and human health (Rapport et al. 1998).

Given the social-ecological underpinning of this research, the integrated concept of ecosystem health described above was deemed the most appropriate. It is noteworthy, however, that the embedding of social values into the definition is criticized. For example, Lackey (2003) questions the fundamental process of determining which societal preferences should take precedence (i.e. resolving competing individual and societal preferences).

Notwithstanding the challenges entailed in the precise definitions of the two concepts reviewed (i.e. 'ecosystem services' and 'ecosystem health') a common ground between them is their inference of a cross-linking relationship between environmental processes and human health. Yet, the notion of human health and what constitutes human health is itself unresolved.

2.2.3 What is human health?

Over the last half century, a significant shift in thinking has occurred with respect to the notion of what constitutes *human health* (Frankish et al. 1996). Early concepts focused on the 'absence of diseases' or dysfunctions as a conceptual definition of health, regarding health from a mechanistic perspective with a biomedical emphasis (Bourne 2010). The biomedical model failed, however, to account for the socio-physical, cultural and psychological factors that can significantly influence health long before the onset of any dysfunctions or ailments (Bourne 2010).

It was not until the mid-20th century that the culturalized tradition of the supremacy of the biomedical model began to be seriously challenged (Bourne 2010). An early far-reaching effort to broaden the definition of health came in 1946 when the World Health Organization (WHO) described human health as "a state of complete physical, mental and social well-being" (WHO 1992). Though more encompassing, the definition was challenged by the impracticality of achieving 'complete' physical, mental and social well-being outcomes. Furthermore, the definition tied physiological health together with the notion of social and mental well-being, thus making it exceedingly difficult to measure health (Frankish et al. 1996). Crisp (2005) refers to the definition as an 'elusive dream' that is 'difficult to operationalize'.

In the early 1970s empirical data began to provide statistically rigorous analyses showing that the health status of people was, in fact, influenced by both biological factors and a plethora of other social conditions (Grossman 1972). Another pronounced *shift* in thinking originated two years later from the works of Hubert Laframboise who prepared the widely circulated Lalonde Report - the White Paper (1974) - followed by the related works of Thomas McKeown (Glouberman and Millar 2003). These works essentially reinforced the notion that health could be affected by changes in both lifestyle and the social and physical environments.

These findings lent new credence and impetus to the general framework of health proposed by the WHO, expedited the evolution of the broadening definition of health and became the foundation for much of present-day research on health and well-being (Bourne 2010). This reshaping led to a notion of health that is associated with the idea that healthy individuals will have the capacity to adapt to, respond to, or control life's challenges and changes (Frankish et al. 1996).

Concurrent with the evolving and expanding definition of health put forward by the biomedical community, discussed above, was a concept of health put forward by the social and economic development community referred to as 'well-being'. Well-being is described as the outcome achieved when individuals have their basic material needs met, experience freedom and choice, and enjoy health, personal security and good social relations. Well-being is placed at the opposite end of a continuum from poverty, which has been defined as a 'pronounced deprivation in well-being' (MA 2005).

According to these broader notions of health and well-being, biophysical factors, such as the rates of mortality, disease and injury are considered only one aspect of health. Yet, despite the narrow definition, the biomedical model is still today the dominant model in many jurisdictions worldwide (Bourne 2010) and human health is often measured and described based only on the biophysical health status of an individual or population, rather than the broader definitions of health available.

Further to the broadening notions of health and well-being discussed above, the works of Laframboise may have also been the precursors to a number of other advancements. These included works such as The Ottawa Charter for Health Promotion (WHO 1986) and others (e.g. Epp 1986) which described the notion of a range of factors which collectively determine the health of individuals and communities, coined by Thomas McKeown as the 'determinants of health' (Glouberman and Millar 2003). The determinants of health take into consideration the social, economic and physical environments in which people live, as well as the personal characteristics and behavior of individuals. Today, there are various typologies of determinants presented in the literature. Some of the more common are summarized in Table 2-2. The utility of the determinants of health to this research are further discussed in section 2.2.4 below.

In summary, broader and more holistic notions of health have evolved through a small sub-component of the biomedical community and a large portion of the social and economic development community. These broader notions (i.e. well-being and the determinants of health) have gradually converged onto a concept of health which entails both physiological health and social well-being. Though the great majority of the biomedical world continues to work with the narrow definition of health, other fields of scholarship are largely engaged in research and applications of a notion of health which encompasses the breadth of factors that comprise human health and well-being. It is this latter notion that is considered in this research.

Table 2-2. A summary of the determinants of health as identified in various works in the literature.

Determinants of Health	HC	MR	PH	FM	WHO	%
Education	X	X	X	X	X	100
Income	X	X	X	X	X	100
Social support networks	X	X	X	X	X	100
Health services	X	X	X	X	X	100
Social status / social exclusion	X	X		X	X	80
Early life	X	X	X	X		80
Employment & working conditions	X	X	X	X		80
Gender		X	X	X	X	80
Physical environments	X		X	X	X	80
Personal health practices and coping skills	X		X	X		60
Race / Culture		X	X			40
Biology and genetic endowment	X		X		X	40
Social environments			X	X		40
Aboriginal status		X				20
Disability		X				20
Food insecurity		X				20
Housing		X				20
Income distribution		X				20
Unemployment and job security		X				20

HC - Health Canada (2004); MR - Mikkonen and Raphael (2010); PH - Public Health Agency of Canada (2012); FM - Association of Faculties of Medicine of Canada (n.d.); WHO - WHO (2015a)

2.2.4 The cross-linking relationship between human and ecosystem health

The literature provides strong support that ecosystem services are fundamental to human health. Reviews conducted by Sandifer et al. (2015) and Keniger et al. (2013) revealed over 200 studies demonstrating the psychological, cognitive, physiological, social, esthetic, cultural, recreational, spiritual, disease exposure regulating, material providing, and resiliency increasing contributions of ecosystem services to human health and well-being. The studies referenced vary in rigor and the results are often correlative rather than cause-effect in nature. Thus, despite detecting certain cross-linking relationships between human and ecosystem health, many studies are challenged to show causality. This is not surprising given that causal pathways between ecosystem services and downstream health outcomes are typically long and complex, often involving multiple intervening and potentially interacting factors along the way (Birley 2002, Braveman et al. 2011). As a result of this complexity, Hough (2014) argues that it is unlikely that casual relationships between human health outcomes and ecosystem services can be demonstrated.

A more pragmatic approach may be to examine relationships between ecosystem services and the determinants of health. For example, the provisioning services of fish harvest from marine ecosystems can be readily linked to certain determinants of health (e.g. fish as physical nutrition, income, employment and in some cases, access to culture). They are not, however, easily linked to specific physiological health outcomes.

Despite the pragmatics, causal pathways between ecosystem services and the determinants of health are also challenging. One, individual determinants do not act in isolation (Public Health Agency of Canada 2012). Instead they are often associated with one another, both in individuals and in communities. For example, poor education is associated with lower income. Thus, it is difficult to measure the determinants individually. Two, the challenges are accentuated by a range of feedback loops occurring between the determinants. For example, lower income may lead to poor nutrition, poor nutrition to more illness and time off work, time off work leads to yet lower income (Association of Faculties of Medicine of Canada n.d.). These uncertainties make accurate causal relationships between changes to ecosystem services and the determinants of health difficult to demonstrate.

It is noteworthy that despite the challenges, there are compelling reasons for on-going research to develop better approaches to understanding the relationships between changes to ecosystem services and the resulting changes to the determinants of health. This is often the central focus in communities facing potential environmental impacts and asking how projects will impact the ecosystem and its services, and how those impacts will affect the determinants of health. The landscape value attribute concept, applied through a PPGIS methodological framework, may offer new insights to this question.

2.2.5 Landscape value attributes

The early works of Tuan (1974), (Tuan 1977) and Relph (1976) recognized important relationships between people and their natural environments, describing people as active participants in the landscape - thinking, feeling, acting and receiving information from both observation and experience. They found that people could gain 'perception' and thereby attribute 'meaning' to landscapes, ultimately developing what was termed a 'sense of place'. Brown and Weber (2011) contend that "people are place-makers - we differentiate place from space by attaching meaning and values to space."

Williams and Vaske (2003) describe two types of attachments that people share with places: *place dependence* and *place identity.* Place dependence (a functional attachment) reflects the importance of a place in providing features and conditions that support specific goals or desired activities. Such attachments are based specifically on activities that take place in a setting and may range from being very personal to very public (e.g. a heritage site in a national park). Place identity (an emotional attachment) is described by Williams and Vaske (2003) as the "symbolic importance of a place as a repository for emotions and relationships that give meaning and purpose to life". Place Identity has been described as a component of self-identity (Proshansky et al. 1983), that enhances self-esteem (Korpela 1989), increases feelings of belonging to one's community (Tuan 1980), and is an important component of communications about environmental values and policies (Cantrill 1998). Place identity generally involves a psychological investment with a place that tends to develop over time (Giuliani and Feldman 1993). When place identity attachments are strong, resource management conflicts tend to intensify as different segments of society assign different kinds and degrees of meaning to the same places (Williams and Vaske 2003).

Zube (1987) further builds on this phenomenon and describes the human-landscape relationship model; proposing that individuals who develop such place attachments are often capable of associating a quantifiable range of values to places (Brown 2005). Many studies (see Landscape Values PPGIS Institute 2015a) have proposed typologies of landscape value attributes to reflect the human-landscape relationship (see Table 2-3). Brown (2012a) described landscape value attributes as spatially referenced layers of human perceptions. Brown and Reed (2011) asserted that the "human process of valuing landscapes results in structural and distributional patterns on the landscape that, although not directly observable, constitute latent patterns of social and psychological complexity that can ultimately be measured and quantified".

Table 2-3. Landscape value attributes identified in various studies found in the literature.

Value Attribute	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**12**	**13**	**14**	**%**
Scenic / Aesthetic	X	X	X	X	X	X	X	X	X	X	X	X	X	X	**100**
Recreation	X	X	X	X	X	X	X	X	X	X	X	2	X	X	**100**
Scientific / Learning	X	X	X	X	X	X	X	X	X	X	X	X	X	X	**100**
Spiritual	X	X	X	X	X	X	X	X	X	X	X	X	X	X	**100**
Economic	X	X	X	X	X	X	X	X	X		X	X	X	X	**93**
Cultural / Heritage		X	X	X	X	X	X	X	X	X	X	X	X	X	**93**
Life Sustaining	X		X	X	X	X	X	X	X	9	X	X	X	X	**93**
Therapeutic		X	X	X	X	X	X	X	X		X	X	X	X	**86**
Biodiversity	X		X	X	X	X	X	X	X		X	X	X	X	**86**
Existence / Intrinsic	X		X	X	X	X	X	X	X		X	X	X	X	**86**
Wilderness	X				X	X	X	X	X		X	X	X	X	**71**
Future			X	X	X	X	X	X			X		X		**57**
Historic	X	X	X	X				X	X			X	X		**57**
Subsistence			X	X				X		4			X		**36**
Ecological		X													**7**
Moral / Ethical		X													**7**
Intellectual		X													**7**
Social interactions										X					**7**
Water										X					**7**
Habitat										X					**7**
Genetic materials										X					**7**
Special Places				X	X		X		X			X			**36**

1 - Rolston and Coufal (1991); 2 - Manning et al. (1998); 3 - Reed and Brown (2003); 4 - Brown et al. (2004); 5 - Brown (2006); 6 - Raymond and Brown (2006); 7 - Brown and Raymond (2007); 8 - Alessa et al. (2008); 9 - Brown and Reed (2009); 10 - Brown et al. (2012); 11 - Brown and Weber (2012); 12 - Brown and Reed (2012); 13 - Brown (2005); 14 - Brown et al. (2015)

'X' denotes the inclusion of a value attribute in the given study. Numbers represent the number of sub-categories of that value attribute that were considered in the study.

Brown and Raymond (2007) cite several examples of scales that have been developed for the purpose of quantitatively measuring sense of place. For example, the New Ecological Paradigm (Dunlap et al. 2000) attempts to measure the environmental or ecological worldview of the public, including environmental attitudes, beliefs and values. The Natural Area Value Scale (Winter and Lockwood 2004) distinguishes between and determines the relative strength of use, non-use, and intrinsic values for nature. The Place Attachment Scale developed by Williams and Vaske (2003) is one of the first validated scales to systematically identify and measure sense of place, referred to as 'place bonds'.

Though these scales have been useful in planning, Brown and Raymond (2007) argue the need for the development of better place-based analytic tools that can more directly address the geographic dimensions of place. Brown (2005) discusses the need for measuring both the spatial distribution of 'important places' (geography of place) and the underlying 'perceptual rationale' for why those places are importance (psychology of place). This form of data could potentially be engaged by land use managers in various types of trade-off analysis.

There is, however, no definite consensus on how to specifically measure sense of place; especially in the context of diverse socio-cultural conditions (Kaltenborn and Bjerke 2002) and few techniques that explicitly provide for the inclusion of this form of knowledge into the planning and analysis (Brown et al. 2004). Much of the work that has been done focuses on collecting qualitative data about the connections of people with special places

(Mitchell et al. 1993, Brandenburg and Carroll 1995); data that are not easily integrated with existing biophysical inventories (Brown 2005). As shown in Chapter 5, landscape value attributes can be measured with a degree of accuracy, thus providing new opportunities to understand the spatial relationships between people and the ecosystem.

2.2.6 Landscape value attributes and well-being

The difficulties of linking ecosystem services to human health and well-being or the determinants of health were discussed above. Yet, a core element of this research is the social-ecological system; a clear inference of linkage between the health and well-being of humans and the services of ecosystems. As discussed above, landscape value attributes represent the values that humans place on the benefits derived from ecosystems. Implicitly, humans value those benefits because they contribute positively (directly or indirectly) to their determinants of health.

For example, the literature demonstrates that biodiversity, as an ecosystem service (Table 2-1), plays an important role in providing outputs that directly affect human well-being (Gómez-Baggethun et al. 2010, TEEB 2011, Sandifer et al. 2015). This link is also clearly captured through the lens of the landscape value attributes which place a high value on biodiversity (Mahboubi et al. 2015) (also see references in Table 2-3). This example serves to demonstrate that the valuing of ecosystem services (in this case biodiversity) through the lens of the landscape value attributes may, in fact, be an accurate reflection of the actual importance of those ecosystem services to human well-being.

Many of the other ecosystem services listed in Table 2-1 do not have the same weight of evidence to demonstrate their direct linkages and contributions to human health and well-being. However, based on the example of biodiversity above, insights gained from the knowledge of people can contribute to accurately describing those linkages. The accuracy of the knowledge gathered is an important underpinning of the validity of the measurements. Thus, the collection of 'expert' knowledge, rather than general public knowledge, may offer opportunities to achieve more accuracy.

2.2.7 Classes of expert knowledge

Over the past several decades, environmental governance has evolved internationally from a government-regulates-industry model to one that frequently involves joint regulation among government bodies, civil society, and industry across a range of sectors (Janicke 2008). This has resulted in a vast increase in the actors involved (Janicke 2008). Yet, Brown et al. (2004) demonstrated that not all residents or users of a place have equal knowledge. They assert the need to distinguish between those who 'care' about a place, but have little knowledge about it, and those who 'know' about the place.

The literature proposes various classes of knowledge. These include the knowledge resident at the various levels of authority in civil society, government and industry (Janicke 2008); in *policymakers, managers and professionals*, *academic institutions,* and *community members* (Boelen 2000); the knowledge held by *individuals, organizations* and the *community,* both *specialized and holistic* (Brown 2007), and knowledge that is *abstract (theoretical)* and *case-specific (practical)* (Pohl and Hadorn 2007, 2008, Pohl 2010).

The process of creating collective knowledge from among the various knowledge cultures is the basis of transdisciplinarity; a condition where the research has largely overcome disciplinary silos (i.e. the multidisciplinary condition) and has become associated with knowledge well beyond single perspectives of knowledge (Parkes 2011). The core prerequisite of the collective approach is the establishment of a "shared focus" among participants resulting in a "holistic understanding" that all can share, and the establishment of conditions that support "mutual understanding" where participants not only listen, but also hear one another; a condition rarely achieved under current management conditions (Brown 2007). The factors facilitating and restraining the incorporation of some of these knowledge classes into a collective whole are summarized in Table 2-4.

Table 2-4. A summary of some of the facilitating and restraining factors found in the various classes of knowledge expertise.

Knowledge Classes	Facilitating Factors	Restraining Factors
Adapted from Boelen 2000 (presented in the context of a collaborative process for integrated health planning)		
Policymakers	Broad vantage point and legal basis to establish long-term vision, prioritize needs and mobilize resources.	Tendency for political bias can influence public trust. Role may not extend far into translating policy to synergistic action. Lack of continuity can mean short-term support.
Managers	Typically have authority to reallocate human and material resources within the institution to support initiatives.	Tendency for vertical rather than horizontal or intersectoral approaches to solving problems.
Professionals	Often have direct contact with people. Governed by code of ethics. Positioned to implement policy. Valuable source of feedback from the field.	Potential influence/bias of corporate values and interests. Tendency to partner with like-minded organizations. Competition among professionals.
Academic institutions	Well-positioned to apply research methodology to problems. Typically held to a high standard of quality.	Can become isolated from social context. Specialization can occur at the expense of holistic vision. Inadequate leadership can limit multidisciplinary approaches.
Community members	Tend to problem-oriented approaches. Increasing awareness and influence. Valuable volunteer force.	Can be excessively demanding of outcomes. Challenging to retain in long-term partnerships. Can be influenced/ biased by the media.
Adapted from Brown 2007		
Individual	Grounded in personal experience. Perspectives linked to a desire for personal safety and global security.	May be dismissed as *personal anecdotal knowledge* or *non-knowledge*.
Community	Derived from shared events (stories and traditions existing in each community).	Can be disregarded as being non-credible.
Specialized	One of the dominant modes of knowledge of our time. Constructed by many disciplines and frameworks.	Can be challenging to non-specialists. Tendency to concentrate on single factors.
Organizational	Typically have well-established goals and an action agenda. Tend to have strategic approaches to sustainability.	Sustainability can become focused on long-term survival of the organization and its profits, rather than to people or planet. Difficulty achieving unified action.
Collective (holistic)	Synergetic potential through integrating diverse knowledge and capacity.	

2.2.8 Monetary approaches to measuring social-ecological and health impacts

The previous sections described some of the cross-linking relationships understood to exist between ecosystem health, the flow of ecosystem services and human health/the determinants of health (section 2.2.4). Landscape value attributes were also discussed in the context of better understanding and valuing ecosystem spaces through the human lens (section 2.2.5 and 2.2.6). It was further discussed that despite recognizing certain cross-linkages, the task of quantitatively measuring them is challenged. Yet, institutions such as the European Integrated Maritime Policy and the European Biodiversity Strategy to 2020 (CEC 2007, 2011) are calling for new research to develop better methods of quantifying ecosystem services in order to better integrate the information into decision-making processes.

Boyd and Banzhaf (2007) argues for economic approaches to measuring the contributions of ecosystem services to humans and proposes monetary units as an appropriate universal unit of measure, thus allowing for important analyses, such as compensation or cost/benefit analyses, to be conducted. Economic models operate on monetary valuation methods. Monetary valuation normally involves deriving a fair or proper equivalent value in money or commodities for the ecosystem service of interest (Hadley et al. 2011). The term 'equivalent' is used to represent the sum of money that would have an equivalent effect on the welfare/well-being of individuals (Hadley et al. 2011). Monetary valuation can also be applied to non-market ecosystem services, thus allowing their inclusion in the cost-benefit analysis. Thus, the total economic value of an ecosystem can be derived by determining the sum of its use and non-use values. Use-values can be consumptive (e.g. the value of fish harvested) or non-consumptive (e.g. the cultural value attached to the ecosystem or its value in terms of climate regulation through sequestering carbon). Non-use values are related to the values that humans place on ecosystems for their existence (Hadley et al. 2011). A number of studies have attempted to approximate the total monetary value of a given ecosystem, examining both use and non-use ecosystem services (see Johnston et al. 2002, Olewiler 2004, Wilson 2008, e.g. de Groot et al. 2012, Costanza et al. 2014).

There are a number of approaches to conducting monetary valuation. They fall into two general categories: estimating equivalents to prices (*pricing approaches*) and estimating economic values (*valuation approaches*). The two approaches are distinct and may produce very different results given that the price derived for a good or service can differ greatly from its economic value (Hadley et al. 2011).

2.2.8.1 Pricing approaches

There are three methods of pricing: deriving market prices, opportunity costs and replacement costs (see Table 2-5 for a summary of each method including pros and cons). Pricing approaches are generally simple to apply for ecosystem services that have a market value. For example, a report prepared for the BC Oceans Coordinating Committee (GSGislason & Associates Ltd. 2007) generates a market price for all of the commercial or market components of the Pacific Ocean of BC. Market pricing is also an important planning and management component of Fisheries and Oceans Canada (DFO). For example, a running tally of the annual tonnage of each species harvested from each pacific fisheries management area (PFMA) is converted to a dollar amount. As an example, the data show that in 2010 approximately 473,000 pounds of sockeye salmon were harvested from PFMA 3, which is then converted to $601,000 of market value (DFO 2013c).

There are, however, large gaps in the data. In some cases the data are simply absent. For example, areas not open to commercial fishing will not have data to reflect the species that may be present there. In other cases, the data are absent due to the difficulties of measuring certain non-commercial and non-market values such as aesthetic, cultural, traditional, and recreational values (De Groot et al. 2002, Millennium Ecosystem Assessment 2005). These data gaps can be significant given that non-market values are often essential to human welfare. For example, De Groot et al. (2003) argue that despite not providing direct economic benefits, the non-market values of ecosystem services can, in fact, far outweigh their direct consumptive and productive use values.

2.2.8.2 Valuation approaches

Addressing data gaps presents a significant challenge. There are two categories of valuation approaches that can help address this: stated and revealed preferences (see Hadley et al. 2011). Contingent valuation (CV) is an example of the former and is regarded as one of the preferred valuation techniques available. It was the chief method used to determine the environmental impact of the Exxon Valdez oil spill (Carson et al. 2003). The method simulates a market by way of a questionnaire which presents individuals with hypothetical situations, giving them choices to buy or sell specific ecological services. Individuals are asked to trade off gains and losses to ecosystem services against money by expressing their agreement with certain preventative measures together with an associated monetary cost. It determines what people would be willing to pay (WTP) to prevent specified changes in the quantity or quality of their environment and the uses thereof, or what they would be willing to accept (WTA) in compensation for specified impacts (Carson and Hanemann 2005).

De Groot et al. (2003), (2012) argue that with the help of market and shadow pricing techniques it is, in fact, possible to derive a monetary value for all of the goods and services provided by natural capital. Carson and others argue that where other methods may be adequate for valuing the cost of impact to certain tangible marine uses (e.g. commercial fishing), CV is the only way to estimate non-use values (i.e. where there is no direct involvement with the resource) (Carson et al. 2003). De Groot et al. (2003) contend that "for most types of values it is possible to arrive at a monetary estimation of their (relative) importance to human society (in terms of their expressed or stated willingness to pay for the continued availability of a given good or service)." There are also a number of other valuation approaches that have been used to value ecosystem services. Each has its strengths and weaknesses and ideal applications. Table 2-6 presents a summary of some of the more popular valuation approaches in use, including their pros and cons.

Table 2-5. A summary of various pricing approaches available for the valuation of ecosystem services, including the pros and cons of each approach.

<table>
<tr><td rowspan="6">Pricing Approaches</td><td rowspan="2">Market Pricing</td><td rowspan="2">Estimates market prices for ecosystem services that are traded, either in local or international markets.</td><td>Pros</td><td>• Relatively simple.</td><td rowspan="6">Croitoru (2007)

Pearce et al. (2006)

EPA (2000)</td></tr>
<tr><td>Cons</td><td>• Can only estimate direct use values.
• Prices can be distorted by market failure.
• Only a partial measure of value.</td></tr>
<tr><td rowspan="2">Opportunity Cost</td><td rowspan="2">Estimates the benefits that are foregone when a particular action is taken. For example, revenues from timber sales versus the lost opportunity of reaping the benefits of other forest products.</td><td>Pros</td><td>• Can be relatively simple.
• Can be useful where policy precludes access to an area.</td></tr>
<tr><td>Cons</td><td>• Can only estimate direct use values.
• May require detailed public surveys to determine economic and leisure activities in the area in question.
• Only a partial measure of value.</td></tr>
<tr><td rowspan="2">Replacement Cost</td><td rowspan="2">Estimates the cost of providing an alternative resource with the same function as the ecosystem service. Example, the cost of building a man-made flood defense system to replace the flood protection service of a wetland.</td><td>Pros</td><td>•
• Relatively simple.
•</td></tr>
<tr><td>Cons</td><td>• Can only estimate direct use values.
• Only a partial measure of value.</td></tr>
</table>

* Summarized from Hadley et al. (2011)

Table 2-6. A summary of the major valuation approaches used to value ecosystem services, including the pros and cons of each.

Valuation Approaches	Stated Preferences Method	Contingent Valuation (CV)	Uses public surveys to determine what people would be willing to pay (WTP) or willing to accept (WTA) for a specified gain or loss to a given good or service.	Pros	• Can estimate use and non-use values. • A widely used and much researched environmental valuation technique. • Applicable to many ecosystem services.	Georgiou et al. (1998)
				Cons	• Can suffer from a wide range of biases. • Resource intensive. • Complex statistical analysis.	
		Choice Modeling Experiments	Contingent ranking: Participants rank alternative scenarios in order of preference. Choice experiment: Participants choose a preferred option among several scenarios with associated costs. Analysis yields WTP for each scenario.	Pros	• As with CV but more flexible.	Hanley et al. (2006)
				Cons	• As with CV but even more consideration to design to avoid biases. • Statistical analysis even more complex.	
	Revealed Preferences Method	Travel Cost Method	Estimates costs incurred by individuals travelling to sites (i.e. travel expenses and lost earnings in time) as a proxy for the recreational value of the site.	Pros	• A well-established technique. • Based on actual observed behavior.	Font (2000)
				Cons	• Only estimates use values. • Applicable mainly to recreational sites. • Confounding factors create complications. • Resource intensive. • Complex Statistical analysis.	

Hedonic Pricing	Values services such as landscape amenity, air quality, and noise by determining the effect of these on a marketed good (usually the housing market).	Pros	• A well-established technique. • Based on observed behavior and existing data.	Leggett and Bockstael (2000)
		Cons	• Only estimates use values. • Applicable only to environmental attributes likely to affect price of housing and/or land. • Depends on awareness of property owners. • Prices can be distorted by market failure. • Data intensive. • Complex Statistical analysis.	
Averting Behavior Defensive Expenditure	Similar to Travel Cost and Hedonic Pricing, but based on costs incurred to avoid negative impacts. Example, buying double glazed windows to avoid noise.	Pros	• Sound theoretical basis. • Uses data on actual expenditures	Bresnahan et al. (1997)
		Cons	• Not a widely used methodology. • Can only estimate use values. • Limited to households that spend money to offset environmental hazards/nuisances. • Depends on awareness of those affected. • Appropriate data may be difficult to obtain.	
Cost of Illness & Lost Output Method	Based on the cost of illness and lost output calculations. Example, air pollution can lead to an increase in medical costs to treat the associated health impacts, as well as, a loss of wages and profit.	Pros	• Theoretically sound. • Useful if exposure-response relationship exists. • Can be relatively simple.	Davies (2006) Kuik et al. (2000)
		Cons	• Can only estimate use values. • Uncertainty regarding exposure-response. • Prices can be distorted by market failure. • Complex and resource intensive if exposure-response relationships not established.	

* Summarized from Hadley et al. (2011)

2.2.8.3 Challenges to economic models of valuing ecosystem services

The economic valuation approaches reviewed above are capable of providing monetary values for all ecosystem services, including those services with non-consumptive use and non-use values. Certain jurisdictions rely on economic approaches to fill data gaps with monetary values (van der Ploeg et al. 2010). But despite their popularity, such valuation techniques have also been heavily criticized due to the difficulty of capturing monetary values for many ecosystem services (Shmelev 2010, Chan et al. 2012, de Jonge et al. 2012), the hypothetical character of the simulated market (Hausman 1993) and cultural biases leading to problems of credibility (Diamond and Hausman 1994). Hadley et al. (2011) recognize the complex challenges associated with monetary valuation but contend that these approaches can, nevertheless, help shed light on the human well-being benefits and costs derived from ecosystems. Martin-Lopez et al. (2009) assert that economic values do not, and cannot capture the full value of ecosystems; particularly the value of services that fall outside of the sphere of markets (such as cultural and spiritual services).

It is also uncertain the extent to which monetary valuation might be embraced in the public sphere, especially given the growing consciousness of the expansive immaterial contributions of ecosystem services to human well-being. Satterfield and Roberts (2008) note that it is often these very immaterial ecosystem benefits that become the central theme of public discontent around natural resource management decisions. The statement of the Columbia River Intertribal Fish Commission (2010b) that, "without salmon returning to our rivers and streams, we would cease to be Indian people" is a statement of monetary inestimability of a cultural ecosystem service, and speaks poignantly to some of the complexities of monetary valuation.

It is noteworthy that the criticisms associated with monetary ecosystem valuation (CV in particular) were also considered by a panel of the National Oceanic and Atmospheric Administration (NOAA), concluding that CV studies do convey useful information for damage assessment, provided they follow a number of stringent guidelines (Arrow et al. 1993, Carson et al. 2003). Hadley et al. (2011) recognize the complex challenges associated with monetary valuation but contend that these techniques can, nevertheless, be used to help shed light on the human well-being benefits and costs derived from ecosystem changes.

2.2.9 Environmental, health and social assessment frameworks

In the previous sections of this chapter a range of core concepts related to human and environmental health and the challenges of their measurement and integration were presented. Three communities of scholarship are distinctly engaged in working with these concepts and addressing the challenges discussed within the field of research and practice. Their works are published in a broad range of environmental, health and social sciences literature and categorized as one of three assessment frameworks:

1) **Environmental assessment (EA)** designed to assess or predict the potential environmental impacts of projects;

2) **Health impact assessment (HIA)** designed to identify, predict and evaluate the likely changes in health risk to humans (both positive and negative, single or collective) of a project on a defined population;

3) **Social impact assessment (SIA)** which seeks to investigate and understand the social consequences of planned change.

2.2.9.1 Environmental Assessment

EA is the only assessment framework, among those considered, that is legislated in Canada (CEAA 2012). The overarching objective of the legislation is to "encourage federal authorities to take actions in a manner that promotes sustainable development in order to achieve or maintain a healthy environment and a healthy economy" [s4.1]. The EA process was established as a means of achieving this objective by assisting regulatory bodies to assess and predict the potential environmental impacts of projects, programs or policies and provide plans and strategies to mitigate those impacts and minimize the risks of adverse effects (Kwiatkowski and Ooi 2003, Yap 2003, Health Canada 2004, Noble and Bronson 2005).

The process framed by the CEAA 2012 is outlined in Figure 3-2. In summary, proponents wishing to undertake a project will supply a project description [s8] to the CEA. The CEA will determine whether the project will require an EA [s10] and whether it or another Agency (the National Energy Board - NEB, the Canadian Nuclear Safety Commission - CNSC, or another federal agency) should be the responsible authority [s14.4a-d] and [s15a-d]. Projects may require an EA if they are listed in the regulations and/or the Minister of the Environment (Minister) is of the opinion that the project may cause adverse environmental effects, or that public concerns related to those effects warrants the EA. The latter considerations do not pertain to projects that fall to the NEB or the CNSC as the responsible authority, all of which require EAs. As an alternative to appointing a responsible authority to oversee the EA, the Minister may appoint a Review Panel instead (its members appointed by the Minister and supported by the CEA) [s38.1 & 2].

Both categories of EAs could solicit the involvement of the Provinces into the process. In fact, the Minister is required to allow a requesting provincial process to **substitute** for a federal EA if it is deemed that the provincial EA is an appropriate substitute. In the event of a substitution, the Province conducts the EA, but the federal government has the ultimate decision-making power. Alternatively, there is also the provision for an **equivalency** under CEAA 2012. In this instance, if the provincial process meets all of the conditions for a substitution, the Minister may recommend to the Governor in Council that a designated project be exempted from the application of CEAA 2012 [s32.1].

Once an EA is deemed necessary, a scoping of the project proposal [s19.2] reveals the environmental effects that may need to be examined and planned for. Environmental effects may range from impacts to species and habitats [s5.1a], to the health and well-being of people [s5.1b & c] or other relevant matters [s19.1j]. Additional studies may be required to ascertain the full implications of any potential impacts. The final review is returned to the responsible authority, Minister or Government in Council for a decision process.

2.2.9.2 Health Impact Assessment

Another approach to impact assessment that initially evolved in tandem with the EA process in Canada is the HIA. HIA emerged from the field of public health, with major milestones achieved following a number of major international conferences, but has (over a period of two decades) received variations in attention and profile in the Canadian policy scene.

An important impetus for HIA arose in 1986 at the First International Conference on Health Promotion held in Ottawa, where the WHO, together with Health Canada and the Canadian Public Health Association, agreed on the Ottawa Charter for Health Promotion (WHO 1986). The Ottawa Charter views health in the context of a reciprocal interaction between the person and the environment and recognizes the elements of our social environment including, peace, shelter, education, food, income, social justice and equity as a part of that context, and thus, relevant to human health. This international Charter was explicit in its recognition that the physical environment is important to human health, and expressed the need for a "stable ecosystem and sustainable resources" (Health Canada 2004) as well as recognizing the 'socio-ecological' context for health (WHO 1986). Next, the United Nations Conference on Environment and Development (the Earth Summit), was held in Rio de Janeiro in 1992 where over 150 member states adopted Agenda 21; an action plan to guide future strategies for health and environmental activities on a national and international level. Over a decade later, the WHO established the Commission on Social Determinants of Health, bringing together a global network of policy makers, researchers and civil society organizations to give support in tackling the social causes of poor health and avoidable health inequalities (health inequities) (CSDH 2008).

These events corresponded to another period of Canadian leadership in the development of HIA. Building on the broad definition of health as "a state of complete physical, mental and social well-being" (WHO 1992), the works of Frankish et al. (1996), McKeown (Glouberman and Millar 2003) and others (see section 2.2.3 above), Health Canada introduced the 'determinants of health' in Canada as one of a series of efforts focused on consolidating health as a conscious concern for all sectors in society, not just the 'health sector' (Health Canada 2004). These same determinants are still in use today in Canadian policy (e.g. Public Health Agency of Canada 2011).

The objective of an HIA is to determine the potential effects of proposed policies or projects on the health of a population (be it on specific health indicators or the broader determinants of health). HIA methods use an array of data sources and analytic tools including input from stakeholders to examine the distribution of health effects within a population (National Research Council 2011). The HIA process involves five-steps: screening to establish whether the HIA is warranted, scoping to plan the extent of application and the resources required, appraisal or analysis of available information, reporting and, lastly, evaluation and monitoring to manage any effects (St-Pierre and Mendell 2011).

2.2.9.3 Social Impact Assessment

Another distinct community of scholarship and practice also examining the question of impacts to human is the SIA community. Despite significant overlaps between HIA and SIA, little effort has been made to combine them (Becker and Vanclay 2003). Instead, they have occurred as parallel processes. As with HIA, the SIA has also not been fully or systematically integrated into the EA process (Yap 2003).

The Interorganizational Committee on Principles and Guidelines for Social Impact Assessment (2003) defines SIA as "an area of systematic inquiry which seeks to investigate and understand the social consequences of planned change". SIA involves analyzing, monitoring and managing the social consequences of development with the primary purpose of bringing about a more ecologically, socio-culturally and economically sustainable and equitable environment (Burdge 2003, Vanclay 2003). Vanclay (2003) describes social impacts as changes to one or more of eight parameters: people's way of life, their culture, their community, their political systems, their environment, their health and wellbeing, their personal and property rights, and their fears and aspirations (for their own future, that of their children and their community).

Although SIA publications have increased over the years from approximately 100 citations per year in the 1980's to over 600 citations in 2010 (Esteves et al. 2013), its practice in the Canadian context is very limited. According to Vanclay and Esteves (2011), its potential contribution is not being fully achieved in the way it is being practiced, largely due to the "limited understanding and skills of those who commission SIAs" (Esteves et al. 2013).

The challenges associated with the application of these three assessment frameworks (as discussed above and in Chapter 3) are compounded by the challenges of integration which have become a paramount concern in recent times. The efforts of these communities of scholarship to integrate their works have not been successful. Appropriate methodological frameworks are needed to advance the integrations process. Two such frameworks are presented in the following section: ecohealth and PPGIS.

2.2.10 Ecohealth and PPGIS: Approaches to integration

The EA process was described in Chapter 1 as the first significant opportunity for an integrated analysis of impacts. Positioned at the "junction of science, policy, and politics" (Cashmore et al. 2010), Mindell et al. (2010) contend that the EA process can offer a "systematic strategy to promote synergies among health and environmental institutions and disciplines".

Yet, as described in Chapter 3, the EA framework has fallen short of achieving integrated analysis, and instead tends to focus heavily on biophysical impacts. Frustrations over the poor integration track record and the many challenges to integrated analysis have resulted in an appeal, in certain jurisdictions, for a stand-alone HIA process, distinct from the EA process (see section 3.2). Despite these appeals, many argue for the continued pursuit of integration as a more effective approach (Briggs 2008, Morgan 2011, Tajimaa and Fischerb 2013). Methodological theories and tools provided by the fields of ecohealth and PPGIS offer relevant insights useful to the challenge of integrating social, ecological and health factors.

2.2.10.1 Ecosystem approaches to health

Ecosystem approaches to health (ecohealth), consistent with the principles of the 2005 Millennium Ecosystem Assessment, are underscored by the critical insight that human health and well-being are important outcomes of effective ecosystem management (Parkes et al. 2008) and that health is the result of complex and dynamic interactions between the determinants of health and between people, social and economic conditions and ecosystems (Charron 2012). Thus, the primary objective of ecohealth is to provide appropriate and equitable interventions aimed at reducing or reversing the negative health effects of ecosystem change. To achieve these objectives, ecohealth is largely integrative in nature. For example, it is participatory, bringing together academic, government and civil-society stakeholders, and transdisciplinary, integrating knowledge across and beyond academic disciplines (see Parkes et al. 2005, Charron 2012).

Ecohealth approaches do not necessarily provide specific tools or procedures. Instead they are a "mindset that orients a process of inquiry" (Charron 2012). For example, Waltner-Toews (2011) contends that ecohealth cannot be heavily prescriptive in its methods, as it is not a prescription but rather an approach to health rooted in complex systems thinking and stakeholder engagement. Instead ecohealth draws on the many disciplines related to each specific problem being addressed in order to determine the most effective steps to take (Fèvre et al. 2013). Case studies demonstrate that ecohealth approaches can be effective in developing innovative solutions to integrated human and environmental health challenges (Charron 2012).

The objectives of this research were focused on social-ecological integration challenges involving a range of disciplines including ecology, health, economics and legislation. Pragmatic forms of integration were needed to determine appropriate steps to address the objectives. Furthermore, an important data source in the analyses was that of human knowledge. Given the focus of ecohealth on disciplinary integration, methodological customization of approaches, and its participatory underpinning, ecohealth provided a valuable framework for this research.

2.2.10.2 Public Participatory geographic information systems

The term 'public participatory geographic information systems' was conceived at the 1996 meeting of the U.S. National Center for Geographic Information and Analysis (NCGIA). In general, PPGIS emerged from the context of developed countries, while the similar and related approach of 'participatory GIS' or PGIS describes participatory planning approaches applied in rural communities of developing countries. PPGIS is an outcome of a merger between Participatory Learning and Action (PLA) methods and geographic information technologies (Rambaldi et al. 2006, Brown and Pullar 2011). It describes how GIS technology could be used to support greater inclusion and empowerment of marginalized populations (Brown and Reed 2009) by engaging participants in the identification of ecosystem services. Brown and Reed (2011) argue that PPGIS can provide a useful coarse filter for identifying the location of ecosystem services, including many intangible or abstract services.

PPGIS also addresses one aspect of the need for measurement by tapping the knowledge of the general public to quantifiably and spatially detect a range of social and ecological hotspots (Brown 2012a). The PPGIS survey instrument is sent to every household in the community (Brown and Reed 2009) and the data collected are assumed to represent the views of the "silent majority" (Alessa et al. 2008), the broad views of the entire local population (Brown and Reed 2012).

Brown and Reed (2009) discuss the difficulty of natural resources management planning in an environment where the values held by various groups conflict, the goals of the planning process are ambiguous or contested, and the scale of planning and analysis is varied. Similarly, Lachapelle et al. (2003) notes the poor performance of traditional rational-comprehensive planning models under these 'messy' conditions. They argue the need for a standardized quantitative method of collecting and interpreting the values that the public holds for natural spaces (Brown and Reed 2009). PPGIS can potentially fulfill this objective (Brown and Reed 2009), offering decision-makers a "common operating picture of the problem domain and its geospatial characteristics" (Sani and Rinner 2011).

It is noteworthy that among the various approaches proposed in the literature to harmonize social and environmental concerns as an aid to natural resources planning and ecosystem management, those that are place-based or place-specific are attracting increased attention in many parts of the world (McIntyre et al. 2004). Yet, government adoption of PPGIS methods for decision-support has lagged due to a number of barriers, the most prevalent being mistrust of the precision of the data (Brown and Reed 2011). The 'perceptual' nature of PPGIS data stands in contrast to *expert-driven* GIS systems (perceived as highly accurate). Thus, PPGIS data are often regarded to be more about the participatory process than the generation of rigorous spatial data (Brown and Reed 2011).

2.3 Conclusions

Investigating the range of considerations involved in environmental planning and management processes, such as the EA and HIA, requires attention to several bodies of relevant scholarship from fields such as ecology, health and economics. Moreover, an integrative process of investigation is necessary in order to fully appreciate the dynamic interactions that can occur between the findings of these fields. Investigation of the literature related to each field revealed certain core concepts needing further exploration. For example, the notion of human health introduces questions of the operational definitions of human health: physiological health, well-being and the determinants of health. The same questions arise with respect to the concepts of ecosystem health and ecosystem services. Integrative considerations between human and ecosystem health led to a review of core concepts related to the cross-linking relationships between them. The determinants of health share a common ground with the concept of landscape value attributes. These attributes are based on human knowledge and the process of human valuation of landscapes. Thus, a focus on the various forms of knowledge discussed in the literature becomes relevant to the investigation. Conversely, economic approaches also allow for the valuing of ecosystems. Therefore, an understanding of monetary approaches to measuring ecosystem values is also required. This research examined these core concepts in the context of several important health and environmental processes including the EA and HIA. The analysis was framed by two relevant methodologies, ecohealth and PPGIS, both of which were introduced as appropriate frameworks due to their integrated nature, their aptitude to cope with complex systems problems, their inclusion of human knowledge and, in the case of PPGIS, the availability of spatial tools.

Chapter Three. Challenges and opportunities of integrating human health into the Environmental Assessment process: the Canadian Experience contextualized to international efforts

3.1 Introduction

Environmental industries with significant ecological impacts, such as those of the natural resource extraction sector, have long served as an important component of the Canadian economy (Hessing et al. 2005). The combination of forestry, mining, fishing and oil and gas extraction, together with secondary processing, refining and the activities of the transportation industries, have resulted in significant ecosystem impacts (e.g. AXYS 2002, Schneider et al. 2003, Lee et al. 2009, Krzyzanowski and Almuedo 2010). Collectively, the environmental impacts of these activities contributed to Canada being ranked 104th out of 146 countries for environmental stewardship (Esty et al. 2005). Deteriorated ecosystems can adversely affect the determinants of human health and ultimately health outcomes (MA 2005, Parkes et al. 2008, Charron 2012); where health outcomes are measured by a change in mental, social and physical well-being (Gibson Parrish 2010).

A central challenge for governments with resource-based economies is to balance the demand for resource extraction revenues, while containing environmental impacts within acceptable social, ecological and health limits. Many governments have established legislation to help achieve this purpose. In Canada, the Canadian Environmental Assessment Act (CEAA) governs the evaluation of proposed projects to determine whether they will proceed and how their anticipated impacts will be managed (CEAA 2012). The Environmental

Assessment (EA) is the goal of this legislation and is used to assess the potential environmental impacts of projects and to plan strategies to mitigate those impacts in order to minimize the risks of adverse effects to both human and environmental health (Canadian Environmental Assessment Agency 2014).

Yet, despite existing legislation and notwithstanding the efforts of the environmental, health and social sciences, with the associated tools of Environmental Impact Assessment (EIA), Health Impact Assessment (HIA) and Social Impact Assessment (SIA), the inclusion of human health into the EA process has had limited success. In the Canadian context, these limitations have led to increasing public concern over the health impacts of proposed projects, on-going frustrations among proponents over the time and resources required to address health, and growing demands placed on regulators to balance these interests in a politically and legally charged environment. These challenges have led to a sense of urgency and renewed attention to the health/EA interface.

This paper explores some of the current challenges of integrating health considerations into EA processes in Canada and internationally. A scoping review of the HIA literature helps to identify the key issues that may be responsible. The key issues are framed within the Canadian CEAA which was revised in 2012, and discussed in the context of the status quo practice of EA. Recommendations are proposed to help address the key issues.

3.2 Background and Context: Health and the Canadian Environmental Assessment Process

According to the 'Best Practices Principles' of the International Association for Impact Assessment (IAIA), the HIA process should serve to inform and influence decision-making, and ensure that health protection and promotion are effectively integrated into proposals and plans (Quigley et al. 2006). More specifically, the HIA should serve as an intersectoral process, engaging health experts, project proponents and other key players including communities affected by a proposal, into the decision-making, to the degree appropriate (see Tajimaa and Fischerb 2013). Quigley et al. (2006) contend that the HIA process need first define what constitutes health status in a given context and then identify the health inequalities that may arise from the proposal, addressing any related cross-cutting health issues. With these elements in place, they argue that the HIA should be recognized as a 'license to operate'.

Since its beginnings, the CEAA has recognized that human health is an important consideration. For example, the U.S. National Environmental Policy Act (NEPA) 1969, the predecessor and impetus for institutionalizing the EA process in Canada, made reference to the "human environment". In 1973 the U.S. Council on Environmental Quality clarified this to mean "the natural and physical environment" and "the relationship of people with that environment" and required agencies to assess the aesthetic, historic, cultural, economic, social or health effects of projects (Galisteo 2002). Thus, EAs have long been mandated to include health and social considerations in their assessments (Deng and Altenhofel 1997, Yap 2003). The most recent legislation (CEAA 2012) directly addresses the protection of human health [s4.2], including the "health and socio-economic conditions" of Aboriginal peoples [s5.1c(i)] and others [s5.2b(i)]; the latter applying only when a project requires the federal authority to exercise powers or duties under an Act of Parliament other than CEAA 2012 (Government of Canada 2012).

Yet, the emphases of EA practices in Canada and elsewhere have largely focussed on impacts to the biophysical environment, leaving the great majority of EAs without appropriate health and safety assessments or considerations to broader health, cultural, social and economic impacts (Steinemann 2000, Murphy and Kuhn 2001, Yap 2003, Health Canada 2004, Harris et al. 2009, Morgan 2011). In cases where social health issues are monitored, they rarely seem to be monitored well and are certainly not treated with the same rigor as the monitoring of biophysical impacts (Noble and Bronson 2005, Morgan 2011). Wright (2011) notes that health assessments conducted during the EA process often amount to no more than a stand-alone HIA. Harris-Roxas et al. (2012) contend that government health agencies typically view HIAs as a novel activity rather than as a core capability.

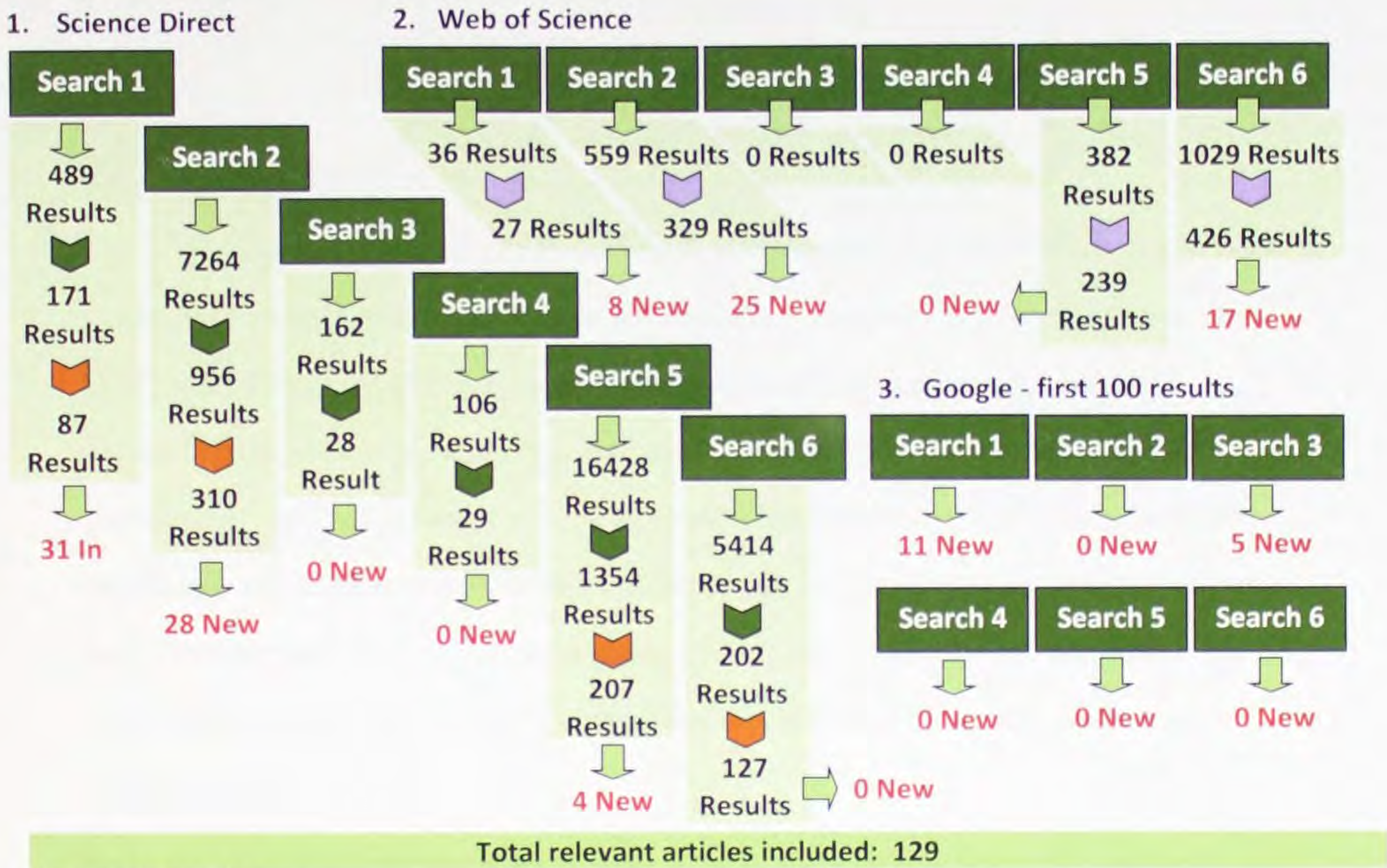
Attempts to apply HIA have grown substantially in the past 20 years (Harris-Roxas and Harris 2013, Lee et al. 2013); particularly in nations such as Australia, Norway, Sweden, the United Kingdom and the United States. In some jurisdictions, such as England and Sweden, HIAs are conducted a stand-alone processes, applied to a wide range of public policy decisions that are not subject to the EA (Kemm and Parry 2004, Bhatia and Wernham 2008). In most jurisdictions, however, HIAs are not integrated as part of assessment legislation, which leaves best practice HIA to be conducted in an *ad hoc* manner outside of any legislative or regulatory requirements (Wismar and Ernst 2010). In Canada, Australia and New Zealand, health considerations are channelled through the legislated EA process which does not require stand-alone HIA (Noble and Bronson 2005, Morgan 2011, Wright 2011) and, as such, are rarely consistent with 'State of the Art' HIA practices (Harris-Roxas et al. 2012).

The poor results obtained from addressing health considerations in existing EA processes have led some localized government jurisdictions to attempt to institutionalize the HIA (Banken 2001, Lee et al. 2013) or address health through other channels (see for example OCMOH 2012). A number of institutions have begun piloting new enterprising projects including the U.S. Health Impact Project, the EU IMP(3) Project, and the Australian HIA Connect (Health Council of Canada 2010, Morgan 2011, Tarkowski and Ricciardi 2012). These projects and initiatives demonstrate the potential of respective nations to use HIA (Blau et al. 2007), the potential of HIA to account for health considerations (Harris and Spickett 2011) and the ability of local health departments to play a constructive role in the EA process (Bhatia 2007).

Canada took certain initial steps to help develop a broader understanding of the relevance of environmental dynamics to health impacts (e.g. Ottawa Charter for Health Promotion, 1984) and showed leadership investigating the usefulness of the determinants of health to inform the HIA (Health Canada 2004). However, Canadian efforts have not kept pace with international efforts and the publications and reports produced have not resulted in significant innovations in practice. As stated by the Health Council of Canada (2010), "Canada has a history of producing landmark documents on health promotion that are greeted with enthusiasm but don't stimulate as much action – or the kind of action – as expected (p29)". Despite efforts in Canada and internationally, the integration of health considerations into the EA process is significantly challenged (Bhatia and Wernham 2008, Harris and Spickett 2011, Morgan 2011, Morrison-Saunders and Retief 2012, Lee et al. 2013) and does not generally meet the best practices principles of the IAIA. In response to these concerns, this paper seeks to examine the HIA literature from the last 15 years to better understand the challenges and opportunities to enhance HIA, with particular attention to the dynamics arising when HIA is conducted within the EA process.

3.3 Methods

According to Levac et al. (2010) scoping reviews provide valuable syntheses of findings which help inform and contextualize subsequent systematic reviews and primary studies. This scoping review followed the general steps suggested by Arksey and O'Malley (2005) of identifying the research question, locating relevant studies, selecting works from those studies, and charting and summarizing the findings. The purpose of this scoping review was to examine the challenges of applying HIAs, particularly in the context of EAs, through a thematic categorisation of issues discussed in the literature. The literature sought was that published between the years 2000-2015 in the English language in peer reviewed journals and government sources. Three search engines were used to source literature including: ScienceDirect™, Web of Science™ and Google™. The search criteria applied, the filters used to short-list results and the sequential results achieved at each stage of search are presented in Figure 3-1 below. Table 3-1 presents the categorization of those results by thematic issues, scope and geography.



Keyword searches

Search 1: *"health impact" "environmental assessment"* ; Search 2: *"health impact" environmental impact assessment* ; Search 3: *Canada EA "health impact"* ; Search 4: *Canada integrated EA "health impact"* ; Search 5: *Integrating "human health" environmental impact* ; Search 6: *HIA*

Science Direct Topic inclusion filters

impact assessment, environmental impact, health impact, eia, public health, human health, world health, environmental, health, impact, result, human.

Science Direct Publication Site inclusion filters

Environmental Impact Assessment Review, Public Health, Environmental International, Science of the Total Environment, Encyclopedia of Environmental Health, Environmental Research, Health Policy, Journal of Environmental Management, The New Public Health, International Encyclopedia of Public Health, Ecological Economics, Environmental Science & Policy, Land Use Policy, Resources Policy, Health & Place, Journal of Environmental Sciences.

Web of Science Research Area inclusion filters

Environmental Sciences Ecology, Public Environmental Occupational Health, Health Care Sciences Services, Urban Studies, Government Law, Demography, Biodiversity Conservation, Social Sciences Other Topics, Science Technology and Other Topics.

Figure 3-1. A schematic overview of the databases, search criteria and filters used to identify literature related to the challenges of applying health impact assessments and integrating health considerations into the environmental assessment process. The sequential results achieved at each stage of search are shown.

3.4 Review of findings

An overview of the articles found in a scoping review of the literature is presented in Table 3-1. The findings are presented by themes related to the key challenges of applying health impact assessments and integrating human health considerations into the EA process. This section expands on the findings of Table 3-1 and examines the issues and challenges of integrating health into the environmental framework. The findings are also reviewed in the context of the Canadian EA legislation to help identify some of the driving forces behind them. An analysis of the issues also provides the basis for a number of recommendations to aid future EA planning and research which are discussed in the following section.

Table 3-1. A summary of articles found in a scoping review of the literature related to the challenges of applying health impact assessments and integrating human health considerations into the environmental assessment, categorized by theme, scope and geography.

Total number of articles found	**129**			
Case-study-based analyses (national or project level)	63%		Number of case-studies cited within articles	190
Theoretical analyses	37%			
Geographic focus	**Focus of articles**	**Geography case studies**	**Issues to application/integration**	**% of articles**
Europe	**50%**	**31%**	**Government Intervention needed**	**48%**
European Union	46%	29%	(Leadership, Policy, Institutionalization)	(16, 20, 12)
Eastern Europe	4%	2%	**Methodological & procedural limits**	**45%**
Asia	**9%**	**4%**		
East Asia	7%	3%	**Skills, capacity and expertise gaps**	**26%**
Middle East	2%	1%	**Intersectoral & disciplinary collaboration**	**26%**
Australasia	6%	15%		
Africa	5%	3%	**Need for public participation**	**16%**
Americas	**22%**	**45%**	**Data quantification challenges**	**10%**
USA	15%	38%	**Process pace and the need for efficiency**	**10%**
Canada	6%	6%		
South America	1%	0%	**Cross-linking & analytic complexity**	**10%**
International	4%	2%	**Poor promotion of HIA**	**4%**
Corporate	2%	1%	**Other issues**	**17%**

3.4.1 Taking stock of nearly half a century of effort in Canada

As discussed, the importance of considering the human health impacts of environmental projects has long been recognized in Canada. The Canadian federal government adopted its first piece of related legislation, the Environmental Assessment Review Process (EARP), over four decades ago. It soon became apparent, however, that the scope of the EARP with respect to assessing health impacts was very limited. A decade later, Canada (through Health Canada and the Canadian Public Health Association) contributed to innovative thinking about the determinants of health through its contributions to the production of the Ottawa Charter for Health Promotion (WHO 1986). This initiative increased recognition in Canada and internationally that the fundamental conditions of health, including peace, shelter, education, food, income, a stable ecosystem, sustainable resources, social justice and equity, are relevant to human health and warrant a 'socioecological ' approach to health (WHO 1986).

A more pragmatic step for the assessment of health impacts in Canada came when the EARP was replaced with CEAA 1992. The new legislation included more explicit language about health. For example, it defined 'environmental effects' to include, among other things, effects on "human health and socio-economic conditions" [s2] and required their consideration in the EA process [s16.1]. In the same year, Canada took another step at the 1992 United Nations Conference on Environment and Development (the Earth Summit) held in Rio de Janeiro. Together with over 150 member states, Canada adopted Agenda 21, an action plan to guide future strategies for health and environmental activities on a national and international level. This rekindled concerns in Canada about the potential health impacts of environmental projects. Yet, over a decade more passes for the next pragmatic step. In 2004, Health Canada introduced HIA and the 'determinants of health' as one of a series of efforts focused on consolidating health as a conscious concern for both human and environmental health sectors in Canada (Health Canada 2004). These determinants are still in use today in Canadian policy (e.g. Public Health Agency of Canada 2011) and variations of them including such considerations as income and employment, social status and social safety networks including connections to culture, levels of education, the conditions of the physical environment including ecosystem health, and others are considered in various bodies of scholarship (Health Canada 2004, Mikkonen and Raphael 2010, Public Health Agency of Canada 2012, Association of Faculties of Medicine of Canada n.d.).

Despite these milestones and landmark documents, notwithstanding the mandate of CEAA 2012 to protect human health, human health considerations remain poorly considered in the Canadian EA process. Following nearly half a century of effort in Canada, we see a public increasingly weary and resistant to large environmental projects, while more capable of addressing those concerns in legal spheres (e.g. Cryderman 2014). Despite these precedents and their limitations, the last half century has also been a source of learning. Analysis of literature from the last 15 years suggests that the challenge of HIA and the integration of health into the EA process are underpinned by certain dominant issues, which are presented below.

3.4.2 Issue 1: Government intervention

The need for government intervention was identified in almost half of the studies reviewed as an important factor for HIA implementation (Table 3-1). The literature (16% of the studies reviewed) describes the vital role of government leadership and the need for broad system support for the practice and implementation of HIA (O'Mullane and Quinlivan 2012). Spickett et al. (2011) calls on the government health sector to take the lead on advancing HIA. Another 12% of the studies reviewed argue for more than just support, but rather the need for formal institutionalization of HIA within government and/or other organizations as a key element to successful application (e.g. Winkler et al. 2013). Approximately, 20% of the studies reviewed call for yet more formal leadership by way of standard public policy (mandatory law) requiring the integration/application of HIA (Knutsson and Linell 2010, Drewry and Kwiatkowski 2015). They argue that the current legislation does not mandate the inclusion of health (and well-being) considerations to the extent necessary, contributing to a lack of incentive (Burdge 2002, Salay and Lincoln 2008) and resulting in ad hoc applications of HIA which are often triggered only by the efforts of public health advocates who recognize important health implications of proposals that would not otherwise be addressed (Lee et al. 2013). Many are urging federal health and environmental regulatory agencies to 'step up' and provide more formal guidance (Burdge 2002, Bhatia and Wernham 2008, Wright 2011) and rigorous legislation that creates legal obligations for HIA which mirror those of EAs (O'Neil and Solway 1990, Salay and Lincoln 2008). Reis et al. (2013) conclude that the need for an effective and robust science-policy interface has never been more pressing.

Conversely, YESA (2003) conducted a study of EA legislation in over thirty developing countries. They found, based on examining the definitions of what constitutes the 'environment', that the legislated scope in most of the countries examined is inclusive (Cited in Yap 2003). In the USA, Bhatia and Wernham (2008) found that case studies demonstrated the adequacy, scope and power of existing statutory requirements for health analysis within the EA. In Canada, there is no language in CEAA 2012 that would preclude an inclusive approach. Even where specific language is not present, the CEAA does give the government authority the discretion to consider "any other matter relevant" [s19.1(j)].

These opposing views of the adequacy of existing legislation may be bridged by the notion that although legislation is key (Lee et al. 2013), it is not alone sufficient to ensuring the inclusion of health considerations (Banken 2001, McCaig 2005). For example, McCaig (2005) reports that HIAs were often not undertaken, even in instances where the legislation appeared to require it. Thus, in many jurisdictions, more than a legislation gap there is a 'policy-action gap' (Kearns and Pursell 2011) – a gap in leadership and support needed to apply the existing legislation. Bhatia and Wernham (2008) contend that it is the responsibility of the 'actors' to recognize the legislation for its intended purpose and actualize it as a tool for public good. Harris-Roxas et al. (2012) produced a state of the art HIA report, together with nine members of the HIA Section of the IAIA, noting that the practice of HIA enjoys limited recognition and institutional support compared to some other forms of impact assessment. Furthermore, though government clearly plays a significant role in supporting the application of legislation, to effectively lead the process, government agencies need first to develop internal capacity and an adequate understanding of the tools and methodological applications of HIA (Bekker 2005, McCaig 2005, Noble and Bronson 2005, Harris and Spickett 2011).

3.4.3 Issue 2: Methodological and procedural limits

Nearly half of the studies reviewed identified methodological and procedural challenges as key issues to applying HIAs and integrating health considerations into the EA (Table 3-1). The issue is rooted in the many differing perspectives of health that have emerged from "different worlds" (researchers, policy makers, administrators, economists, social advocates, public health practitioners, epidemiologists, etc.) with differing languages, opposing frameworks and little consensus around the criteria by which to evaluate health issues in the decision-making process (Buffett et al. 2007). These variations in the political, socioeconomic and administrative settings in which health is addressed have led to significant variations in the design and applications of HIAs (Lee et al. 2013) and a multitude of approaches and methods to doing health impact studies (Briggs 2008, Mindell et al. 2008, Lee et al. 2013). Kemm (2005) writes, "no longer does one speak of the way of doing an HIA, but rather of many different ways".

Harris-Roxas and Harris (2011) contend that these different forms of HIA serve different purposes and are not necessarily in competition; rather they allow HIA to be responsive to a range of population health concerns and purposes. Conversely, the variations are also a source of confusion among proponents and the public in terms of which to use and what to expect, as well as the types of data collected and the methods used to collect it. Pope et al. (2013) argue that although there are substantial strengths in the plethora of assessments available, there is a lack of clarity with regards to how they fit together, generating a somewhat confusing picture. Without a clear methodological framework, the outcomes of HIAs are often determined at the discretion of those conducting HIAs rather than conforming to a particular standard (Lee et al. 2013) and the data acquired across case studies are often disjointed, highly contextualized and difficult to compare (see Erlanger et al. 2008, Mindell et al. 2008).

Aside from the challenge of navigating the plethora of differing HIA frameworks, those that are in place are characterized by significant technical and methodological gaps (Linzalone et al. 2014). HIA methodologies are largely not well-formulated (Cole et al. 2004), clear (Krieger et al. 2003) or comprehensive (Hebert et al. 2012). There is also a general lack of validated tools (Kraemer et al. 2014) associated with the methodologies. Existing tools are not well-developed, practical or adequately operational for comprehensive application (Huang 2012, Drewry and Kwiatkowski 2015). Lhachimi et al. (2010) investigated six generic models of HIA and found none to fulfill the criteria of being considered a standard HIA tool. The models were either too technically advanced for general application or oversimplified.

3.4.4 Issue 3: The challenge of quantitative analysis

Approximately 10% of the articles reviewed in this study identified quantification as a key means of improving the applications of HIA and the integration of health into the EA process. The focus of many HIAs is to assess multiple effects on human health, including social, economic and environmental effects. Thus, HIAs will often involve, and be pressed to integrate, diverse types of evidence collected by both qualitative and quantitative methods including, baseline health status and vulnerability data, empirical studies, original qualitative research (e.g. from focus groups, structured and unstructured interviews, and group or expert consensus processes), as well as, local knowledge from community organizations and residents (Bhatia and Seto 2011).

Integrating these diverse sources of data requires a certain degree of methodological rigor which quantification, in certain contexts, may provide. According to Bhatia and Seto (2011), quantification of health impacts can offer numerically significant thresholds, quantitative social and public health objectives, and economic valuation; thus allowing for an 'apples to apples' comparison among alternatives and, thus serving as an effective means of conveying the magnitude of population-level health impacts. Cole et al. (2005) contends that quantification in HIA can offer valuable information for decision-makers that is often not otherwise available. O'Connell (2009) supports the quantification process as a means of producing estimates for those factors that have a sufficient base of research. These arguments are supported by the growing number of quantitative HIAs being implemented (Mesa-Frias et al. 2014) and recognized as a means of gaining greater insight and understanding of health-related impacts (Thomas et al. 2014).

The EA process in Canada also has a tendency to focus on quantitative analyses, with a distinct preference for 'hard' physical data over 'soft' social data (Yap 2003). This has been referred to as the EA 'quantification syndrome' and can result in a focus on parameters that are easiest to measure, rather than those that are most important, and can produce results that are 'quantified but wrong' so long as they are not 'qualified and untestable' (Mayda 1996). The quantification syndrome is also criticized for 'driving out' other information, leaving regulators with 'number-hungry' analyses (Leape 1980) which can critically de-contextualize the risks being measured (Steinemann 2000).

In the context of the HIA, O'Connell (2009) cautions that quantified HIA is not an infallible process and argues for its appropriate application in order to avoid giving the "illusion of certainty that belies the complexity of the interactions involved (p306)". In practice there are no standard models (Lhachimi et al. 2010), few standardized tools (O'Connell 2009) and limited data (Mindell et al. 2008) for quantitative estimations of the full range health effects. The tools that are available are applicable to a limited range of policy and decision settings and their precision and validity is uncertain as are their methodological assumptions (Bhatia and Seto 2011). According to Cole and Fielding (2007), limited data on the effect estimates of interest and the baseline characteristics of the affected population make the purely 'quantitative/analytic approach' to HIA infeasible.

Thus, quantification in HIA has proven challenging, with one notable exception - the Human Health Risk Assessment (HHRA) and Ecological Risk Assessment (ERA). It is noteworthy that HHRA and ERA are risk assessment frameworks and do not reflect the broader implications of health impacts assessments (Briggs 2008). Furthermore, they typically focus narrowly on single toxic endpoints of single chemicals in a single medium from a single source (WHO 2013), while many of the risks facing society are systemic in nature –complex risks, set within wider social, economic and environmental contexts (Briggs 2008). Thus, many national and international organizations call for much more holistic approaches, not only considering multi-chemical, multimedia, multi-route and multi-species exposures (WHO 2013), but also the effects on broader ecological health, as well as mental and social well-being (WHO 2006). Notwithstanding these limitations, the drive towards quantification has given HHRA and ERA relatively broad acceptance and usage in the EA community and, in certain jurisdictional contexts, these approaches have become the dominant approach to assessment of health considerations in the EA process.

The tendency (or bias) towards quantitative HHRA and ERA approaches illustrates two related challenges. On one hand, there is a need to improve quantitative approaches to assessing health impacts for the purpose of integration into the EA process, where appropriate. On the other hand, there is a need for skills and approaches to complement quantitative techniques with qualitative approaches if, and especially when, quantification is insufficient or inappropriate. These challenges link closely with issues 4 and 5 in terms of the expertise required to handle integrative approaches and the complexity of the analysis needed to integrate qualitative and quantitative techniques.

3.4.5 Issue 4: The gap in expertise and collaboration among the actors

Given the complexities involved in conducting assessments of health impacts, considerable expertise is required. Yet EAs are often conducted by consultants who may not be well-versed with respect to health effects, the determinants of health, social and economic theory, or be adequately trained in SIA (or HIA) methodology (Yap 2003, Harris et al. 2009, Esteves et al. 2013). This can serve as a bias in favor of the familiar terrain of quantitative HHRA, as discussed in Issue 3 above. Over a quarter of the articles reviewed reported limitations in expertise as a key issue to conducting HIAs and integrating health into the EA (see Table 3-1). For example, in Quebec, Canada, lack of knowledge related to the determinants of health and impact assessment was found to be the main obstacle to implementing HIAs within the EA process (Mendell 2010). Morrison-Saunders and Therivel (2006) argue that additional thought needs to be given to who is undertaking the HIA and the integration of health considerations into the EA process.

With limitations to expertise and expert experience, results from past EA studies, where human health was considered in the EA process, may be a source of learning and capacity building. However, these are rare as HIA has not been widely implemented (Lee et al. 2013). In addition, Yap (2003) notes that many existing studies are considered proprietary and the details of the methods and designs are inaccessible to either the public or decision-makers. These data-communication gaps may exist between the many actors involved as an intersectoral issue (Lee et al. 2013). The need for intersectoral and disciplinary collaboration and coordination was identified in over a quarter of the studies reviewed as important to applying and integrating HIA into the EA process (Table 3-1).

3.4.6 Issue 5: Analytic complexity

Analytic complexity across health and environmental boundaries, especially in the wider social, economic and environmental contexts, is another major challenge to integrating health impacts into the EA (Briggs 2008, Naddeo et al. 2013) and is cited as a key issue to applying and integrating health considerations in 10% of the articles reviewed. Causal pathways linking environmental impacts to the determinants of health and to downstream health outcomes are typically long and complex, often involving multiple intervening and potentially interacting factors along the way (Birley 2002, Braveman et al. 2011). This complexity makes it exceedingly difficult to predict or quantify project long-term impacts on health with any degree of accuracy (Noble and Bronson 2005).

Waltner-Toews (2011) argues for the essential need of systems approaches to address complex issues at the interface of human and ecosystem health. Yet, the current models of environmental management in the Canadian context may not lend themselves well to the advancement of systems approaches. For example, the status quo EA typically focuses on isolated components of the ecosystem, such as fish and fish habitat, aquatic species and migratory birds; aligned with the language of CEAA 2012 which specifically references the Fisheries Act, Species at Risk Act, and Migratory Birds Convention Act, respectively [s5.1a]. The management of species including fish, wildlife and forests, as well as the protection of the air, water and soil in those systems, are typically apportioned and delegated to a variety of distinct government agencies, such as Fisheries and Oceans

Canada, the Ministry of Environment and the Ministry of Forests, Lands and Natural Resource Operations (see BC Government 2013, DFO 2014d). This division of responsibility among agencies, with arguably minimal cross-fertilization of capacity, is maintained by a number of factors including the Constitution Act of Canada (1867), as well as, certain pragmatic issues stemming from the single-species focus of Canadian natural resources industries.

3.4.7 Issue 6: Project pace and the need for efficiency

The objective of the EA process is to "encourage federal authorities to take actions in a manner that promotes sustainable development in order to achieve or maintain a healthy environment and a healthy economy" (Government of Canada 2014). The latter 'economic' objective can overshadow other objectives, especially under external influences such as Canadian economic development policies and federal budget obligations. Furthermore, the pace of economic development can place considerable pressure on governments to overcome the EAs "extended regulatory approval process" (Berkow 2014), pushing the EA to achieve greater 'efficiency'. But the pursuit of efficiency may be occurring at the cost of marginalizing health considerations.

To illustrate, the federal government originally established the Environmental Assessment Review Process (EARP) in 1973 to help cabinet assess the environmental effects of federal decisions (Hopkins-Utter 2012). But the EARP was simply a cabinet policy with no compliance or enforcement mechanisms (Meadows n.d.). Thus, neither environmental nor human health concerns were consistently or adequately examined within projects. The EARP functioned this way for over a decade (1973-1984) at which time cabinet elected to pass it as a Guideline Order, thereafter referred to as the EARPGO. Inadvertently, the non-discretionary language of the EARPGO (e.g. using terms such as "the Minster shall") made it vulnerable to court challenge. A 1989 landmark federal court decision (Rafferty-Alameda Dam Project Inc v. Saskatchewan) (Harrison 1994) resulted in a ruling that the EARPGO constituted legally-binding environmental assessment obligations for federal authorities (Meadows n.d.). Thus, the EARPGO was prodded into a law of general application, creating space for further court challenges (Sadar and Stolte 1994). Indeed, the lack of consistent practice and application under the EARPGO led to a series of court challenges that ultimately caused the enactment of the 1995 *Canadian Environmental Assessment Act* (CEAA) (Horvath and Barnes 2013); thus becoming an important part of Canadian legislation.

The CEAA was also found to be cumbersome and inefficient, resulting in a ministerial review in 2001 followed by amendments in 2003 (Hopkins-Utter 2012). The amendments focused on new mechanisms of support to help proponents more efficiently navigate the EA terrain and gave new powers to the federal Minister of the Environment to establish the scope of projects (i.e. the content and extent of matters to be covered). These new mechanisms of efficiency became the impetus for a wave of new project approvals in Canada, while health considerations received little attention, despite the introduction of the

Canadian HIA Handbook a year later (Health Canada 2004). Additionally, the CEAA was mandated to undergo a parliamentary review within seven years (Becklumb and Williams 2011, Hopkins-Utter 2012). The seven year review in 2010 resulted in a few additional amendments but was followed by a report from the Standing Committee on Environment and Sustainable Development that found significant inefficiencies triggering its repeal in 2012 and replacement with CEAA 2012 (Government of Canada 2012, Hopkins-Utter 2012).

CEAA 2012 seeks further efficiency through various mechanisms.

1) It exempts EAs for the majority of projects in federal jurisdiction that are not included in the *regulations* (Horvath and Barnes 2013); the regulations list all 'major' project types deemed likely to have significant adverse environmental effects. The Minister of Environment also has the authority to remove project types from the list [s84a]. To what extent human health is considered when projects are exempted from the EA process will become clear with time. In empirical terms, when CEAA 2012 came into force in July 2012, approximately 3000 EAs were immediately cancelled, many with no corresponding provincial assessments (Gage 2013). Thus, where the federal government previously reviewed 4000 to 6000 projects per year, it is projected to review only 20 to 30 (Scott 2012).

2) CEAA 2012 allows for the provinces to request that their provincial EAs be a substitute for [s32.1] or deemed equivalent to [s37.1] an EA otherwise falling under the authority of the Canadian Environmental Assessment Agency. In a substitution, the federal

government relies on the provincial EA, but makes the final decision itself (i.e. the provincial EA fulfills the requirements of CEAA 2012). In the case of an equivalency, the federal government relies entirely upon the provincial EA, including the ultimate decision, exempting the project from CEAA 2012 (Heelan 2013). In both cases, the provincial EA must consider the same health impacts as would a federal assessment [s34.1a]. However, the degree to which the different regulatory mandates, jurisdictional powers, expertise and capacity of provincial authorities will affect considerations of human health impacts will become apparent in the field of practice.

3) CEAA 2012 also seeks efficiency through streamlining public participation and imposing mandatory timelines on public comment periods and reviews. For example, it gives the responsible authority or review panel discretionary power to determine participation and, in certain instances, to limit that participation to 'interested parties' (Courtney 2012, Hopkins-Utter 2012). How these limitations to public input will affect the extent to which authorities understand and appreciate the potential for projects to impact human health and well-being will become apparent with time.

In summary, the Canadian EA process was designed to serve as an important decision-making tool, to balance economic development with important human health and environmental concerns in a process that engages both technical experts and the public, and to draw together available research and analysis with the goal of maximizing benefit, minimizing impact and providing follow-up monitoring to help manage risks and uncertainties (Health Canada 2004, CIDA 2013). The drive towards efficiency may, however, serve to impede the process from achieving these objectives, instead marginalizing human health considerations and ultimately galvanizing environmental groups who argue that the Canadian EA process is focused on economic interests (Hopkins-Utter 2012). Nevertheless, if health considerations are to find pragmatic application in the EA process, the issue of efficiency is an important consideration.

3.5 Discussions and Recommendations

The key issues related to applying HIAs and integrating health considerations into the EA process were presented above. A closer examination of those issues reveals additional insights for addressing them. Those insights may also be examined within the framework of the CEAA 2012 as the basis for enhancing the status quo EA process with respect to its integration of health considerations. The status quo EA process, as governed by CEAA 2012, is presented in Figure 3-2.

The figure demonstrates the typical pathway of the status quo EA process beginning with a project description [s8] prepared by the proponent, allowing the CEA to determine whether the project will require an EA [s10] and whether it or another government agency should be the responsible authority [s14.4a-d] and [s15a-d]. The EA process could alternatively be assigned to (1) a Review Panel [s38.1 & 2], (2) upon request of a Canadian Province, enter into a substitution agreement (i.e. the EA is conducted by the Province, but the federal government has the ultimate decision-making power), or (3) enter into an equivalency agreement (i.e. the Provincial EA and its decisions replace the CEAA 2012 federal process [s32.1]). If an EA is required, a scoping of the project proposal [s19.2] is conducted to determine the potential environmental effects. Effects may include impact to species and habitats [s5.1a], to the health and well-being of people [s5.1b & c] or to other relevant matters [s19.1j]. Driven by the most prevalent language of CEAA 2012 and a long list of precedents set by past EAs, the health effects addressed by the status quo application of the legislation will often involve applications of HHRA and health and safety studies, but exclude consideration of broader health impacts. Mitigation and planning are proposed to help manage those effects within allowable limits and a proposal is returned to the responsible authority and Minister or Government in Council for a decision process, as shown.

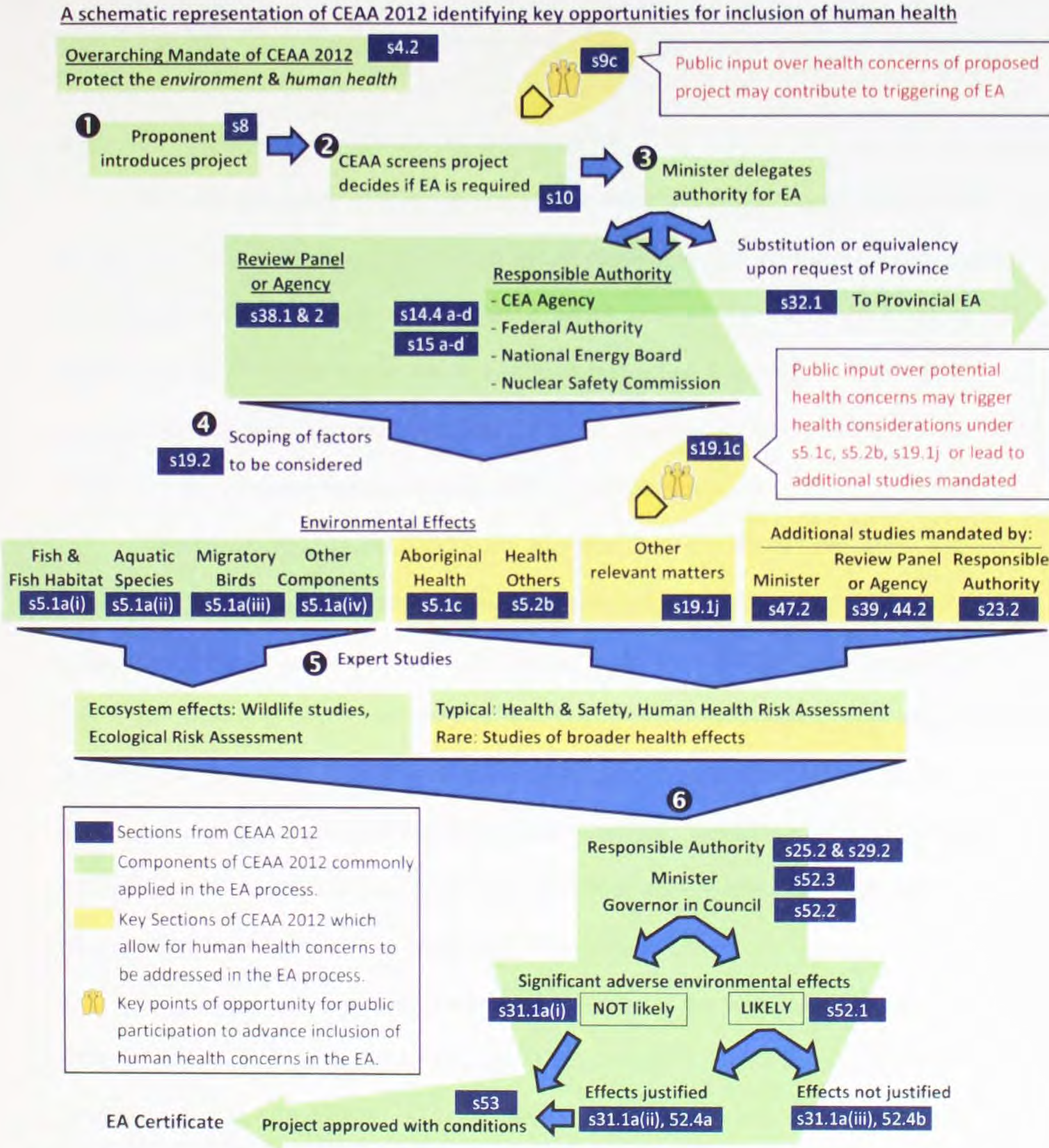


Figure 3-2. The Canadian Environmental Assessment Act (Government of Canada 2012) defines a process with six major steps: (1) proposal and description of the project by a proponent; (2) screening by the CEAA; (3) delegation of authority for the review; (4) scoping to identify the factors to be considered; (5) expert studies; and (6) decision on the project's fate. Within these steps, the Canadian Environmental Assessment Act, 2012 identifies a series of discrete points where health considerations may be addressed (yellow shading), though not necessarily triggered in all environmental assessment processes. A modification of this figure was published in Parkes (2015).

3.5.1 The status quo EA and the need for change

The Canadian experience would suggest that without a deliberate effort to reshape the process, EAs will continue to follow the typical pathway shown in Figure 3-2, focusing on biophysical impacts with some consideration of health and safety and HHRA, but leaving broader considerations to human health largely unaccounted. A growing body of recent examples of EA applications illustrates that the status quo practice of EA, deficient in its considerations of human health, is facing significant challenges.

Precedent-setting court rulings in the Supreme Court of Canada have significant implications for proponents moving forward with projects requiring an EA on *untreatied* (Aboriginal) lands. Important examples include: Delgamuukw v. British Columbia (1997) which confers Aboriginal peoples with the right to possess ancestral lands; Haida Nation v. British Columbia (2004) which requires government, and by extension proponents, to engage in a meaningful process of consultation in good faith with First Nations; Tsilhqot'in Nation v. British Columbia (2014) which recognizes First Nations title claims over lands they historically occupied, continually inhabited and exclusively use.

These rulings are emerging as a force of evolution to the status quo EA process, forcing proponents to give more serious attention to issues of human health and well-being. In addition to First Nations rights and title, public pressure across affected communities for proponents and government to appropriately identify and mitigate the health risks of proposed projects is also proving to be a significant consideration in the political arena (e.g. Pluim 2012). Government agencies and proponents choosing to ignore or allay these considerations run the risk of encountering significant delays and legal challenges.

The Enbridge Northern Gateway Project (ENGP) is a good illustration. The ENGP was a proposed $6.5 billion project involving construction of 1,177 km of pipeline to carry diluted bitumen from the Athabasca oil sands in Alberta (Northern Gateway 2014) across six or more indigenous nations in northern BC (Alberta Oil 2014). The proposed pipeline would cross provincial boundaries and, therefore, fall under federal jurisdiction; though provincial government support was also required in the issuing of construction permits. The proponent managed to navigate the federal EA process after $400 million of investment and four years of effort (Vieira and Dawson 2014). The federal EA office issued its Decision Statement on June 17, 2014 finding the project "likely to cause significant adverse environmental effects...to certain populations of woodland caribou and grizzly bear", but that the "adverse environmental effects... are justified in the circumstances" (Canadian Environmental Assessment Agency 2014). The proponent was granted approval with the stipulation that it meet 209 conditions related to those effects (Canadian Environmental Assessment Agency 2014). An additional five conditions were also set by the provincial government (Northern Gateway 2014).

Despite achieving EA approval, the announcement immediately triggered a number of significant legal challenges. For example, several BC First Nations groups filed appeals with the Federal Court seeking to overturn the panel recommendation (Laanela 2014a), BC Nature issued notice of a lawsuit to challenge the federal Cabinet decision (BC Nature 2014) and the northern municipalities of Terrace, Prince Rupert and Smithers voted to oppose the project (Moore 2014). These examples serve to illustrate the complexities that can arise in the EA process when the broad range of social and ecological effects introduced by projects is inadequately considered. They are also a testament to the potential inefficiencies that could result, contrary to the purpose of the many amendments that the CEAA has undergone in decades past.

3.5.2 Recommendations and alternatives

This review has identified some of the key issues that may be responsible for the poor integration of human health into the EA process and contextualized them to the status quo EA process governed by the CEAA 2012 legislation. Based on the findings a number of recommendations are proposed in order to address the issues. Table 3-2 summarizes the issues identified and the opportunities to address them.

Table 3-2. A summary of the key issues responsible for the poor integration of human health considerations into the Canadian EA process and recommended solutions within the context of the Canadian Environmental Assessment Act (CEAA 2012).

Issues	Recommendations
1. Need for government intervention to advance applications of the legislation (48%)[1]	The protection of human health (a core mandate of CEAA 2012) need also become a core mandate of the EA process. Application of HIA within the EA process requires focused promotion by government (16%) and other (4%) agencies.
2. Need for effective and appropriate standard approaches and methods (45%) to measuring health impacts	Disciplinary and intersectoral coordination (26%) may be a key factor to developing and standardizing integrated HIA methodologies and tools, together with standardized data measurement protocols.
3. Need for better quantitative/ qualitative data integration (10%)	
4. Need for expertise (26%)	Disciplinary and intersectoral collaboration (26%) and public participation (16%) in a meaningful process of engagement can bring significant new capacity to the process.
5. Need to address analytic and human-environmental health cross-linking complexity (10%)	A well-conceived collaborative structure may better reflect the complex systems thinking needed for deriving systems solutions.
6. Demand for efficiency (10%)	A number of established and effective information gathering methods that do not entail excessive cost or time may be applied. Significant new efficiencies may be gained through reducing legal challenges to EA decisions, as well as, long-term medical costs of poor health integration and planning.

[1] (%) - percent of studies identifying the issue as important

3.5.3 Issue 1: Government intervention

The task of integrating health considerations into the EA process requires a dedication of the actors involved to achieving the spirit of the CEAA 2012 mandate: to *promote sustainable development* while maintaining a *healthy environment* and *protecting human health*. Achieving this mandate requires both a champion and a structure. With respect to a champion, Harris and Spickett (2011) contend that progress on the health/EA integration front rests on developing broad system support for HIA across government, led by the health sector. In the Canadian context, this would involve Health Canada federally, Ministries of Health provincially, as well as, health authorities in local and regional contexts. Cole et al. (2004) found HIA to be most successfully applied in those places where governments made a commitment to promoting the process among the actors across sectors.

With respect to structure, the efforts of the champion agency may be more effective if structured in the framework of collaboration among the actors (a recommendation put forward by 26% of the studies reviewed). Bhatia and Wernham (2008) and others (Carmichael et al. 2013, Linzalone et al. 2014) conclude that the key issues to integration need be addressed through a disbanding of existing organisational and professional silos, followed by greater collaboration among the institutions responsible for EAs, as well as, public health institutions. Thus, as the lead agency in the health/EA integration process, the selected government health sector agency would then be responsible for coordination of the actors, and to lead the collaborative process towards an integrated outcome.

3.5.4 Issue 2: Methodological and procedural limits

Development of tools and approaches for integrated analysis are key requisites to advancement of the health/EA integration challenge. The collaborative process discussed above can serve as a valuable starting point for identifying key research priorities, such as integrated health studies focused on the environmental determinants of health, systems research, cumulative effects and others. The diversity of themes involved reinforces the need for collaboration across sectors including health, environment, policy and legislation.

3.5.5 Issue 3: The challenge of quantitative analysis

Data collected by differing methodologies from diverse disciplines can face certain compatibility issues and may, therefore, be difficult to integrate (e.g. the quantitative/qualitative data integration challenge). Quantifying data (e.g. health and environmental data) in a standardized methodological framework may present certain opportunities for improved integration, as observed in the wide usage and general acceptance of HHRA and ERA discussed above. The need for improvements to both the tools and approaches to quantifying health impacts is recognized. However, as noted above, quantification may be inappropriate in certain contexts and, in many instances, a combination of quantitative and qualitative results is likely. Thus, arguably more urgent than quantification, is the need for approaches to interpreting qualitative/quantitative results in an integrative manner.

3.5.6 Issue 4: The gap in expertise and collaboration

Given the breadth of complex issues involved in the practice of EA and HIA, it is unreasonable to expect a single expert with adequate disciplinary knowledge and experience. Instead, expertise is likely to be found through expert collaboration in a well-conceived collaborative structure. In the context of the Canadian EA process, the challenge of health integration could benefit greatly from such a structure. The collaborative structure should allow for effective engagement among the actors and their relevant knowledge strengths (Fischer et al. 2010). See Boelen (2000), Brown (2007), Pohl and Hadorn (2007) and Mahboubi et al. (2015) for a discussion of knowledge strengths.

It is also important to note that the *approach* to engagement must surpass traditional multi-disciplinary practices which are ideally suited to addressing narrow concerns and not generally effective at creating new knowledge. The approach to engagement should evolve to become aligned with the spirit of the (2004) Supreme Court ruling (Haida Nation v. British Columbia) which speaks to a *meaningful process of consultation*. Brown (2007) argues that the core prerequisite of the collective approach is the establishment of a 'shared focus' among participants resulting in a 'holistic understanding', but contends that this is a condition rarely achieved under current management conditions. Diduck and Mitchell (2003) describe the status quo EA process as legitimating rather than participatory, empowering or equitable. A more sophisticated *interdisciplinary*, or *transdisciplinary*, approach to engagement may be a more ideal setting for this purpose (Parkes et al. 2005, see examples from ecohealth studies in Charron 2012).

3.5.7 Issue 5: Analytic complexity

The complexity of integrating health concerns into the EA process is not easily resolved. The complexity can, however, be partially managed and alleviated by the collaborative approach discussed above. First, the collaborative framework could serve as an effective means of circumventing the typical piece-meal or fragmented examination of environmental factors; an approach inconsistent with the principles of systems thinking as described by Waltner-Toews (2011). Second, the collaborative approach could help facilitate the complex process of interpreting and integrating diverse qualitative and quantitative health and environmental data sources, avoiding the over-extension of data beyond their limits; an approach that can belie the complexity of the interactions involved (O'Connell 2009). Third, collaboration among the actors can serve to create increased operational complexity. Yet, this is a condition that may be appropriately reflective of the natural systems within which the key issues and challenges identified are nested (see Parkes et al. 2008, Waltner-Toews 2011, Charron 2012, Fèvre et al. 2013 for discussions on ecohealth as a health-centered systems-based approach based on disciplinary and sectoral collaboration for deriving complex solutions to complex problems). Finally, the management of complexity could benefit from approaches that stage the analysis into manageable units. See section 6.6 (page 191) for a discussion on staging complexity.

3.5.8 Issue 6: Project pace and the need for efficiency

Increasing the breadth or depth of considerations of health within the EA process will likely correspond to an increase in the time and resources required. However, there are a number of established and effective information gathering methods that do not entail excessive cost or time. These include reviews of primary and secondary literature and case studies of similar projects (Yap 2003), use of rapid appraisals (Kosa et al. 2007, Mindell et al. 2010), key informant interviews (Elliott and Williams 2004, Ahmad et al. 2008), focus groups, workshops (Bond et al. 2013) (Kearney 2004), mini surveys in the field and the limited and judicious use of experts through Delphi and nominal group techniques (Yap 2003). Where several techniques are used, the information collected may be triangulated for improved results (Yap 2003).

It is also noteworthy that increases in cost and time incurred to more fully address health in the EA process could potentially be offset by new efficiencies gained. For example, court challenges to the results of the ENGP EA process, discussed above, illustrate the extraordinary costs and time delays that can result from inadequate considerations being given to broad project-related health effects. Any measures that help to reduce such legal conflicts would also help to increase efficiency.

Finally, improving the integration of health considerations into the EA process could also help to avert many long-term medical costs within the community over the life of the project. In one example, Hutton and Haller (2004) reports a minimum 300% net benefit in medical cost savings resulting from water and sanitation interventions. Yet, the true cost benefits could in fact be significantly higher, were a broader range of health considerations included in the analysis. Determining those additional cost benefits is, however, a difficult task and in many jurisdictions the basic environment and health data may be missing or incomplete for such an assessment (WHO 2015c).

3.6 Conclusions

Following nearly a half century of effort in Canada to integrate health concerns into the EA process, we find limited progress in practice. Numerous issues are reported in the literature as responsible including, the need for government intervention, gaps in methodology and tools, limitations of capacity and expertise, poor intersectoral, disciplinary and public collaboration/participation, challenges of data quantification and analytic complexity, and the need for process efficiency. The status quo application of the CEAA 2012 has not been effective at addressing these issues. A number of recommendations are made as a starting point for improved integration. First, a commitment by the actors involved to adhere to the core mandate of the CEAA 2012, which includes the protection of human health. Second, the achievement of intersectoral, disciplinary and public collaboration, led by government, ideally the health sector. In the wake of past efforts and important milestones, the Canadian context provides a timely opportunity for a new era of leadership and innovation at the interface of health and environmental assessment.

Chapter Four. A spatial approach for the integration of economic, social, ecological and marine protection legislation data for marine resource management

4.1 Introduction

Coastal marine ecosystems, home to 44% of the world's population (UNEP 2010), provide humans with important economic, ecological and social benefits. Coastal areas are among the most productive and biologically diverse ecosystems, providing habitat for approximately 80% of the 13,200 known species of marine fish (UN-Oceans 2015). Oceans play a significant role in maintaining ecosystem functions which yield valuable current and future sources of food for humankind (UN-Oceans 2015). In 2012, an estimated 80 and 90 million tonnes of seafood were harvested through global marine fishing and aquaculture, respectively (FAO 2014). These accounted for almost 17% of the global intake of animal protein (often the only means of subsistence for many coastal populations) and supported the livelihoods of 10-12% of the world's population, providing a wide range of economic opportunities (FAO 2014).

Yet, the oceans of the world remain heavily impacted by pollution, overfishing, introduced species, habitat loss and species extinction (UNDP 2015). An estimated one third of coastal regions face a high risk of degradation (World Oceans Network 2013), while half of global fish stocks are fully exploited, and a quarter are depleted, over-exploited or recovering from depletion (UNDP 2015). In Canada, 2009 fish catches were the lowest on record since 1984 (Auditor General of Canada 2012).

A range of factors are found to be responsible for the adverse impacts (see NOAA 2006). Among them is fragmented management by the actors (WHO 2015b) including fisheries, private industry, water authorities, local government, housing authorities and others. This fragmentation can result in poorly planned and managed coastal development, ultimately leading to ecosystem degradation and, thus, the loss of livelihoods (UNDP 2015). Public concerns elicited by such losses are focussed on areas associated with important economic, environmental or social values. Clayoquot Sound and the Great Bear Rainforest are important examples in Canada (Low and Shaw 2012). Responding to these concerns, governments often intervene by way of establishing guidelines and legislation (Field and Olewiler 2011); measures which are themselves often created under conditions of uncertainty and narrow criteria (e.g. Gibson 2012).

The range of difficulties encountered when considering economic, ecological and social ecosystem values in local or regional management planning is not surprising due to a number of factors. First, social-ecological values are inherently difficult to measure (see National Research Council 2004, Simpson 2011). Second, there are significant economic and biophysical data gaps, and an even greater scarcity of social data. Third, much of the data are too coarse, occurring over broad ecosystem scales, rendering the results of limited use to detailed local planning.

Despite the challenges of incorporating social-ecological-economic data into analyses, the development of appropriate approaches to integrating them may be a core requisite to improving ocean management. Thus, the objectives of this study were (1) to acquire and examine several commonly available economic, ecological, social and legislated datasets pertaining to the marine environment, (2) to conduct a detailed spatial analysis of the selected dataset in order to detect marine spaces considered important for each theme, and (3) to conduct an integrated analysis of the combined data in order to gain a more complete picture of the spatial distribution of important social-ecological-economic-legislated spaces. Focusing on an area of the Pacific North Coast of Canada as a case study, this research proposes an approach for analyzing various categories of marine spatial data to identify important social-ecological-economic locations in the coastal marine ecosystem as a tool for coastal and marine managers and planners.

4.2 Background

Canada is an ocean nation whose economy, environment and social fabric are considered inextricably linked to the oceans and their resources (DFO 2002). It has the world's longest coastline, stretching over 243,000 km along three oceans and the second largest exclusive economic zone in the world (DFO 2005, 2010). **Economically**, Canadian ocean industries, including commercial and recreational fishing, seafood production, energy production, transportation, shipping and tourism collectively generate 329,000 jobs and contribute $39 billion a year to Canada's gross domestic product (2008 data) (DFO 2014c). **Ecologically**, oceans provide important regulating services to climate, waste treatment, disease and natural hazards, as well as the supporting services of nutrient cycling, the building of biologically mediated habitats and primary production (Molnar et al. 2009). **Socially**, many coastal communities throughout Canada rely on the oceans and their resources to meet nutritional needs, advance scientific learning, as a source of spiritual and inspirational enrichment, and for the practice of traditional cultures (Assembly of First Nations 2003, Molnar et al. 2009, CRIFC 2010b, Chan et al. 2011).

The systems approach to sustainability proposed in the early works of Passet (1979) characterises and respects the co-evolution of these three elements (i.e. economic, social and environmental) as three spheres of a dynamic 'triple bottom line' required for achieving sustainability (O'Connor 2006). The triple bottom line is described as a complex quality criterion. It affirms that economic activity, while having its specific imperatives (innovation, profits, etc.), must nonetheless be in the service of the wider social sphere, and that the biophysical environment does hold the potentialities for sustaining economy and society (O'Connor 2006). O'Connor (2006) also argues for the demarcation of a fourth fundamental category of organisation, the political sphere, whose role is regulation of the economic and social spheres and, thus, of relations with the environmental sphere. The four spheres are collectively referred to as the 'tetrahedral model' of sustainability.

The region of ocean along the north and central coast of BC, Canada, referred to as the Pacific North Coast Integrated Management Area (PNCIMA), is of particularly high ecological, social and economic importance (Molnar et al. 2009). Proposed projects involving development and use of the northern portion of the PNCIMA have faced much contention. Examples include the Enbridge Northern Gateway Project (BC Nature 2014, Coates 2014, Moore 2014), liquefied natural gas extraction, piping, processing and shipping (The Council of Canadians 2014) and others (see examples in Carleton Ray and McCormick-Ray 2013). Notwithstanding the expected economic benefits of these proposed developments, there are also a range of conflicts that have arisen in the public sphere focussed on the potential for adverse impacts to the social-ecological system.

The conflicts are, in part, rooted in a messy convolution of the four 'spheres'. Thus, an approach to investigation is required that allows for a disentangled spatial analysis of each of the individual spheres, followed by an integrated analysis of the collective spheres. de Jonge et al. (2012) argue that the need for more comprehensive environmental accounting frameworks has never been greater. To achieve the objective of individual and integrated analyses, spatial data pertaining to each of the spheres is required. Also required is an appropriate approach to the integrated analysis and interpretation of the combined data.

4.2.1 Economic data

Several economic valuations of marine ecosystem services were found in the literature (see Philcox 2007). For example, GSGislason & Associates Ltd. (2007) conducted an assessment for all of BC, Molnar et al. (2012) focused on the BC Lower Mainland, and Molnar et al. (2009) examined the PNCIMA. These reports provide summary results for relatively large areas. Despite the utility of these studies for certain planning purposes the scales of analysis were too course for the local context of this research (see Methods below).

Fisheries and Oceans Canada (DFO) fisheries harvest data (DFO 2013e) are among the few finer-scaled data available for the ocean region of BC. The data are collected by a variety of methods[1]. Despite considerable improvements made to the monitoring technologies employed in the Canadian Pacific fishery, the DFO (2013b) contends that there are still important deficiencies in information gathering with respect to coverage of the fisheries, missing or unreliable data (particularly on by-catch and discards), reporting delays and other issues. Nevertheless, the monitoring and resulting data collected have enabled the Pacific fishery full market access, eco-certification and global recognition of sustainable fisheries management (Fraser 2008).

The DFO database reports the quantity of salmon (5 species) harvested from each pacific fisheries management areas (PFMA), as well as shellfish and groundfish catches for all of BC. The data are reported by weight (kg) and landed value. Landed value is considered the price awarded to fish harvesters for their catch, compared to the wholesale value which is approximately twice the landed value (CCPFH 2009). Variations of the data are available through the DFO Fisheries Statistics department which spatially references shellfish and groundfish harvest data to a series of 4 km species grids. Species grids (i.e. 16 km^2 cells) report the total quantity of a species harvested from that cell over a several year period.

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[1] Examples: dockside monitoring for geoduck, sea urchins, sea Cucumber and certain pelagic fish; independent on-grounds monitors for certain dive fisheries; GPS tracking, hydraulic sensor and video camera monitoring for the crab fishery; and vessel monitoring systems and electronic logs (e-logs) monitoring for prawns and salmon (DFO 2013b).

Recreational fishing is another significant component of the British Columbia marine economy. Statistics Canada reports $467 million of direct economic output, $135 million of GDP, 4200 direct fulltime equivalent jobs and $88.9 million of employment income generated from the tidal sport fishery in 2005 (BCMCA 2015). However, the sport fishery data available from the BCMCA, including anadromous fish, crabs, groundfish and prawns, identify important locations for sport fishing activities, rating the relative importance of those locations on a scale of 1 to 5, but do not provide any means of spatially distributing the Provincial fishing revenues to those locations. Thus, sport fishing economic data could not be integrated into this analysis.

4.2.2 Biological and ecological data

Ecological approaches to quantifying the importance of natural spaces remain a challenge. They can vary from measuring the number of species present, particularly rare or endangered species, to calculating the biodiversity of a site. Boyd and Wainger (2003) and others (de Jonge et al. 2012) argue, however, that such measurements do not provide an adequate test of ecosystem value and that there are, in fact, no measures that might be considered a definitive measure of ecosystem quality. Other works focused on ecological indicators as a measure of ecosystem health (TEEB 2011, Layke et al. 2012). Yet, indicators often cover only a minority of ecosystem services and there are still too few data-points per service to conduct statistically meaningful analyses (van der Ploeg et al. 2010, van Reeth 2014). The challenges are particularly pronounced on ocean systems where extensive data gaps, especially in biologically complex near shore regions, have hampered the development of appropriate models; leaving no integrated, whole-ecosystem approach to identifying a comprehensive set of important marine areas (Gregr et al. 2012).

In light of the 2004 convention on biological diversity (CBD) agreement, committing signatory nations to reduce the loss of global biodiversity by developing marine protected areas (Agostini et al. 2008), various approaches were proposed for prioritising marine areas. The criteria considered by these approaches were evaluated and reconciled by Gregr et al. (2012)[1] to produce a final set of evaluation criteria for selecting ecologically and biologically significant areas[2]. Gregr et al. (2012) argue that the final criteria (with the exception of considerations of human impacts and management) are fully encompassing.

These criteria can be used to evaluate a variety of ocean classification systems in use worldwide. Gregr et al. (2012) evaluated 13 of the most popular ocean classification systems, giving the EBSA classification of the PNCIMA conducted by Clarke and Jamieson (2006b) the highest score. The Clarke and Jamieson classification was based on the DFO EBSA approach[3] and implemented in two phases.

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[1] Three such approaches include (1) the DFO (2004) criteria for selecting ecologically and biologically significant areas (EBSAs) developed in response to the Canada Oceans Act, (2) the CBD (2006) criteria for selecting EBSAs and (3) the criteria established by the Food and Agriculture Organization of the United Nations (FAO) (ICES 2008) for selecting vulnerable marine ecosystems.

[2] The final evaluation criteria were based on seven criteria: (1) the suitability of the scale (resolution and extent) and attributes used by the framework to define EBSAs, (2) the feasibility of the framework to interface with available data, (3) the ease of reproducibility according to explicitly outlined methods, (4) the effectiveness of the framework for capturing both physiographic and zoological features, thus demonstrating its biological validity), (5) the capability of the framework to adjust its boundaries to reflect seasonal variations and inter-annual patterns, (6) parsimonious (i.e. ease of application) and (7) applicable (i.e. the method can be applied ocean-wide).

[3] The DFO approach to EBSA selection applied by Clarke and Jamieson (2006b) is based on five criteria including uniqueness, aggregation, fitness consequences, naturalness and resilience.

In Phase I, a modified Delphic process involving regional science experts was used to identify and rank important areas (IAs) for each species and habitat feature (i.e. polygons worthy of enhanced protection based on five biological/ecological criteria) (Clarke and Jamieson 2006a). This resulted in 132 species-related IAs organized in 40 thematic layers. Experts also scored each location with respect to its contribution to each of the five criteria - a score of 1 to 10 yielding a maximum score of 50 points per polygon. In Phase II, IAs were selected based on unique physical features[1]. Fifteen such features were identified with a 73% correlation with Phase I IAs. Phase II IAs not overlapping Phase I IAs were removed leaving the final proposed EBSAs, collectively covering an area of 45,182 km^2 (44.3% of the PNCIMA) (Clarke and Jamieson 2006b).

4.2.3 Social data

There are a number of non-monetary social valuation approaches proposed in the literature. Many are qualitative in nature (e.g. surveys, interviews, focus groups, citizens juries, participatory or rapid rural appraisal and Delphi panels), while others are quantitative (e.g. preference assessment, time use studies, Q-methodology) (see Hadley et al. 2011, Christie et al. 2012). Kelemen et al. (2014) contend that the field of social valuation remains unsettled and approaches are not well formalized. They recommend a 'learning by doing' approach to test and improve the applicability of non-monetary methods in different institutional and socio-political contexts.

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[1] Three categories of unique physical features were considered (1) oceanographic features, (2) bottleneck areas (i.e. bathymetric and topographic features constraining species distributions to specific areas) and (3) sponge reefs (bioherms).

In general, spatial social data related to ecosystem services are rare. Those that are collected are typically done using qualitative approaches (see Hadley et al. 2011) and are, therefore, not easily integrated into quantitative analyses. In many Aboriginal communities, the Traditional Use Study (TUS) is a key method of collecting social, ecological and cultural knowledge, also called traditional ecological knowledge (TEK) (see Government of Alberta 2003).

A related dataset often available in local communities is Local Ecological Knowledge (LEK). LEK data are collected from individuals with long standing exposure to the local ecosystem. The method involves a mapping exercise where participants draw on their knowledge of the marine environment as well as noted and log book data to drawn features on maps representing important marine locations (see Booth et al. 2005). The data only indicate the presence of an ecological service; typically sea life, recreation and cultural services. The data occur as points, lines and polygons of variable sizes (i.e. from 1 to several hundred km wide features). LEK data are not quantitative but rather incidence-based (i.e. presence/absence) and therefore not ideal for quantitative analyses. Thiago et al. (2010) describe certain quantitative approaches that may be applied to incidence-based data to identify statistically significant clustering of locations repeatedly identified as having species and uses present.

4.2.4 Legislated marine protection data

Canada has national and international responsibilities for the protection and conservation of its marine ecosystems[1]. The establishment of federal marine protected areas falls under the mandate of Fisheries and Oceans Canada, Parks Canada, and Environment Canada[2]. The BC provincial government may also create marine protection measures by establishing provincial parks, ecological reserves and conservancies in the marine environment. Recent land-use agreements have resulted in a significant increase in these areas on the BC Central and North Coasts and Haida Gwaii (J.G. Bones Consulting 2009). Newly proposed projects with potential impacts to ecosystems are a further impetus to implement protection measures through various forms of legislation. The process of establishing protection measures follows a systematic and collaborative approach involving the DFO as the key steward, the provinces and territories, and Aboriginal groups, industry, academia and environmental organizations. The specific approaches/guidelines used to select areas for protection are available at DFO (2009b), DFO (2012a) and DFO (2012b). The

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[1] Example, as a signatory to the United Nations Convention on Biological Diversity, Canada agreed to an international target of conserving 10 percent of marine areas by 2020 through networks of protected areas and other conservation measures (Auditor General of Canada 2012).

[2] Three core programs are involved in the establishment of protected areas: (1) **Marine Protected Areas** (MPA) established by Fisheries and Oceans Canada under the Oceans Act to protect and conserve important fish and marine mammal habitats, endangered marine species, unique features and areas of high biological productivity or biodiversity. (2) **Marine Wildlife Areas** established by Environment Canada to protect and conserve habitat for a variety of wildlife, including migratory birds and endangered species. (3) **National Marine Conservation Areas** established by Parks Canada to protect and conserve representative examples of Canada's natural and cultural marine heritage, and to provide opportunities for public education and enjoyment (DFO 2014a, b). In addition to the three core programs, migratory bird sanctuaries, national wildlife areas and the marine components of national parks are other important elements of the network (DFO 2014a).

outcome is any of a range of protection measures. Figure 4-1 depicts how Fisheries and Oceans Canada views the range of potential protection measures set on a nominal scale ranging from 'highly legislated' to mostly 'voluntary' compliance.

Figure 4-1. The range of ocean management and protection measures available in the Canadian legislation, placed on a relative scale from highly legislated to managed based on voluntary stewardship initiatives (from DFO 2005).

4.2.5 The need for integrated analysis

Integrating data across the four spheres discussed above poses certain additional challenges. As contended by Lawrence (2007), the most difficult methodological task is that of integrating analyses of environmental impacts across disciplines into an overall evaluation framework. Translating impacts into monetary units following the approaches described by Hadley et al. (2011) can help facilitate integrated analysis. However, given the many limitations identified among economic approaches to valuing ecosystems, many argue the need for new methodologies to more effectively integrate economic, ecological and social values (De Groot et al. 2002, Lawrence 2007, Kumar and Kumar 2008, Spangerberg and Settele 2010). Lopes and Videira (2013) contend that new approaches are particularly important in marine and coastal ecosystems which are characterized by high degrees of complexity and inaccessibility/invisibility of their goods and services. Yet, the four spheres of data discussed typically occur in differing and largely incompatible units. Thus, integrated analyses will necessitate a significant degree of qualitative interpretation.

4.3 Methods

To achieve an integrated analysis of the four spheres (i.e. economic, social, environmental and legislation), relevant and appropriate data pertaining to each of the spheres were required. The data sought were required to meet certain criteria as follows. First, the data had to lend themselves to spatial referencing. Second, the data had to be at a scale appropriate for the study region. According to de Jonge et al. (2012), the appropriate scale for integrating socio-economic and ecological information is the scale of the 'habitat' - the level at which ecological functioning, human activities and the socio-ecological compartment can be measured. Third, the data had to exist and have available primary data pertaining to any of the 4 spheres, or an integrated analysis of primary data producing new data representative of one of the four spheres. Fourth, for the purposes of spatial statistical analysis, the data could be either quantitative or incidence-based (i.e. presence/absence data). The data considered in this research and their attributes are presented in Table 4-1.

The specific methods of analysis applied to each dataset are described below and summarized in Table 4-1. In some instances, the data were analyzed using the ArcGIS 10.1 Hotspot Analysis Spatial Statistic tool which is based on the Getis-Ord Gi* statistic. The hotspot tool reports the statistical significance of clustering among high and low value features by proportionally comparing the local value of a feature to the sum value of all features. When local values differ beyond what might be expected by random chance then a statistically significant probability (p) results (also see Appendix C for a discussion pertaining to the use of the ArcGIS Hotspot Analysis tool). Thus, in this research, hotspots are considered to be 'important' areas for the value being measured (i.e. areas with significant clustering of high value features). Finally, an integrated analysis of the results of the four analyses was also conducted. The specific methods used for each of the analyses are discussed below.

The PNCIMA was selected as the general study area for this research. The PNCIMA is one of Canada's 5 large ocean management areas and has been selected by federal, provincial and First Nations governments as a unit of study in several marine use planning processes, such as the PNCIMA marine use planning process (PNCIMA, 2011). The scale of the PNCIMA was found to be relatively appropriate for the data considered in this research. However, the southern extent of the PNCIMA was cut off at 51.5^0 N to match the extents of the LEK data.

Table 4-1. A description of the spatial datasets pertaining to economic, ecological, social and legislated values considered in this research.

	Dataset	Source	Resolution[1]	Description		Analysis
Economic	Commercial salmon fishery	DFO[2] (2013d) online database	PFMAs[3] 10 to 50 km	Commercial fishing catch statistics (2007-2013) referenced to PFMAs	In	Monetary valuation and hotspot analysis based on landed values
	Commercial non-salmon fishery	DFO Fisheries Statistics	4 km grid	Commercial fishing catch statistics on a 4 km grid (2000-2009 & 1993-2004)	In	
	Recreation Fishing	BCMCA (2015)	Values not spatially referenced		Out	
Ecological	EBSA[4] analysis (Phase I)	Clarke and Jamieson (2006a)	2 km to 600 km	Expert knowledge-based selection of EBSAs based on DFO EBSA selection criteria (2006)	In	Hotspot analysis based on expert scoring
	EBSA analysis (Phase II)	Clarke and Jamieson (2006b)	11 km to 600 km	Integrated analysis based on Phase I EBSAs and several categories of unique marine physical features (2006)	In	None. All Phase II EBSAs were considered important
Social	Local Ecological Knowledge (LEK[5])	Booth et al. (2005-2008)	2 km to 600 km	Data identifying important locations for species, habitat, recreation and other human uses (2007/2008)	In	Hotspot analysis based on frequency of presence
	Traditional Use Studies	Data could not be accessed			Out	
Legislated	Federal and Provincial Protected Areas	GeoBC (2008) Natural Resources Canada (2008)	2 km to 70 km	Provincial Parks, National Parks, Conservancies, Candidate National Marine Conservation Areas, and Rockfish Conservation Areas	In	None. All legislated areas were considered important
Integrated analysis of all data selected for inclusion					Overlay analysis to detect overlaps and extents of agreement	

[1] Range of distances measured across the smallest and largest features of the dataset; [2] Fisheries and Oceans Canada; [3] Pacific Fisheries Management Areas; [4] Ecologically and Biologically Significant Areas; [5] Local Ecological Knowledge

4.3.1 Economic data analysis

For salmon species: DFO fisheries harvest data (DFO 2013e) reporting the quantity and value of all salmon harvested from each PFMA were acquired. For each PFMA, salmon harvest data for the five species of salmon (Chinook, Chum, Coho, Pink, and Sockeye) were summed for the years 2007 to 2013. A mean annual salmon harvest value was derived for each PFMA and converted to a per unit area value (i.e. mean dollar (CDN) of salmon annually harvested per km^2). In order to standardize the salmon data to the gridded shellfish and groundfish data, these values were adjusted to report the *mean value of salmon ($) harvested annually per 16km²*. PFMA polygons were then converted to 4x4 km polygon grid cells (consistent with the gridded shellfish/groundfish data) and grid cells were valued with the aforementioned values. Partial grid cells formed on PFMA boundary edges (i.e. smaller than 16 km^2) were valued proportionally.

For shellfish and groundfish species: Spatial data from the DFO Fisheries Statistics department spatially referencing several years of total shellfish and groundfish harvest data to a series of 4x4 km species grids were acquired. The data were recalculated to report the mean annual harvest of each species per grid cell (i.e. mean kg of each species harvested per year per grid cell). Grid cell harvests were then converted to monetary values based on the values and calculations shown in Table 4-2.

For all species: To conduct the required spatial statistics, all polygon grid cells were converted to pixels (4x4 km resolution). Pixels were overlaid for all species, and summed to derive a total value per pixel (i.e. the mean annual value of all species harvested from each grid cell). To detect statistical hotspots, pixels were converted to points; with points placed at each grid cell center. See Mahboubi et al. (2015) for a discussion of point-feature hotspot analysis. Each point was then valued based on the value of the grid cell it replaced (i.e. the total value of that cell). Points were merged into a single layer (approximately 22,000 uniformly spaced points). The ArcGIS 10.1 Getis-Ord Gi* statistic was used to estimate statistically significant clustering of high and low value points using a fixed distance conceptualization of spatial relationships (the most appropriate model for point data) and a 6 km distance band value to ensure a minimum of 1 neighbor for each point.

Table 4-2. The species of the commercial fishery and those considered in the valuation of marine harvest in this research, including the years over which means were derived, the tonnage harvested and the landed values for each species[1].

Salmon (*Oncorhynchus spp)*	**Years**	**Tonnes per year**	**Landed value per year ($ ...000)**	**Price per Kg**
Chinook (*O. tshawytscha*)	2007-13	1,071	8,370	7.81
Coho (*O. kisutch*)	2007-13	779	3,155	4.05
Sockeye (*O. nerka*)	2007-13	4,297	13,950	3.25
Chum (*O. keta*)	2007-13	3,229	4,882	1.51
Pink (*O. gorbuscha*)	2007-13	7,080	4,526	0.64
Non-salmon species				
Dungeness crab (*Metacarcinus magister*)	2001-09	5,653	32,586	5.76
Geoduck (*Panopea generosa*)	2003-09	1,619	31,166	19.26
Octopus (*Enteroctopus dofleini*)	2000-09	63	216	3.44
Prawn (*Pandalus platyceros*)	2000-09	2,072	29,737	14.35
Scallop (several genera)	2001-09	26	156	6.01
Sea cucumber (*Parastichopus* spp)	2000-09	1186	2,121	1.79
Green urchin (*Strongylocentrotus droebachiensis*)	2000-08	84	367	4.39
Red urchin (*Mesocentrotus franciscanus*)	2000-08	3,705	5,759	1.55
Shrimp (*Pandalus & Pandalopsis* spp)	2000-09	1,246	3,354	2.69
Other shellfish (e.g. eusaphids)	2000-09	191	211	1.11
Schedule II species[2]	1996-04			2.28
Zn[3]	1993-04	105,679	85,689	1.74
Groundfish	1996-04			1.09
Sablefish	1996-04	3,739	28,977	7.75
Totals		**141,719 T**	**$255,222**	

[1] Values from the DFO (2013e) presented for all of British Columbia.

[2] Main species under Schedule II license: Lingcod (*Ophiodon elongatus*), dogfish (*Squalus acanthias*)

[3] Main species: rockfish (many spp.), halibut (*Hippoglossus stenolepis*), lingcod (*Ophiodon elongatus*), spiny dogfish (*Squalus acanthias*), skate (many spp), sole (*Parophrys vetulus*) and flounder (*Platichthys stellatus*)

4.3.2 Biological and ecological data analysis

Spatial maps of Phase I IAs collected by Clarke and Jamieson (2006b) for each species and habitat feature in the PNCIMA were acquired from Fisheries and Oceans Canada. These data were analyzed by two methods. **Method 1**: All Phase II EBSA polygons were considered to be homogenously important marine spaces. Collectively, the 15 Phase II EBSAs covered an area of 45,182 km^2 (44.3% of the PNCIMA). **Method 2**: The five attributes of the Phase I IA database including, *uniqueness, aggregation, fitness consequences, naturalness* and *resilience* (each scored by experts on a scale of 1 to 10 per each feature identified) were summed to produce a total score per feature. Polygons were converted to raster images with pixels valued by the total score of the polygon they replaced. For the purposes of visual quality, a fine 0.5x0.5 km pixel size was chosen. Raster images were then converted to points and points were merged into a single layer of approximately 2.2 million points. The Getis-Ord Gi* statistic with a 4 km fixed distance band, thus ensure approximately 64 points per calculation, was used to estimate statistically significant clustering of Phase I IA scores (incidence hotspots). The incidence hotspots ($p<0.05$) covered an area of approximately 26,000 km^2. **Integration**: an overlay of the Phase II EBSAs and the Phase I score-based IA hotspots was conducted to produce the final EBSA hotspots (i.e. the area of overlap between the two analyses).

4.3.3 Social data analysis

LEK data collected for the BC Pacific North Coast and Central Coast (2006-2008) were acquired from Fisheries and Oceans Canada. A summary of the data is presented in Table 4-3.

Table 4-3. A summary of the local ecological (LEK) dataset collected for the north and central coasts of British Columbia (2006-2008).

	Total	Theme				
		Commercial fish species	Recreation[1]	First Nations (cultural)	Ecological	Misc.
No. of data layers	331	187	93	19	11	21
Percent of total	100%	56%	28%	6%	3%	6%

[1] Includes recreation fishing, locations of fish lodges, and the presence of non-commercial wildlife species such as cetaceans, seals and sea lions (i.e. valued for recreational wildlife viewing)

Following the same method as that of Method 2 of the Phase I IA analysis above, LEK polygons representing the presence of species or ecosystem uses were converted to points and merged into a single layer (204,000 points). A 4x4 km polygon grid of the study area was placed over top of the point features and grid cells were valued by the number of points occurring within them. The Getis-Ord Gi* statistic was used to estimate statistically significant clustering of points (i.e. incidence hotspots).

4.3.4 Legislated data analysis

Digital maps of protected areas including, rockfish conservation areas, provincial and national parks, conservancies and candidate national marine conservation areas occurring on the BC Pacific North Coast were acquired from government sources (GeoBC 2008, Natural Resources Canada 2008). It is plausible to assume that the governments of Canada apply these protection legislation to places deemed important and that the extent of that importance is reflected in the degree to which that space is legislated. In other words, locations receiving protection measures on the left end of the scale in Figure 4-1 (e.g. marine protected areas) might be considered more important than those receiving measures on the right of the scale (e.g. migratory bird sanctuaries). However, given the simple ordinal scale of the graph (i.e. sequential but not quantifiable) no quantitative comparison of importance among the areas is possible. Thus, all of the protected areas were given the same ranking and included in the integrated analysis to follow.

4.3.5 Integrated data analysis

The approach to integrated analysis was one of spatial comparisons among commercial fishing hotspots ($p<0.05$), LEK hotspots ($p<0.05$), the final EBSA hotspots and protected areas. A union overlay of the four layers was conducted using ArcGIS 10.1. The output was analyzed for overlaps using two methods. One, the extent of overlap of one layer over the other was determined (i.e. % of layer 1 covered by layer 2). Two, the extent of overlap as a proportion of their combined areas was determined (i.e. overlapping area between 2 layers / combined area). Comparisons were conducted between all combinations of the four layers.

4.4 Results

The results of the analyses described are spatially represented in a series of maps and tables. An integrated analysis of the combined results is also presented in both map and tabular form. The integrated analysis demonstrates the extents of spatial agreement among the individual analyses.

4.4.1 Commercial fishing values

Virtually the entire marine region of the PNCIMA (i.e. an area of approximately 102,000 km^2) was found to have commercial fishing values greater than zero. The average landed value of all species within the PNCIMA was approximately $138 million per year. Within the study area (77,500 km^2) the average landed value was $109 million per year (Table 4-4). The spatial distribution of this total, represented on a 4x4 km grid (i.e. dollars per year per 16 km^2) is shown in Figure 4-2a. High value concentrations are clearly visible in both near-shore areas and open waters (Figure 4-3). Near-shore areas (i.e. 28,700 km^2) account for 37% of the study area but produce 50% of the value of the commercial harvest. Open waters account for the remaining harvest but include certain notable concentrations. For example, an area of approximately 3400 km^2 (4% of the study area) shown in Figure 4-3, accounts for approximately $19 million or 17% of the harvest (Table 4-4). The extent of high value fish harvest clustering is shown in Figure 4-2b. Hotspots ($p<0.05$) are clearly aggregated in several clusters totaling approximately 7,900 km^2 (10% of the study area) but accounting for 49% of the harvest value of the study area.

Table 4-4. Areas of selected regions within the Pacific North Coast Management Area (PNCIMA) and their associated commercial fishing harvest values, expressed as absolute and proportional values.

Region	Area (km^2)	% of study area	Harvest value ($ millions)	% of total harvest value
PNCIMA	102,000	-	$138	-
Study area	77,500	100	$109	100
Near-shore waters [1]	28,700	37	$54	50
Open Waters	48,800	63	$55	50
Northern open water concentration	3,400	4	$19	17
Commercial fishing hotspots ($p<0.05$)	7,900	10	$53	49
EBSA [2] hotspots	7,700	10	$18	17
LEK [3] hotspots ($p<0.05$)	11,600	15	$34	31
Protected Areas	17,000	22	$24	22

[1] Near-shore waters are those within 20 km of the mainland and land masses (islands) > 500 km^2

[2] Ecologically and biologically Significant Areas

[3] Local Ecological Knowledge

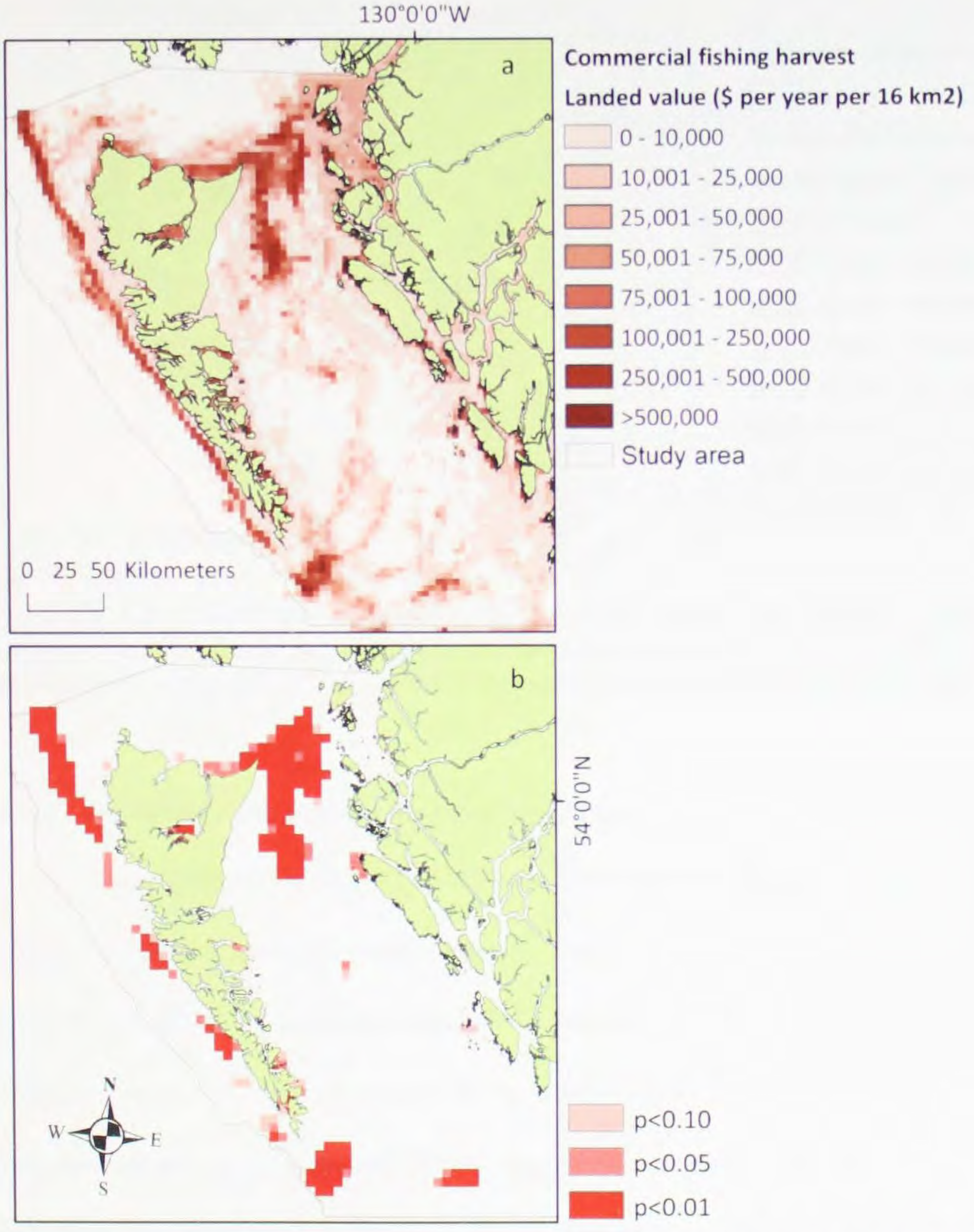

Figure 4-2. Commercial fishing economic data for the North Coast of British Columbia reported on a 4 km grid including (a) the mean multiyear total landed harvest value of all species harvested by the commercial fishery; (b) statistical clustering of high value cells, based on the Getis-Ord Gi* statistic.

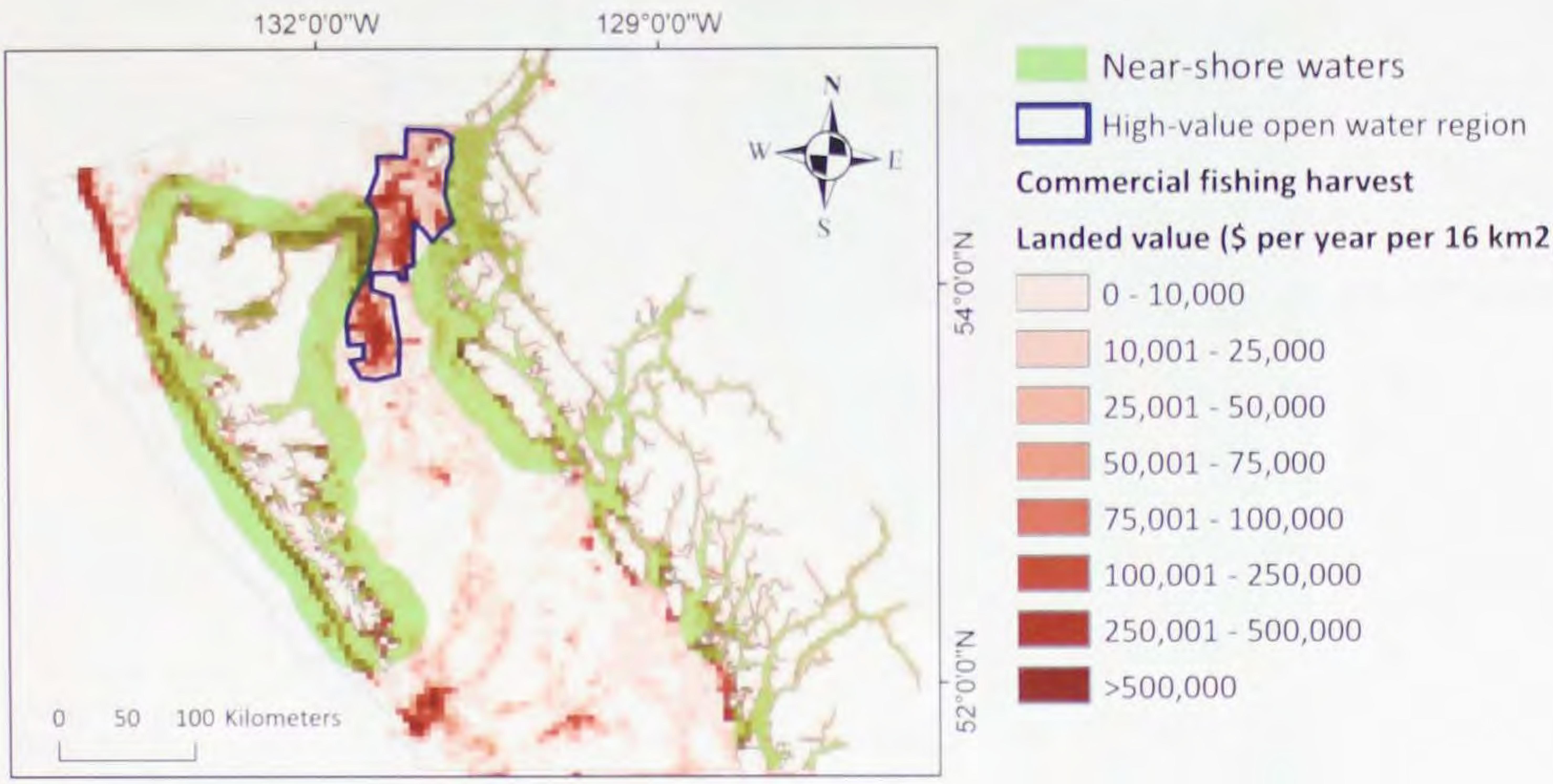

Figure 4-3. A depiction of the near-shore waters (i.e. ocean regions within 20 km of a major shoreline) and open ocean areas of the Pacific North Coast Management Area (PNCIMA). Also depicted is an example of one of the larger commercial fishing high-value clusters in open waters.

4.4.2 Ecologically and biologically significant areas

Phase II EBSAs (Figure 4-4) were found to correspond closely with Phase I IAs mapped by the frequency of expert selection. The association is shown in Figure 4-4a. Phase I expert scored IAs are mapped in Figure 4-4b and their spatial relationship to the Phase II EBSAs is also shown. Hotspot analysis of the Phase I expert scored data produced the hotspot map shown in Figure 4-4c. There was a 16% overlap between the Phase II EBSAs and Phase I expert-scored IA hotspots. The overlap, shown in Figure 4-4d, was considered as the final EBSA hotspots for this research. These EBSA hotspots accounted for 10% of the study area (Table 4-4).

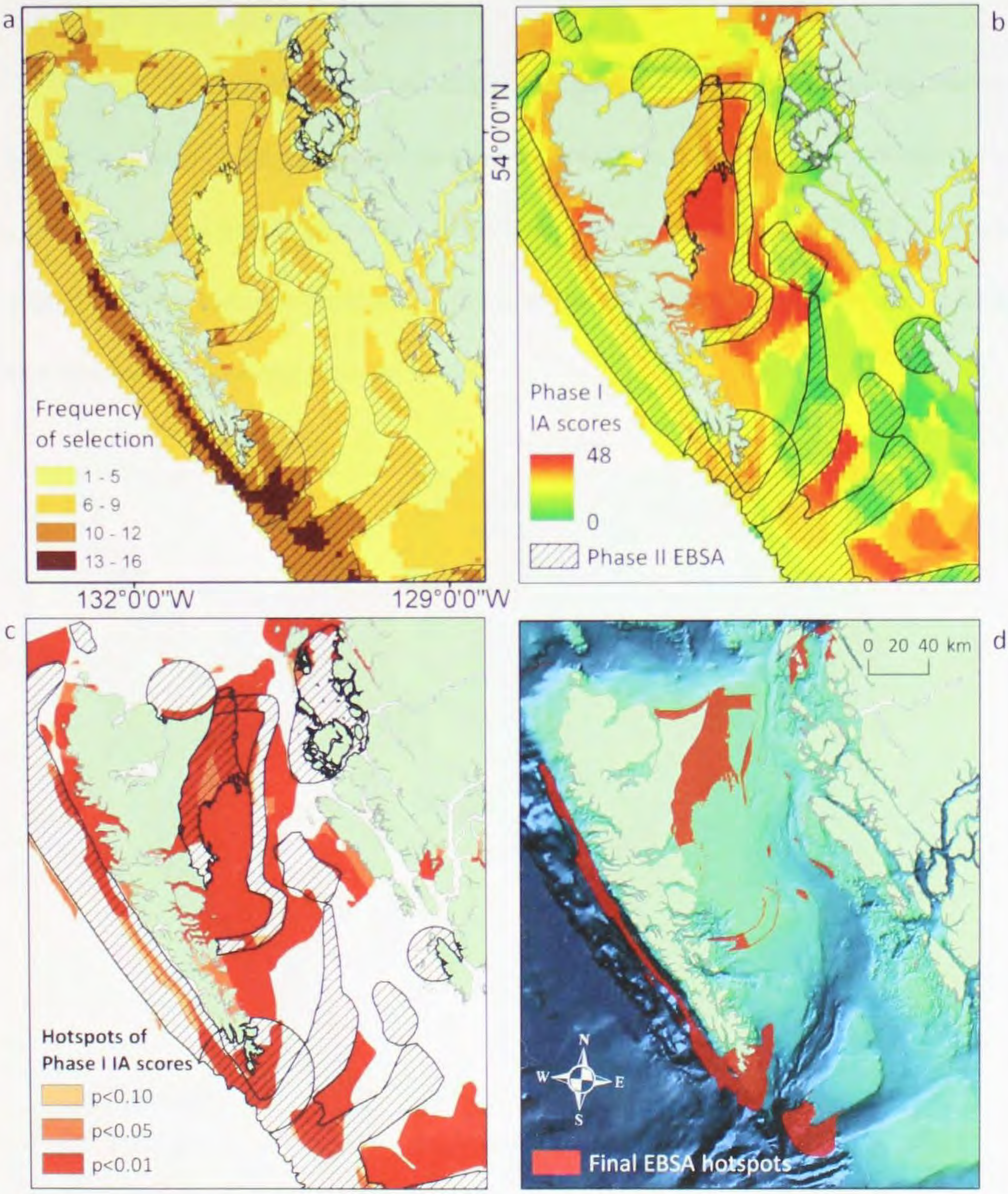

Figure 4-4. Analysis of Ecologically and Biologically Significant Areas (EBSAs) identified for the Pacific North Coast of BC (based on data presented by Clarke and Jamieson 2006b). (a) The final Phase II EBSAs and their spatial relationship with Phase I IAs symbolized by expert-selection frequency. (b) Phase I IAs symbolized by expert-scores. (c) Hotspot analysis of score-based Phase I IAs identifying significant clustering of high scores. (d) The final EBSA hotspots resulting from the overlap of Phase II EBSAs and Phase I score-based IA hotspots.

4.4.3 Local ecological knowledge

Four distinct concentrations of LEK hotspots ($p<0.05$) were detected: the northwest ($5000\ km^2$), northeast ($3000\ km^2$), southwest ($2600\ km^2$) and southeast ($3000\ km^2$) portions of the study area (see Figure 4-5). Collectively, these accounted for 15% of the study area (Table 4-4). It is noteworthy that the focus of the LEK data is commercial fish species; 56% of the data layers collected (see Table 4-3).

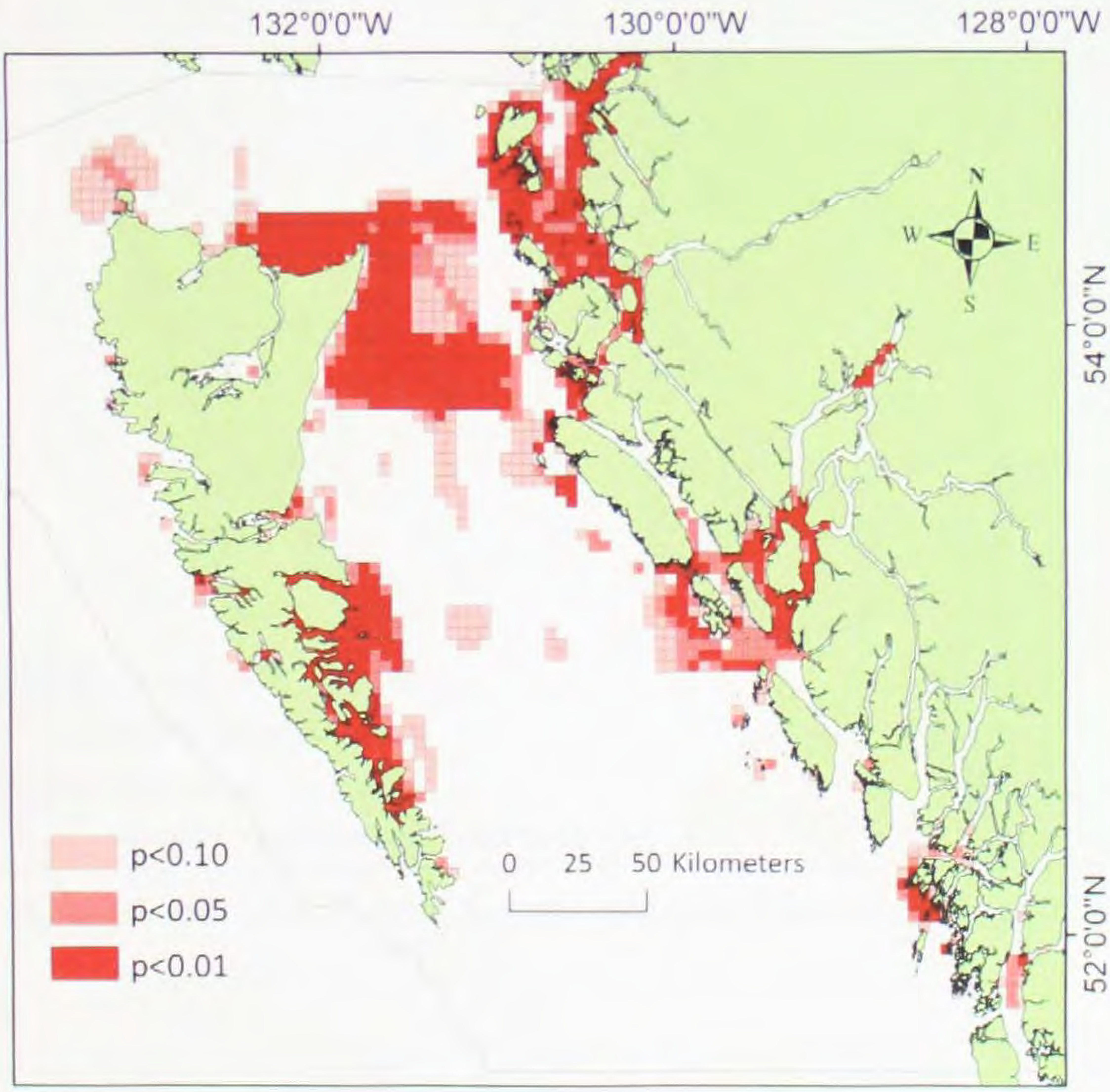

Figure 4-5. Statistical clustering of species occurrences (hotspots) derived from local ecological knowledge (LEK) studies conducted for the North Coast of British Columbia. Hotspots analysis, based on the Getis-Ord Gi* statistic.

4.4.4 Marine protection legislation

Marine legislated locations established on the Pacific North Coast of BC are shown in Figure 4-6. Collectively, these areas account for 22% of the study area (see Table 4-4).

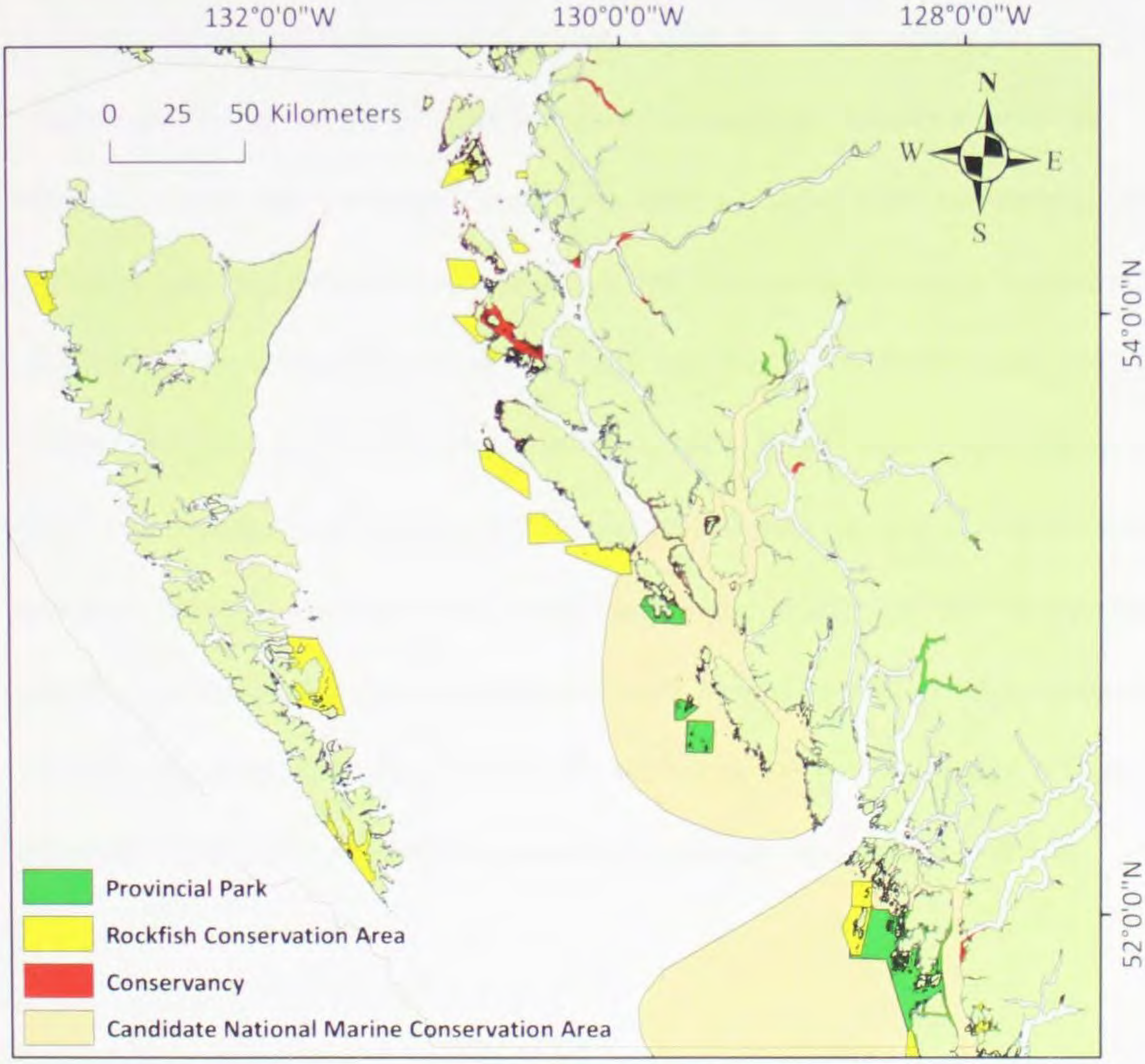

Figure 4-6. Marine legislated areas established on the Pacific North Coast of British Columbia

4.4.5 Integrated analysis

Spatial comparisons conducted between the individual analyses above provided an integrated understanding of the distribution of important spaces in the study area. Overlaps occurring between each analysis are shown in Table 4-5. These include the results of the commercial fishing hotspot analysis compared to the other analyses in order to demonstrate certain relationships as follows: First, a quarter of the commercial fishing hotspots were also identified as important in the EBSA hotspot analysis. Second, almost a third of the commercial fishing hotspots were identified as important by the LEK hotspot analysis. Third, commercial fishing hotspots did not associate well with legislated protected areas (3% overlap). Furthermore, LEK hotspots overlapped nearly a third of the EBSA hotspots. Similarly, protected areas accounted for almost a third of the LEK hotspots, but shared minimal overlap (5%) with EBSA hotspots. Finally, only 3% of the combined area of commercial fishing, EBSA and LEK hotspots were in agreement (see Figure 4-7). No areas coincided for all four analyses (i.e. economic, ecological, social and legislated).

Table 4-5. An integrated analysis showing the extent of agreement between Ecologically and Biologically Significant Areas (EBSAs), commercial fishing and Local Ecological Knowledge (LEK) hotspots (p<0.05) and marine legislated (protected) areas.

Layer	Covered by layer	Portion	Overlap
Commercial fishing (p<0.05)	EBSA final hotspots	25%	15%
EBSA final hotspots	Commercial fishing (p<0.05)	26%	
Commercial fishing (p<0.05)	LEK (p<0.05)	30%	14%
LEK (p<0.05)	Commercial fishing (p<0.05)	20%	
LEK (p<0.05)	EBSA final hotspots	19%	13%
EBSA final hotspots	LEK (p<0.05)	29%	
EBSA final hotspots	Protected Areas	15%	5%
Protected Areas	EBSA final hotspots	7%	
Protected Areas	Commercial fishing (p<0.05)	4%	3%
Commercial fishing (p<0.05)	Protected Areas	9%	
LEK (p<0.05)	Protected Areas	29%	13%
Protected Areas	LEK (p<0.05)	20%	
Overlap of LEK, Commercial fishing and EBSA hotspots		-	3%
Overlap of LEK, Commercial fishing, EBSA hotspots and Protected areas		-	0%

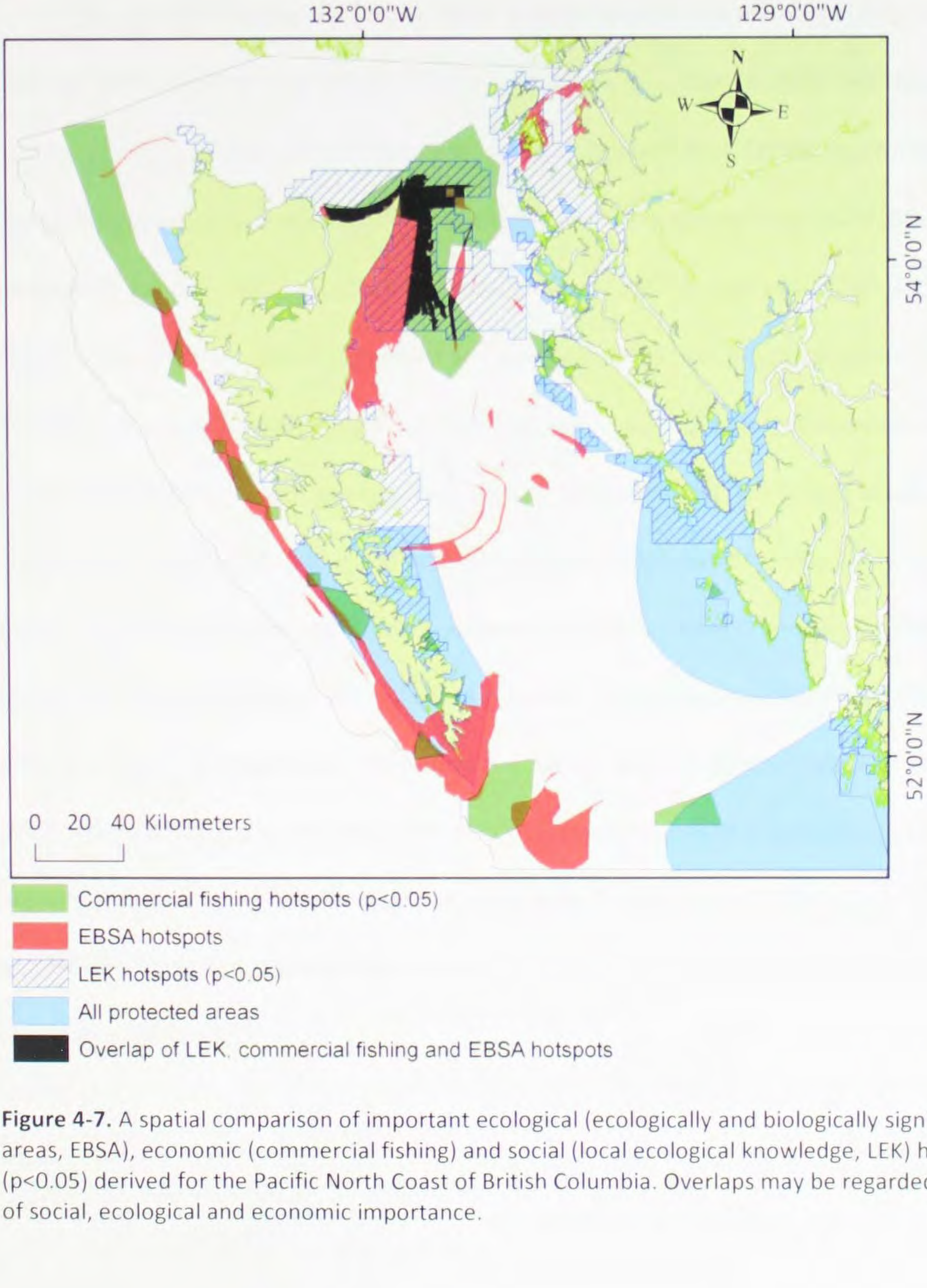

Figure 4-7. A spatial comparison of important ecological (ecologically and biologically significant areas, EBSA), economic (commercial fishing) and social (local ecological knowledge, LEK) hotspots ($p<0.05$) derived for the Pacific North Coast of British Columbia. Overlaps may be regarded as areas of social, ecological and economic importance.

The area of commercial fishing, EBSA and LEK overlap shown in Figure 4-7 above intersected three distinct geographic regions: McIntyre Bay, Dogfish Bank and Hecate Straight (see Figure 4-8). According to a report by O'Donnell et al. (2015), the McIntyre Bay region is designated as the eighth most economically important fishing area in BC; particularly for crab and clam (Crawford and Jamieson 1996, Booth et al. 2005-2008). It is an abundant source of plankton (Crawford and Jamieson 1996) and seasonal habitat for numerous seabirds (Ure and Beazley 2004), fish and whales (Clarke and Jamieson 2006b). It serves as an important location for leisure; particularly for sport fishing and whale watching (Booth et al. 2005-2008, Clarke and Jamieson 2006b). Dogfish bank is seasonally used by a range of seabirds (Morgan 1997, Ure and Beazley 2004, Clarke and Jamieson 2006b) and is an important rearing habitat for cod, flatfish and invertebrate larvae (Clarke and Jamieson 2006b). It is also an important crab fishing and salmon trolling ground (Booth et al. 2005-2008). Hecate Straight is recognized for its high concentrations of zooplankton and is an important area for crab and herring fishing (Perry and Waddell 1997). Both Dogfish Bank and Hecate Straight are whale migration routes with many whale sightings enjoyed by the public (see Table 4-6).

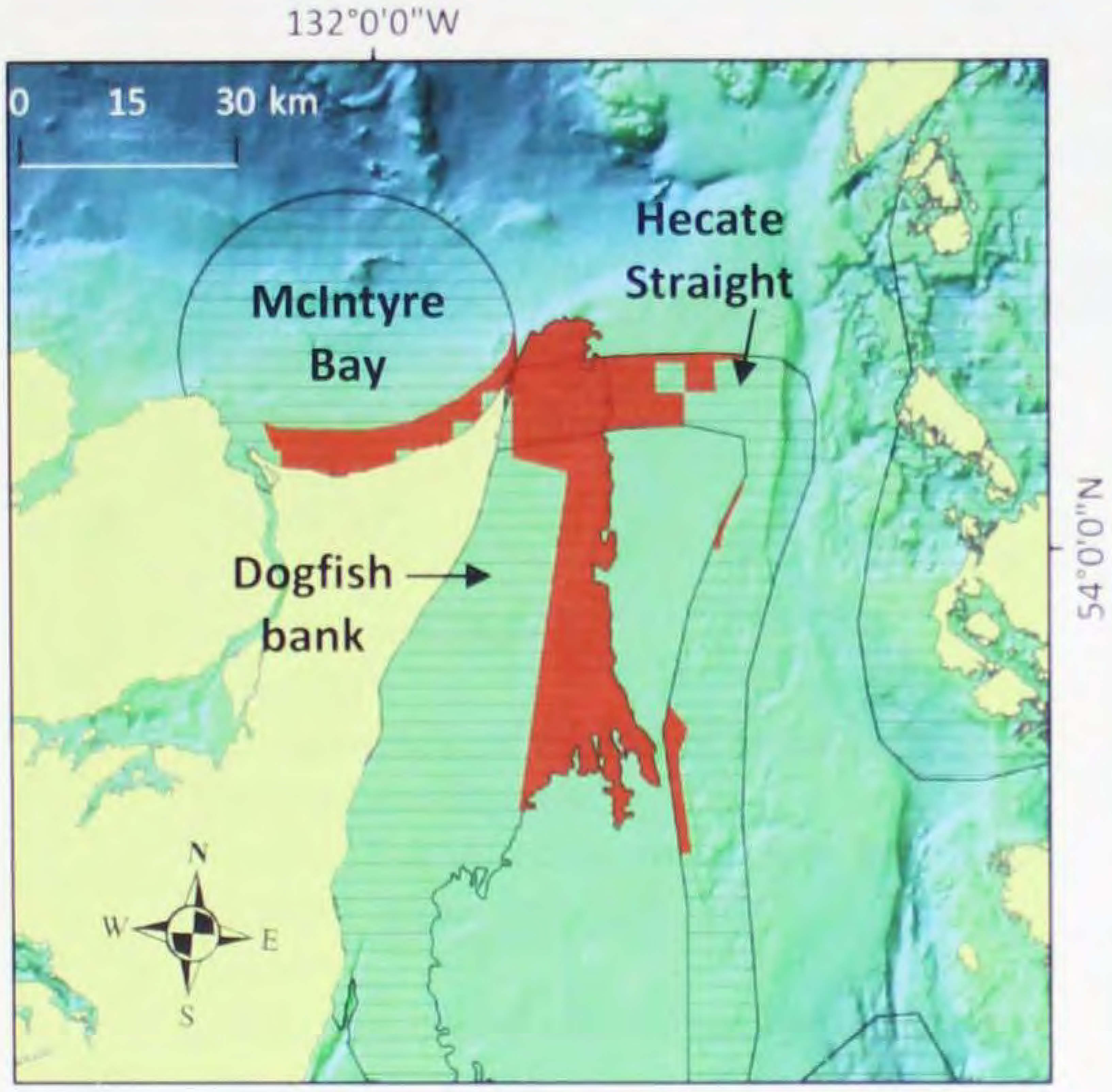

Figure 4-8. Geographic names of areas where economically, ecologically and socially important areas overlap. Red areas are the overlap between commercial fishing hotspots ($p<0.05$), ecologically and biologically significant area hotspots, and local ecological knowledge hotspots ($p<0.05$)

Table 4-6. A summary of the ecological, economic (fisheries) and social characteristics of McIntyre Bay, Dogfish Bank and Hecate Straight.

	Ecological	Fisheries	Social
McIntyre Bay	Concentrations of: - Decapod larvae[1] - Plankton[1] - Seabirds[2] - Eulachon[3] - Humpback whales[4] - Herring[4] - Dungeness crab adults and larvae[1] - Halibut rearing area[5] - Critical habitat northern resident killer whales[4] - Largest razor clam stock in BC[4]	- Crab[1,8] - Clam[4,8] - Salmon troll[8] - Top 10 most economically valuable fishing areas in BC[9]	- Sports Fishing[8] - Cetacean[4,8]
Dogfish Bank	Seabirds: - High densities of phalaropes, herring gulls and ancient murrelets[6] - Migrating sea ducks[2] - Rearing area pacific cod and flatfish[4] - Larval rearing area for high diversity of invertebrate species[4]	- Crab[4,8] - Salmon troll[8]	- Cetacean[8]
Hecate Straight	- Concentration of zooplankton[7]	- Crab[8] - Herring[8]	- Cetacean[8]

[1] Crawford and Jamieson (1996);
[2] Ure and Beazley (2004);
[3] DFO (2000);
[4] In Clarke and Jamieson (2006b);
[5] West Coast Offshore Exploration Panel (1985);
[6] Morgan (1997);
[7] Perry and Waddell (1997);
[8] Booth et al. (2005-2008);
[9] O'Donnell et al. (2015)

4.5 Discussions

This analysis focused on the Pacific North Coast of BC (the PNCIMA) as a case study and drew together spatial data meeting several criteria to locate important economic, ecological and social spaces on the marine environment. Each of the analyses conducted offers certain inferences together with certain challenges and limitations. These are discussed here together with the implications for future research.

4.5.1 Economic analysis

The economic analysis conducted in this research provided a window of perspective on the distribution of economically important marine spaces. Clear areas of economic importance emerged in both the sum total and commercial fishing hotspot maps (Figure 4-2). There are also certain limitations with respect to the data that must be noted. First, the analysis only accounted for the landed value (consumptive use) of marine species harvested through the commercial fishery. Certain other important economic factors were not included, such as the economic contributions of recreational fishing, as discussed above and outlined in Table 4-1. Second, as noted by the BCMCA (2015), the commercial fishing catch data are challenged to represent economic valuations or biological trends for several reasons. One, the data are dependent on areas being 'open' to commercial fishing. Thus, areas not slated for commercial fishing will not have harvest data, but that does not infer that they are less abundant in sea life. Two, the data were screened to meet confidentiality requirements, thus excluding approximately 29% of the data.

The findings of this research highlight the value of economic analyses and the potential benefits for future research to address some of the limitations identified as follows. One, additional spatial data, or spatial referencing of existing data, are needed pertaining to several important economic activities, including recreational fishing and ecotourism. Two, further consideration is needed of the quantification of certain *non-consumptive values* and *non-use values* (e.g. the cultural and aesthetic values of ecosystems and the values attached to ecosystems for their existence, respectively). See Hadley et al. (2011) for a description of use and non-use values.

With respect to the latter consideration, van der Ploeg et al. (2010) proposed several approaches to deriving monetary values for a broader range of ecosystem services that may then be readily integrated into analyses. Among these approaches, Contingent Valuation (CV) is regarded as one of the preferred techniques available. The CV process determines what people would be willing to pay to prevent specified changes in the quantity or quality of their environment and the uses thereof, or what they would be willing to accept in compensation for specified impacts (see Carson et al. 2003, Carson and Hanemann 2005). The suitability of approaches such as CV within the social-cultural framework of the PNCIMA study area needs to be determined.

4.5.2 Biological and ecological data analysis

Unlike the economic analysis, the EBSA data were analyzed outside of monetary considerations. Instead, data generated through a biophysical analysis of the study area (i.e. EBSA Phase II) were combined with data pertaining to expert spatial selection and scoring of the same areas (EBSA Phase I). The analysis provided important insights with respect to the spatial distribution of important ecological spaces.

The data and works of Clarke and Jamieson (2006b), Clarke and Jamieson (2006a) and Gregr et al. (2012), which served as the basis of this analysis, are arguably among the best data available in the study area. However, it should also be noted that the DFO (2013d) reports certain limitations with respect to the EBSA data that may need future consideration. Limitations include the variability of data quality and boundary accuracy, the process of selecting experts and the heavy reliance of the analysis on expert knowledge as opposed to published literature. Furthermore, some of the species data (e.g. salmon related IA data) are temporal (seasonally dependent), but were not captured as such in the analysis. Lastly, given the lack of information and consistency of application, nearshore areas including estuaries, river mouths, beaches, inlets, fjords, and other shallow sub-tidal areas were not adequately represented (DFO 2013d).

Yet, Gregr (2007) contends that the additional ecological data needed to address these limitations are unlikely to become available in the near future. Moreover, the complexity of coastal ecosystems and the limited understanding of coastal ecological processes create additional challenges for measurement. Thus, it may be prudent for future research to focus on approaches to better integrate existing data in order to overcome some of the limitations noted.

4.5.3 Social analysis

The LEK data, as with the EBSA data, were analyzed outside of monetary considerations, and were based instead on the frequency of spatial selection. This yielded statistical hotspots representing areas of social importance. It is noteworthy, however, that despite their social underpinning, much of the focus of the LEK (as noted in Table 4-3 above) is that of the presence of commercial fish species, and excludes many other important social-ecological services (see Landscape Values PPGIS Institute 2015b). Thus, despite the inclusion of the LEK data in this research, additional work is needed to develop more appropriate approaches to collecting and analyzing a broader range of social-ecological values to be incorporated into the decision-making process (see Peterson et al. 2009, Brown 2012a).

4.5.4 Integrated analysis

Individually, each of the datasets analyzed reflected spaces important for a single criterion: economic (Figure 4-2), ecological (Figure 4-4), social (Figure 4-5) and legislation (Figure 4-6). They did not singly provide an integrated understanding of importance. However, when examined collectively (Figure 4-7), areas of overlap provided new insights with respect to certain relationships between datasets, as well as the spatial distribution of ecosystem spaces considered socially, ecologically and economically important. A number of inferences may be drawn from the integrated analysis as follow:

- The 26% overlap of commercial fishing hotspots on EBSA hotspots (Table 4-5) may suggest that EBSAs can, in fact, represent ecological conditions that promote high biological productivity.
- The tendency of LEK hotspots to overlap commercial fishing hotspots (i.e. 30%) is a potential inference that LEK knowledge is associated with commercial fishing (consistent with the species-based focus of the LEK data).
- The poor association of commercial fishing hotspots with legislated protected areas (i.e. 3% overlap) suggests that these activities are largely exclusive of one another (be it deliberate or by circumstance).
- The 29% agreement between LEK and EBSA hotspots suggests some consistency between the two knowledge-bases and their approaches to valuation.
- The 29% overlap of protected areas over LEK hotspots may be an indication of the knowledge and involvement of the public in legislation processes related to marine conservation.
- The minimal overlap between protected areas and EBSA hotspots (5%) may be an indication of the need for new conservation and marine protection measures in the study region.
- By the measures used in this study, the areas of agreement between commercial fishing, EBSA and LEK hotspots would suggest areas of integrated importance (i.e. social-ecological-economic hotspots).

The main challenge of conducting an integrated analysis of the four datasets considered in this research is that they are based on significantly different approaches and units of measure. The commercial fishing data were in monetary units, while the EBSA and LEK analyses were based on incidence data collected from various expert sources. Furthermore, despite the common basis of the latter two analyses (i.e. incidence-based), the different approaches used to collect them rendered their integration challenging.

For example, in all three analyses significant clustering of either incidence or high economic value areas were statistically detected. These were referred to as hotspots when $p<0.05$. These locations were considered important for the values they measured. Yet, these hotspots could not be quantitatively compared across the analyses as they could not be ranked with respect to their relative importance. Thus, though the final integrated map (Figure 4-7) visually represents the range of spatial relationships that exist between analyses, the extent to which each analysis contributes to the overall importance of a site must be determined by a qualitative evaluation of the data.

4.6 Conclusions

Growing pressures for development of marine systems near coastal communities necessitate new approaches to integrated decision-making. Available social, ecological and economic data are often the resources that are at hand. Spatial analyses of four selected datasets (i.e. commercial fishing harvest data, EBSA data, LEK data and marine legislated protected areas) resulted in the detection of various categories of important marine spaces.

An integrated analysis of the combined data produced an integrated spatial perspective of important social-ecological-economic spaces. Integration was, however, challenged by differing approaches to data collection and incompatible units of measure. As new proposals for marine development are introduced to communities in many coastal regions, integrated approaches to collecting and integrating data pertaining to the marine social-ecological system become increasingly necessary. This research demonstrates one such approach. Its utility in practical applications of marine planning and management needs to be tested.

Chapter Five. Using Expert Informed GIS to locate important marine social-ecological hotspots

5.1 Introduction

Coastal communities often rely on the marine environment for natural resources to help maintain livelihoods (Brotherston and White 2006), meet nutritional needs, advance scientific learning (Molnar et al. 2009), practice traditional knowledge (see Assembly of First Nations 2003, CRIFC 2010a, Chan et al. 2011) and more. Current projects and new proposals involving development and use of coastal waterways, such as the shipping of crude oil, the construction of natural gas plants, the expansion of ports and many others (see examples for northern British Columbia, Canada in Carleton Ray and McCormick-Ray 2013) could result in significant marine environmental impacts.

A major challenge facing natural resource managers and environmental decision-makers is to plan for these impacts, with the objective of minimizing adverse effects to both human and ecosystem health. These decisions would be facilitated by an understanding of the locations of marine spaces associated with important social or ecological values (e.g. economic opportunities), and the degree to which those spaces are important. With such an understanding, anticipated impacts could be spatially distributed with the objective of avoiding spaces with the highest social-ecological values. This begets several challenging questions: what criteria should be considered as a measure of importance? How do we measure those criteria? How do we integrate and analyze those measurements in order to draw inferences and make environmental management decisions? This paper proposes xGIS as a tool to help answer these questions by drawing on local environmental knowledge expertise to identify important social-ecological marine spaces.

Insufficient data and a limited understanding of the complex cross-linking relationship between human and environmental health (Birley 2002, Noble and Bronson 2005, Braveman et al. 2011) have posed challenges to answering the questions posed. The challenges were partially addressed by the early works of Tuan (1974, 1977) and Relph (1976), and the more recent works of Bechtel and Churchman (2002), who recognized a relationship between people and their environment. They described people as active participants in the landscape - thinking, feeling, acting and receiving information from both observation and experience. In doing so, they gain 'perception' and thereby attribute 'meaning' to landscapes, ultimately developing a 'sense of place'. Zube (1987) and others (Brown 2005) further build on this phenomenon and describe the human-landscape relationship model; proposing that individuals who develop such place attachments are often capable of associating a quantifiable range of values to places (Brown 2005). Rolston and Coufal (1991) and others (see Landscape Values PPGIS Institute 2015a) proposed an iterative list of landscape value attributes (see Table 2-3) to reflect the human-landscape relationship. Brown (2012a) described these landscape value attributes, as "layers of human perceptions" that can be spatially referenced and overlaid on the physical landscape. Brown and Reed (2011 p.1) asserted that the "human process of valuing landscapes results in structural and distributional patterns on the landscape that, although not directly observable, constitute latent patterns of social and psychological complexity that can ultimately be measured and quantified". Therefore, in the fields of natural resource and environmental management, the sense of place can help bridge the gap between the science and the management of ecosystems (Mitchell et al. 1993, Brandenburg and Carroll 1995, Williams and Stewart 1998, Eisenhauer et al. 2000, Brown 2005) and help predict resource conflicts (Brown and Raymond 2007).

There is, however, no definite consensus on how to specifically measure sense of place; especially in the context of diverse socio-cultural conditions (Kaltenborn and Bjerke 2002) and few techniques that explicitly provide for the inclusion of this form of knowledge in the planning and analysis (Brown et al. 2004). Much of the work that has been done focuses on collecting qualitative data about the connections of people with special places (Mitchell et al. 1993, Brandenburg and Carroll 1995); data that are not easily integrated with existing biophysical inventories (Brown 2005).

The field of PPGIS addresses one aspect of this need for measurement by tapping the knowledge of the 'general' public to quantifiably and spatially detect a range of social and ecological hotspots (Brown 2012a). A PPGIS survey instrument is sent to every household in the community (Brown and Reed 2009) and the data collected are assumed to represent the views of the 'silent majority' (Alessa et al. 2008), the broad views of the entire local population (Brown and Reed 2012). While many approaches (e.g. Valipour 2014) are generally applicable to large regional or national scales, PPGIS occurs at a scale that is useful for local planning.

The xGIS approach is an adaptation of PPGIS - focussing on the knowledge of local experts, rather than the general public, as a means of improving spatial accuracy. In such instances, the public is not viewed as one homogenous group to be randomly surveyed, but rather consisting of individuals with specific areas of knowledge and expertise. In such cases, a stakeholder-participant transition process is needed. Parkes (2011) described this process as beginning with the determination of existing knowledge strengths and deficits across multiple stakeholders, followed by a transitioning phase when stakeholders are invited to become research participants; thus establishing the 'participatory research community'.

While xGIS appears to be a promising tool, there have been few studies demonstrating how it can be applied in the context of environmental management or its challenges and limitations. This paper explores the use of xGIS as an innovative approach to detecting and quantifying the spatial distribution of important social-ecological hotspots in the marine ecosystem based on expert social-ecological knowledge from the local community. We use this case study to demonstrate the usefulness of xGIS in the field of marine resources management. The specific objectives of the study include: (1) to determine the feasibility of recruiting expert participants; (2) to apply xGIS to detect important marine social-ecological hotspots; (3) to critically evaluate the methodology and results. In an era of increasing inclusion of civil society in environmental decision-making (Janicke 2008), we propose that xGIS can serve as timely and useful tool to help bridge the gaps.

The region of northwest BC where this case study was applied is broadly consistent with many small coastal communities. It has a relatively small population (less than 20,000) and is comprised of residents of a wide range of ethnic origins, including Aboriginal peoples (37.5%) (Statcan 2010). Socio-economic conditions (10.7% unemployment) and levels of education (57% with no post-secondary education and 28% aged 25-64 with high school as their highest educational attainment) all lag behind the BC average (In Stantec 2014), as do most health indicators (Fang et al. 2010). The region is composed of a mountainous temperate rainforest ecosystem and a highly productive and bio-diverse marine ecosystem (PNCIMA 2011). The economy is based largely on natural resources, including fishing, forestry, energy, transportation and tourism (BC Stats 2014). The general region includes a small central community and a number of outlying First Nation's villages. Subsistence living is common, especially in outlying communities.

5.2 Methods

Four major components were involved in the development of the xGIS tool: selecting who to survey, applying the survey instrument, determining the extent of surveying, and analyzing the data. These components are described below. Human research ethics approval was obtained from the Research Ethics Board of the University of Northern British Columbia.

5.2.1 Selecting expert participants

The knowledge categories 'specialized' and 'individual' proposed by Brown (2007) were deemed the most reflective of the range of local expertise in the study region and were, therefore, used in this study. These two categories were further sub-divided as shown in Table 5-1. The initial selection of experts was based on individuals known to the primary investigator as being widely recognized and accepted in the region as having significant local marine spatial knowledge. These individuals were contacted and invited to become part of the initial participatory research community. The design and objectives of the study were explained. They were then asked to recommend others who they felt had a high level of knowledge related to the study (referrals). Recurring referrals were contacted and the same method applied. The procedure was repeated until adequate participation was achieved (see discussion on sample size below). Each participant was asked to self-identify the knowledge category that best described the source of their knowledge, followed by the second, third and fourth categories where relevant.

Table 5-1. Categories of knowledge expertise considered relevant to the study region

Specialized	Individual
Independent consultant/scientist	Politician with a marine portfolio
Government marine scientist	Longtime resident
Government marine administrator	Marine leisurist
Marine watchmen or patrolmen	
Commercial fisher	
Food fisher	
Traditional knowledge	
Sport fishing or Ecotourism guide	
Marine-based NGO* staff	
Community health worker	

*Non-governmental organization

5.2.2 Mapping with participants

Thirteen landscape value attributes were selected for consideration (see Table 5-2). The selection was based on those attributes most commonly used in past PPGIS studies (see Table 2-3). The final attribute selected (special places) allowed for participants to add additional value attributes as needed.

Participants were surveyed individually (i.e. one participant at a time. The value attribute descriptions listed in Table 5-2 were reviewed with each participant. The participant was then briefly taught to use a Wacom® 23 inch pen display connected to ArcGIS 10.1. The participant was asked to begin drawing (digitizing) areas (polygons) on a digital map of the Pacific North Coast Integrated Management Area (PNCIMA), identifying locations deemed important with respect to each attribute. Participants were asked to focus on the North Coast study area delineated in the Marine Planning Partnership (MaPP) framework; an area of approximately 25,000 km^2 extending from the Khutzeymateen Inlet (54.44° N) to just south of Douglas Channel (52.17° N) and within approximately 100 km of the coastline (see Figure 1-1). Various maps of different scales were provided as needed, with place names added as an additional aid.

Upon completion of digitization, the 13 resulting attribute maps were simultaneously displayed on the screen. A total of 33 tokens of various denominations were digitally placed on the side of the screen. The tokens were color-coded and labeled with their values as follows: 2 worth 200 points each, 4 at 100 points each, 17 at 25 points each, 5 at 10 points each, and 5 at 5 points each. This denomination of tokens was selected with the goal of producing a sum total of 1300 points, thus matching many typical PPGIS studies (e.g. Brown and Raymond 2007) where 13 value attributes are provided with 100 points available per attribute for distribution. The participant was asked to move the tokens to polygons on any map in order to value them. Participants were permitted to adjust their allocations until a balance was reached that they felt was reflective of their knowledge and understanding. Participants were informed that they did not have to allocate all 33 tokens if they did not feel justified in doing so.

Table 5-2. The landscape value attributes considered in this study

	Attribute	Description
1.	**Scenic Aesthetic**	I value these areas for their scenery; their mountains, forests, tidelands, bays and islands
2.	**Economic**	I value these areas because they provide income and employment opportunities through industries like commercial fishing, shipping, tourism, or other commercial activity.
3.	**Recreation**	I value these areas because they provide for recreation activities such as boating, sport fishing, or wildlife viewing.
4.	**Life Sustaining**	I value these areas because they help produce, preserve, clean and renew air, soil and water.
5.	**Scientific Learning**	I value these areas because they provide opportunities to learn about the environment through scientific study.
6.	**Biodiversity**	I value these areas because they support and provide habitat for a variety of marine species including animals, plants and birds.
7.	**Spiritual**	I value these areas because they are sacred, religious or spiritually special places and I feel reverence and respect for nature there.
8.	**Existence (Intrinsic)**	These areas are valuable for their own sake, even if I or others don't use or benefit from them.
9.	**Cultural Heritage**	I value these areas because they have features that represent history or provide places where people can continue to pass down wisdom, traditions and a way of life.
10.	**Future**	I value these areas because they provide opportunities for future generations to know and experience them.
11.	**Subsistence**	I value these areas because they provide necessary food and materials to sustain people's lives.
12.	**Therapeutic**	I value these areas because they make me or others feel better, physically and/or mentally.
13.	**Special Places**	I value these places because they are special to me. Please indicate the reason why the place is special to you.

5.2.3 Determining sample size

We used the approach described by Brown and Pullar (2011) to determine the number of interviews needed to achieve spatial convergence (i.e. whereupon the collection of points converge on a collective spatial 'truth'). Assuming an average of 5 polygons identified per attribute per participant in the current study, 13 to 24 interviews would be needed. In the case of this study, this range was used to decide when adequate data had been collected and the stopping rule satisfied. Additionally, non-parametric estimators were used to determine the proportion of the region's experts surveyed (see Section 5.3.3.1). These estimators rely on the number of new or 'rare' experts (singletons) that appear in each successive survey, as compared to doubletons. EstimateS software was used to estimate the total number of experts likely to exist and, thus, the proportion of the expertise tapped at any given point in the study.

5.2.4 Data processing and analysis

To determine the total value (score) of each point on the waterscape, the raw data (polygons) had to be *cleaned* (i.e. any portions crossing onto land erased). Cleaned polygons were each given a default score of 1 point. These were added together with any points allocated to polygons from the token allocation exercise. This produced a total score for each polygon. These scores were converted to a density value (i.e. points per 100 km^2). The polygons were then converted to raster maps (500m pixel resolution) that could be overlaid and summed. Pixels were individually valued according to the density value of the underlying polygon.

The total value of each pixel was determined by grouping raster maps by value attribute and then overlaying and summing their overlapping pixel values. This produced 13 maps (one per value attribute) (Figure 5-2). The 13 maps were then overlaid and summed to produce a sum total map (Figure 5-3).

To detect hotspots (i.e. locations of statistically significant high value clustering), the raster maps were converted to points, with each point valued according to the underlying pixel. These maps were similarly grouped by attribute and combined producing 13 attribute maps. These maps were individually analyzed for hotspots based on the Getis-Ord Gi* statistic in ArcGIS 10.1 (see Section 0). The analysis was conducted using a 6 km fixed distance band to ensure inclusion of adequate numbers of neighboring points in the statistical analysis. This produced a series of hotspot maps (Figure 5-4). Finally, the 13 point-feature maps were combined to produce a single map and analyzed for cumulative hotspots (Figure 5-5).

To determine the degree of agreement among knowledge groups with respect to high and low value locations, the original polygon maps were combined (appended) by attribute producing 13 polygon-attribute maps. The High/Low Clustering Getis-Ord General G statistic in ArcGIS 10.1 was then applied to each map in order to measure the degree of clustering of high and low value locations. The procedure was repeated once more, this time asking the question within knowledge groups (i.e. whether experts within each knowledge category agreed among themselves as to high and low value locations).

Analysis was also conducted to determine the distinctness of the knowledge categories considered. That is, whether participants who self-identify as knowledge category A also self-identify as knowledge category B and so on, or whether they are distinct and unrelated communities of knowledge. The Pearson product moment correlation coefficient (r) was used to compare each pair of variables, measuring the degree of linear relationship between them.

5.3 Results

5.3.1 Participation and sample size

Participants (n=21) appeared generally eager to participate in the study once oriented to the technology (i.e. approximately 10 minutes of training was required per participant to learn to draw polygons onto maps using the hardware and software). Furthermore, despite certain similarities among the landscape value attributes considered (e.g. economic and subsistence) and a degree of uncertainty around others (e.g. existence and future), participants appeared generally confident in their differentiation and understanding of each category; as inferred by the infrequency of explanatory questions posed and the confidence during mapping. The token allocation exercise was also well received and the use of 33 tokens found to be manageable as participants did not appear strained as they employed the many mental checks and balances that need be contemplated before a final decision of allocation was made.

In total, 21 surveys were conducted with participants belonging to one or more of the various knowledge categories listed in Table 5-1. One quarter of the participants considered their length of residency in the region to account for a significant portion of their knowledge base; followed by commercial fishing and general marine leisure as the next two most significant sources of expert knowledge (Table 5-3). Certain categories of knowledge had no participants. For example, no community health workers with marine spatial knowledge were identified; neither in the initial surveys, nor through subsequent referrals.

Table 5-3. The distribution of participants across self-identified knowledge categories.

Knowledge Categories	**No. of Experts**	**% of Total Participants**
Longtime resident	16	26%
Commercial fisher	11	18%
Marine leisurist	7	12%
Traditional knowledge	5	8%
Independent consultant / scientist	5	8%
Marine watchmen or patrolmen	4	7%
Food fisher	4	7%
Sport fishing or ecotourism guide	3	5%
Marine-based NGO* staff	3	5%
Government marine scientist	1	2%
Government marine administrator	1	2%
Politician with marine portfolio	0	0%
Community health worker	0	0%

*Non-governmental organization

5.3.2 Mapping

A summary of the data collected from experts through the survey effort is presented in Table 5-4. Approximately 1500 polygons were collected ranging from relatively small polygons (i.e. cultural, subsistence and recreation) to relatively large polygons (i.e. future, life sustaining and scenic). Over half of the polygons (64%) belonged to 4 categories: biodiversity, economic, subsistence and recreation. These four also received 58% of the total token point allocation with biodiversity as the most important, taking 21% of the point allocation, followed by economics, subsistence and recreation with 16%, 12% and 9% of the points, respectively.

Table 5-4. Summary of data collected in the survey effort including: total tokens allocated to each attribute (absolute and percent), percent of tokens allocated to each attribute by expert category, the number of polygons and associated size ranges by attribute, and the density of token points by attribute.

		Bio	Eco	Sub	Rec	Cul	Sci	Fut	Lif	Exi	Sce	Spe	Spi	The
Total Score		6036	4579	3034	2581	1961	1930	1807	1394	1330	1180	1037	955	456
Score (%)		21%	16%	12%	9%	6%	7%	7%	4%	5%	4%	4%	3%	2%
Long. Res.		21%	14%	12%	10%	6%	6%	7%	3%	5%	4%	6%	4%	2%
Com. Fisher		18%	20%	10%	8%	6%	7%	6%	4%	6%	4%	7%	4%	1%
Leisure		23%	10%	12%	14%	4%	7%	8%	5%	3%	6%	0%	6%	2%
Traditional		18%	15%	20%	7%	6%	11%	9%	6%	4%	1%	0%	2%	1%
Consultant		30%	12%	10%	6%	8%	14%	5%	7%	3%	4%	0%	0%	1%
Patrolman		35%	16%	7%	5%	8%	3%	3%	4%	4%	3%	10%	1%	0%
Food Fisher		15%	16%	21%	9%	8%	2%	8%	7%	5%	1%	0%	8%	1%
Sport Fisher		25%	34%	2%	14%	0%	10%	2%	5%	0%	7%	0%	0%	2%
NGO		13%	12%	4%	4%	14%	5%	4%	4%	17%	5%	12%	4%	2%
Gov. Sci.		36%	5%	23%	9%	0%	1%	20%	2%	0%	4%	0%	0%	0%
Gov. Adm.		0%	20%	21%	4%	1%	0%	21%	0%	4%	2%	0%	15%	13%
No. of Polys		244	288	216	203	146	83	52	44	40	87	12	53	28
Area	**Min**[1]	0.03	0.16	0.04	0.11	0.04	0.17	0.26	5.8	0.26	0.32	0.46	0.26	0.15
	Max	12,471	4,962	2,988	3,764	58,913	85,076	23,302	23,302	8,399	10,913	5,390	4,735	2,647
	Mean	277	201	150	138	139	408	1,603	1,415	843	2,726	1,467	247	222
	Median	26	20	21	22	21	72	137	266	92	32	293	57	18
	Std Dev	1,078	566	418	437	534	1,069	4,220	3,756	1,985	1,222	1,961	714	509
Sum Poly Areas		67,504	57,856	32,450	28,010	20,266	33,864	83,374	62,252	33,738	23,713	17,602	13,116	6,210
Density[2]		21	16	12	9	6	7	7	4	5	4	4	3	2

[1] All values reported in km^2; [2] Points /100 km^2. Bio = Biodiversity; Eco = Economic; Sub = Subsistence; Rec = Recreation; Cul = Cultural; Sci = Scientific; Fut = Future; Lif = Life Sustaining; Exi = Existence; Sce = Scenic; Spe = Special Places; Spi = Spiritual; The = Therapeutic

5.3.3 Analysis

5.3.3.1 Completeness of sampling

The 'expert accumulation curve' demonstrated an exponential function rather than the emergence of an asymptote (Figure 5-1). Thus, non-parametric estimators were necessary to determine the completeness of sampling (i.e. to estimate the total number, or richness, of experts in the region). EstimateS™ software recommended the use of classic statistical analysis rather than bias-corrected analysis due to the coefficient of variation for the incidence distribution of the data being >0.5 (i.e. 0.662 for this data). It also recommended reporting the larger of Chao2 and ICE as the best estimate for incidence-based richness. ICE estimated a richness of 367 (SD = 34), while Chao2 estimated 354 (SD = 42) with a lower and upper 95% confidence bound of 290 and 459, respectively. Thus, the non-parametric estimation method suggests that approximately 367 referrals would have been generated for the region at infinite sampling. Thus, a sample size of 21 would suggest that 6% of the experts of the region were surveyed. See Appendix D for further discussion on the approach used to determine expert richness and the completeness of sampling.

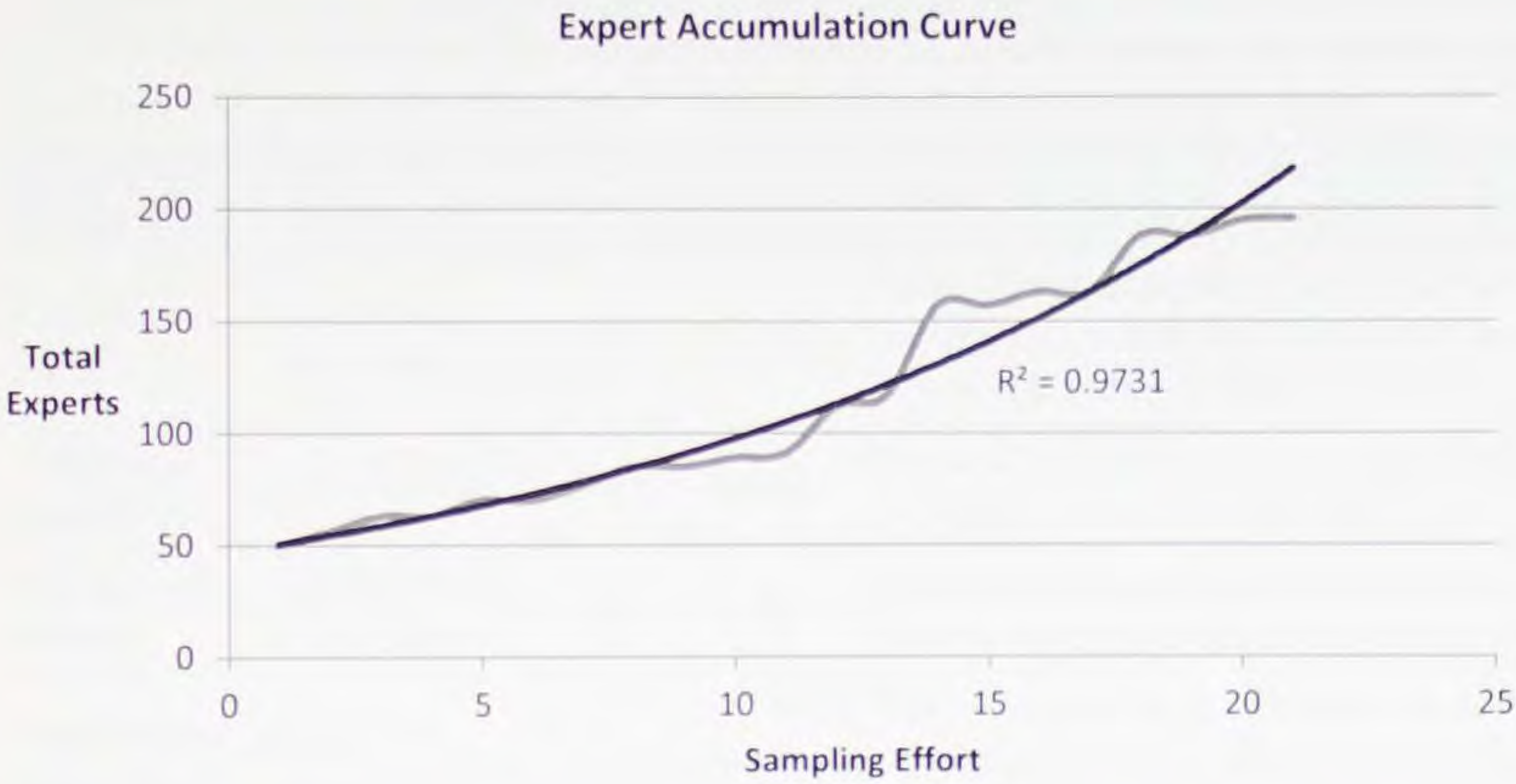

Figure 5-1. Expert accumulation curve constructed from chain referral data. Best fit exponential trend line (dotted) shows no sign of an emerging asymptote.

5.3.3.2 Expert category correlation statistics

The degree of correlations (positive and negative) among the knowledge categories based on the Pearson product moment correlation coefficient (r) is shown in Table 5-5. All comparisons not reported showed no significant correlation (i.e. no association was detected). The table suggests that those with knowledge from commercial fishing also consider their length of residency in the region to be a source of knowledge (r +0.36 at $p=0.106$), that food fishers also have traditional knowledge (r +0.58 at $p<0.01$) and that independent consultants do not consider their knowledge to be a result of commercial fishing (r -0.36 at $p=0.106$) or their length of residency in the region (r -0.48 at $p<0.05$).

Table 5-5. Correlations detected between the knowledge categories considered in this study using the Pearson product moment correlation coefficient (r). Where positive values indicate a positive association between two categories, and negative values a negative association.

Knowledge Category A	Knowledge Category B	r	p*	Comment
Commercial fishers	Long-time residents	+0.36	0.106	Those whose knowledge was based on commercial fishing also had knowledge due to being long-time residents.
Traditional knowledge	Food fishers	+0.58	0.006	Those with traditional knowledge also had knowledge from food fishing.
Commercial fishers	Independent consultants	-0.36	0.106	Independent consultants and scientists did not have knowledge through commercial fishing.
Independent consultants	Long-time residents	-0.48	0.030	The knowledge of independent consultants and scientists was not a result of being long-time residents.

* probability

5.3.3.3 Data clustering

When the data were analyzed by attribute using the ArcGIS 10.1 High/Low Clustering Getis-Ord General G statistic method (fixed distance band), high and low value polygons were found to cluster rather than distribute randomly. The z-scores for attributes biodiversity ($p<0.01$), cultural ($p<0.05$), scientific ($p<0.05$) and therapeutic ($p<0.10$) were significant, indicating that experts generally agreed as to where the most important locations were for those 4 attributes. The remaining attributes were not significantly clustered. However, when the data were analyzed by expert categories, some internal clustering was detected. For example, consultants/scientists, traditional knowledge, and marine patrolmen had strong internal agreement (as measured by the sums of squares) as to where the most and least important locations were found (see Table 5-6).

Table 5-6. Z-scores generated by the High/Low Clustering Getis-Ord General G statistic method to detect agreement among high and low value polygons

	Bio	Eco	Sub	Rec	Cul	Sci	Fut	Lif	Exi	Sce	Spi	The	SS
Long. Res.	2.7	0.7	1.0	0	2.6	2.5	0.6	1.1	1.7	0.2	0.7	-0.5	27
Com. Fisher	0.7	-0.4	1.3	-0.5	-0.6	2.2	-1.4	1.4	-1.4	-0.8	1.1	-0.3	16
M. Leisure	1.2	2.5	0.8	0.7	1.4	-0.7	0.9	0.5	0.9	-0.5	0.7	-0.3	14
Traditional	3.6	0.2	1.2	-0.4	-1.0	3.4	-0.2	-1.0	2.3	-1.3	-0.5	1.9	39
Cons/Sci.	2.5	1.5	4.7	0.2	2.2	-0.1	1.6	-0.8	0.5	0.5	1.3	1.4	43
Patrolman	3.6	1.8	1.1	1.4	0.7	-1.4	-0.5	2.2	-0.8	0.7	-0.7	1.6	31
Food Fisher	-0.3	-0.1	1.9	0.8	-1.0	3.1	0.4	-1.3	2.2	-1.3	-0.3	1.2	25
Sport Fisher	-0.7	2.4	1.5	1.2	-	0.6	-1.1	-	-	-1.0	-	-	13
NGO	3.4	1.7	-0.3	0.3	0.5	-1.7	-0.2	0.2	-0.7	2.2	-	-	23
ALL	4.1	1.5	-0.2	0.2	2.6	2.1	-0.6	1.4	0.8	0.3	1.2	1.7	38

z-score: < -1.65 or $> +1.65$ ($p < 0.10$); < -1.96 or $> +1.96$ ($p < 0.05$); < -2.58 or $> +2.58$ ($p < 0.01$).
SS = sums of squares; Bio = Biodiversity; Eco = Economic; Sub = Subsistence; Rec = Recreation; Cul = Cultural; Sci = Scientific; Fut = Future; Lif = Life Sustaining; Exi = Existence; Sce = Scenic; Spi = Spiritual; The = Therapeutic

5.3.3.4 High value locations and hotspots

Maps displaying concentrations of allocated token points for each of the 12 attributes considered are shown in Figure 5-2. The concentration of points (dark red areas) is visually highest in the biodiversity and economic maps demonstrating the relative importance of these attributes. Overlaying and summing the 12 maps produces a map with the values of all attributes combined as shown in Figure 5-3. Distinct concentrations of points (dark red areas) highlight important areas in the study region.

The recurrence of high and low value points is the basis of detecting hotspots. Hotspot analysis conducted on each of the 12 attribute datasets produced maps with similar conclusions to the cell statistics approach above (see Figure 5-4). Distinct hotspots (red areas) are visible for each attribute. Hotspot analysis on the datasets of all 12 attributes combined produced a final map (Figure 5-5). The map demonstrates the predominance of high value clustering over low value clustering (hotspots). A scatter of important marine spaces was detected, predominantly along coastlines through the entire study area.

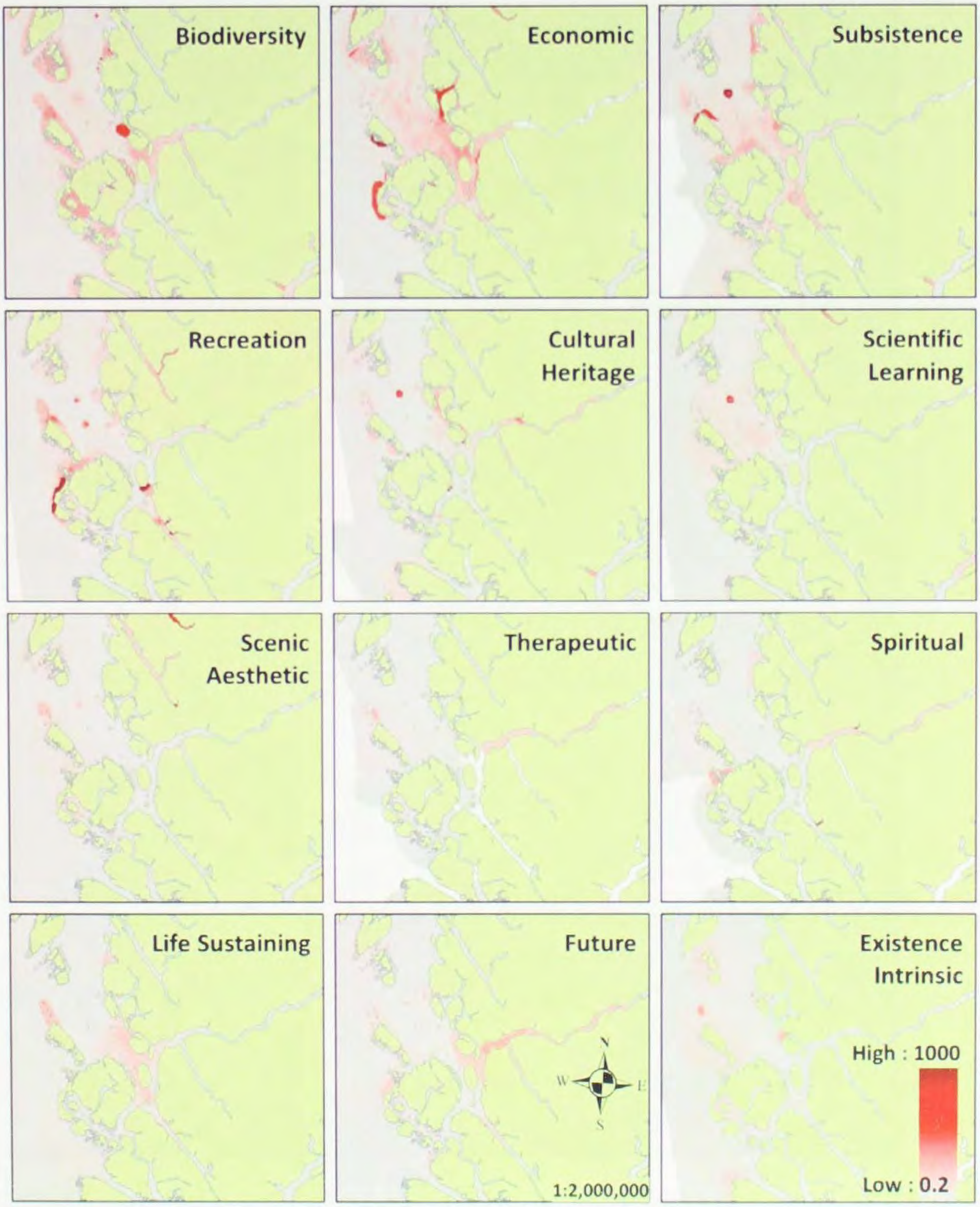

Figure 5-2. Maps displaying concentrations of allocated token points for each of the 12 attributes considered. Values are expressed in points per 100 km^2. The concentration of points (dark red areas) is visually highest in the biodiversity and economic maps demonstrating the relative importance of these social-ecological attributes.

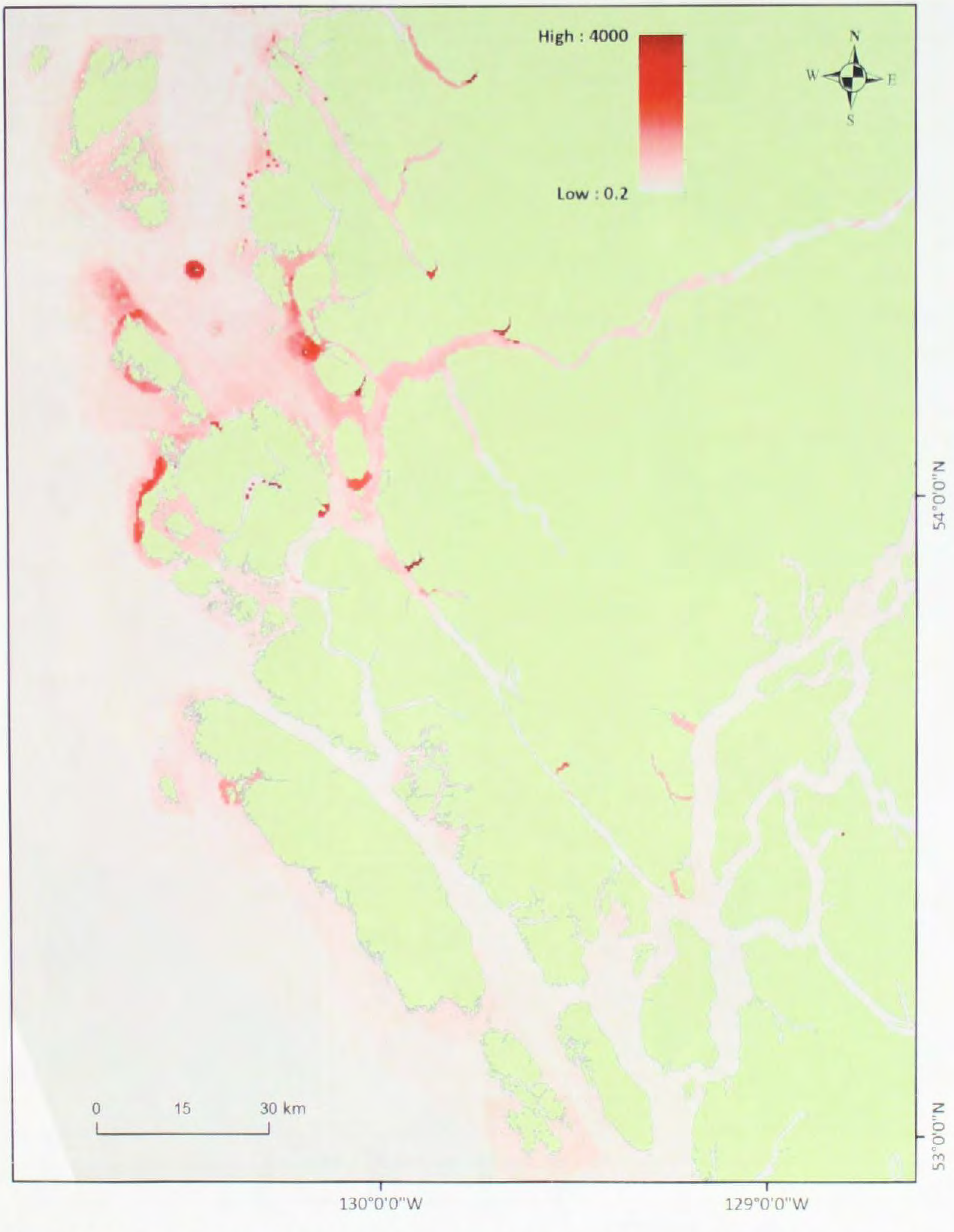

Figure 5-3. A map of the combined values of all 12 attribute maps. Distinct concentrations of points (dark red areas) highlight important social-ecological areas in the study region. Values are expressed in points per 100 km^2.

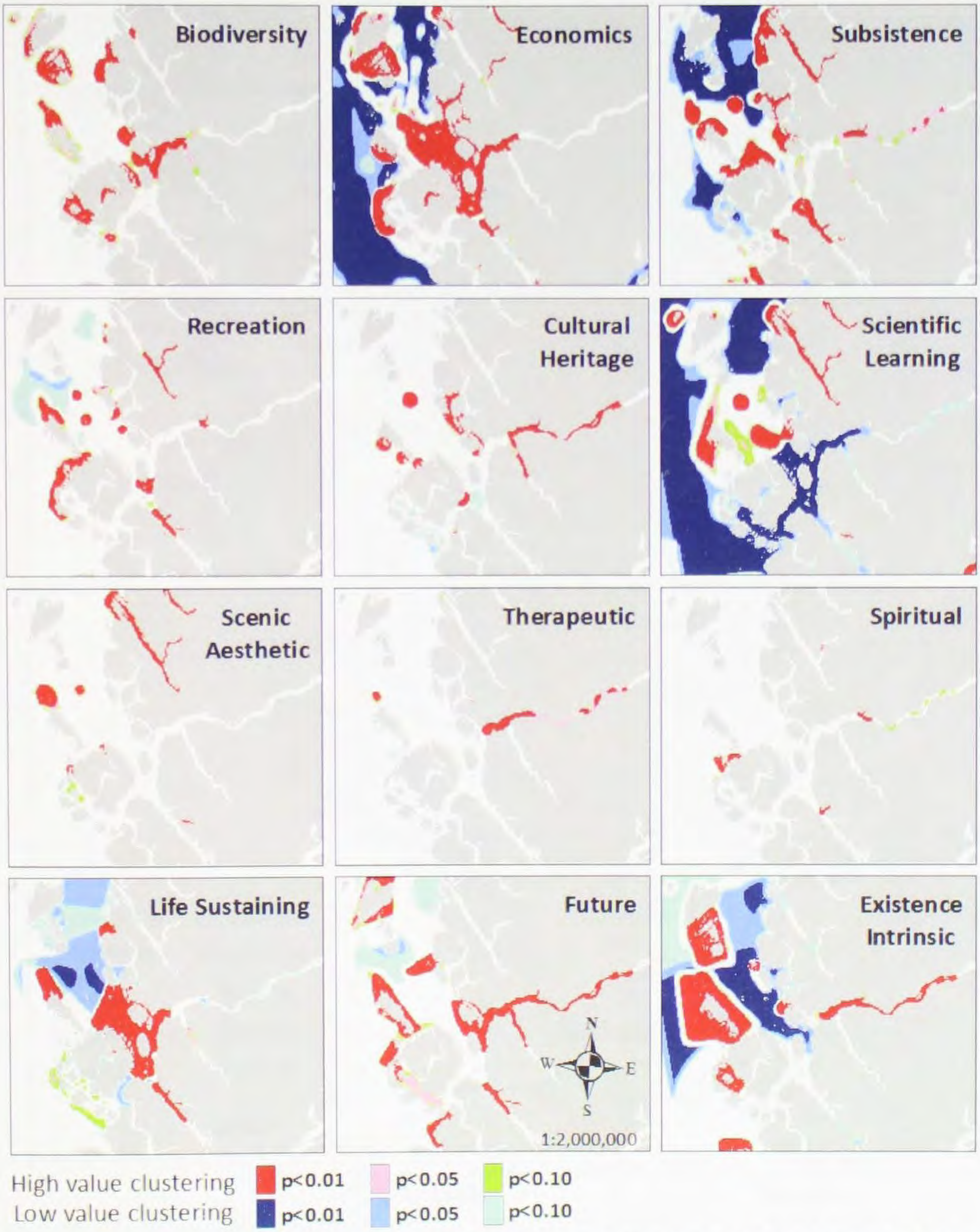

Figure 5-4. Social-ecological hotspot maps produced using the Getis-Ord Gi* spatial statistic applied to each of the 12 attribute datasets. Distinct high and low value hotspots are visible for each attribute.

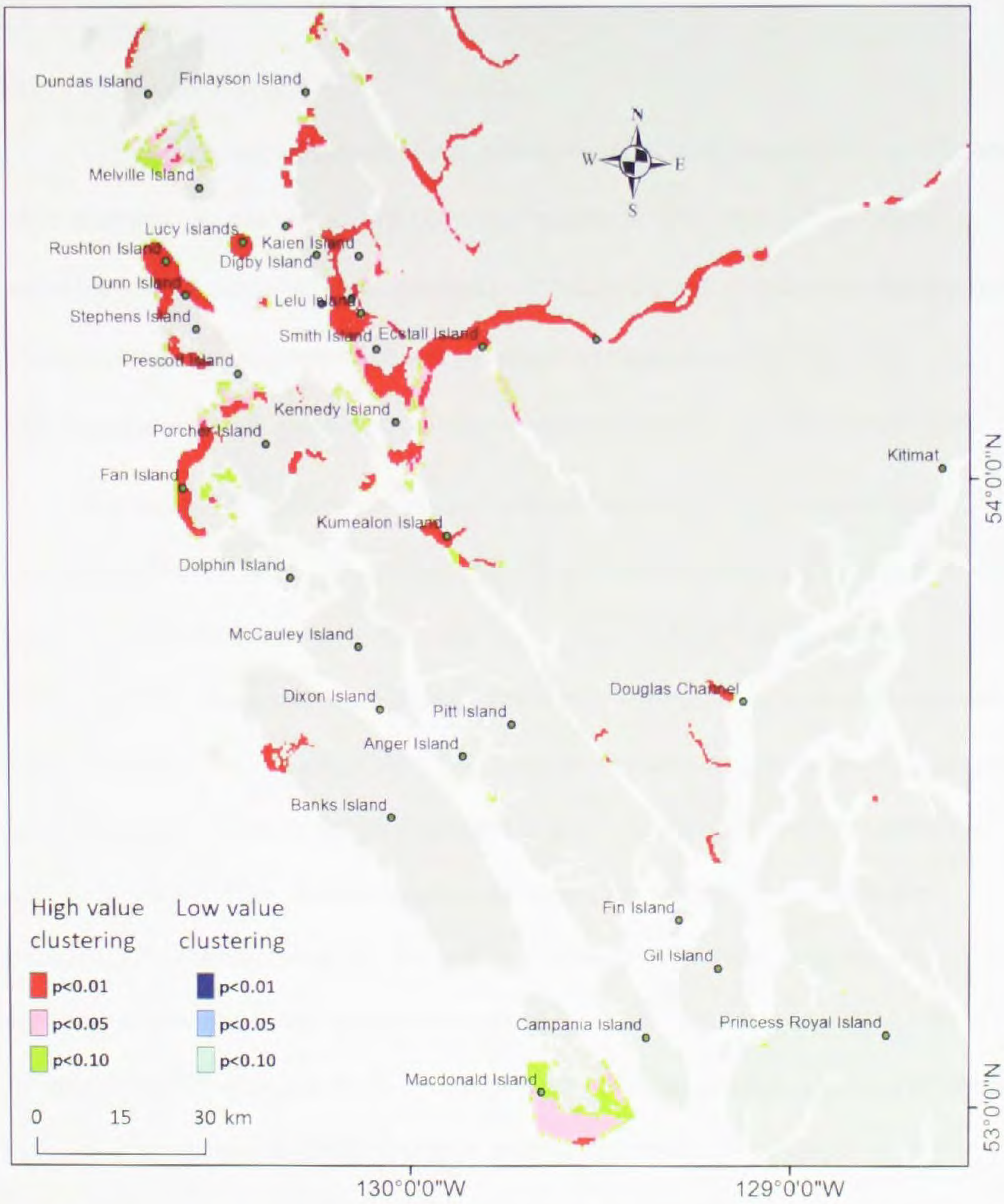

Figure 5-5. Hotspot map produced by applying the Getis-Ord Gi* spatial statistic to the combined datasets of all 12 attributes. The map demonstrates the predominance of social-ecological high value hotspots and displays a scatter of important marine spaces, mostly along coastlines of the study area.

5.4 Discussions

5.4.1 Important locations

xGIS was capable of tapping the knowledge-base of local experts to detect important social-ecological marine spaces. The combined knowledge of local experts produced a comprehensive picture of spatial importance. Although the entire study area was deemed 'important' (i.e. received near complete polygon coverage), when the data were quantitatively analyzed, specific places clearly emerged as being of greater importance.

This study detected biodiversity as the most important attribute considered; receiving the highest point allocation and the highest agreement among experts as to the locations of biodiversity hotspots ($p<0.01$). These results suggest that the spatial distribution of biodiversity hotspots is accurate. Other attributes, such as economics and subsistence, also received a high point allocation. The experts did not, however, generally agree among themselves as to the spatial distribution of important hotspots for those attributes ($p<0.10$). Thus, though deemed important, the spatial accuracy of those attributes is less certain. Similarly, the spatial distribution of cultural, scientific and therapeutic hotspots, though generally regarded to be less important attributes, were in agreement among experts ($p<0.01$, $p<0.05$, $p<0.10$, respectively). The remaining attributes were not in agreement ($p<0.10$); though in some such cases there was some internal agreement (e.g. sports fishers internally agreed as to the locations of economic hotspots). In cases where spatial agreement was not significant, two interpretations might be considered: (1) expert knowledge is not reliable for that value attribute, or (2) expert knowledge is complementary rather than contradictory. If the assumption held is that participants are indeed 'experts' and their knowledge is valid and accurate, then the latter interpretation might be more appropriate.

The analyses also provide for specific insights with respect to the spatial distribution of marine-based activities, as well as, inferences as to the importance of those locations to the well-being of people. The hotspots detected by all attributes combined (Figure 5-5) provide a distinct view of the overall distribution of importance. The final analysis could serve as a starting point of discussion among stakeholders to examine possibilities of risk avoidance to the most important locations and the sensibility of risk acceptance to the least important. The hotspot maps (Figure 5-4 and Figure 5-5) are perhaps more defensible due to their statistical rigor. However, the token point density maps (Figure 5-2 and Figure 5-3) may be more comprehensible to the general public.

5.4.2 Chain referrals

The chain referral method used in this study was found to be an effective means of identifying and soliciting expert participation for most knowledge categories. Prior acquaintanceship between the first round of participants and the investigator was found to be an important factor as 'cold calls' to unfamiliar experts resulted in very few positive responses. Subsequent rounds of invitations to individuals recommended by previous participants were also relatively successful (regardless of prior acquaintanceship), demonstrating the importance of inferred familiarity evoked by one expert recommending another.

A potential challenge arising from the method is that of referrals veering disproportionately towards one or two expert categories at the exclusion of others. In this study, for example, 3 expert categories (long-time residents, commercial fisherman and marine leisurists) made up 56% of the surveys. This may be due to the natural demographics of these experts in the region or to the tendency of experts knowing and, thus, referring like experts. Future studies might consider the appropriateness of either setting goals for the numbers of participants sought across each knowledge category, or weighting the data to balance the influence of each knowledge category in the analysis.

Some knowledge categories, such as *community health workers*, had no referrals. This may be an indication of the rarity of health experts with marine spatial knowledge or, alternatively, an indication of disconnection between environmental and health professionals; where the former is unfamiliar with the knowledge of the latter. Given the complex cross-linking relationship between human and environmental health, this gap should be an important consideration in future studies (also see Section 5.4.5 below).

5.4.3 Maximizing data capture during mapping

Brown and Reed (2009) recommend that participants should map approximately three to six points per attribute; yielding about 78 locations per participant (assuming 13 attributes are considered). They argue that providing fewer points risks not allowing a meaningful set of locations to be identified, while providing too many points risks 'littering' the map with every possible location at the expense of being unable to differentiate the 'truly important' from the 'common'. The xGIS method permits participants to identify as many locations as they know about; thus collecting their 'entire' body of knowledge. However, in the second part of the survey, they are asked to highlight the most important of those locations by applying the finite tokens provided. Works by Brown (2005) showed that when participants were given a choice to map infinite points, many ultimately mapped a relatively small number, particularly those less familiar with the study area. Thus, unrestricted mapping is not expected to erode the quality of the data by proliferating the number of low accuracy data points.

5.4.4 Determining the relative importance of the value attributes

PPGIS methods often provide participants with 100 points per value attribute to allocate to important locations on the map. Participants will often allocate the full 100 points associated with one value attribute, move to the next and do the same, and so on. This is procedurally simple, which may be important in certain PPGIS studies, but two challenges arise as a result: (1) every attribute receives a score of 100 points, making it impossible to compare attributes to one another to determine the relative importance of each; (2) participants, compelled to fully complete the survey, may tend to allocate all tokens, including those associated with attributes they may be unfamiliar with. Brown (2012b p.293) argues that "a well-conceived PPGIS system would not force or even encourage responses from participants for spatial variables that are beyond the intellectual or experiential capacity of the participant" as this can introduce greater spatial error to certain attributes. This source of error is avoided in xGIS by permitting participants to allocate their pool of token points disproportionately across attributes, thus allowing them to emphasize those that are most important and increasing the likelihood that participants will focus on what they know.

5.4.5 Assumptions and weaknesses

Perhaps the most significant assumption and source of uncertainty in this study is whether the knowledge-base of the community of experts invited to participate in this study is valid and accurate. Related studies in other jurisdictions demonstrate that local knowledge is in fact reliable, especially among local experts. For example, Brown et al. (2004) showed that community members who 'regularly traverse' an area, had a relatively broad and accurate knowledge of important locations. They argue for the difference between those who 'care' about a place, but have little knowledge about it, and those who 'know' about the place, and suggest the frequency of area use (familiarity) to be a good, though imprecise, indicator of such knowledge. Brown and Reed (2011) measured the difference in accuracy between the spatial knowledge of experts and that of the entire community (the general public). They found significantly lower spatial error among experts: 14.5% error for random households, compared to 7.4% for conservation area visitors and 5.9% for the volunteer public.

Additional uncertainties include whether the data are adequately complete to draw conclusions, and whether the final maps produced represent marine spaces that are truly most important to the social-ecological system. Further study is needed to compare the results of this work to other related data sources including marine economic data, biophysical studies, etc. to examine the degree of agreement. Other approaches to validation might entail presenting the hotspot maps produced to diverse local experts for additional critique.

A final consideration to be noted is that of an imposed methodological bias in participant selection. The xGIS methodology requires individuals to possess a specific form of knowledge expertise in order to become recognized and invited to be participants. This was referred to as local expert spatial knowledge. The consequence of this bias is that the knowledge of certain experts (e.g. community health workers), whose knowledge may not be spatial in nature, were not included in the data collection. This orientation to spatial knowledge was purposely selected in order to create the spatial framework needed. Future research may examine how non-spatial forms of knowledge may also be included in the methodology.

5.5 Conclusions

This study has demonstrated that xGIS is an effective method for collecting and analyzing local knowledge expertise to detect and value important social-ecological marine spaces; producing hotspots with a high degree of spatial statistical significance. The analysis also determined the relative importance of the value-attributes considered. The xGIS tool is intended for application at the local or regional level, as it is based on experts having detailed knowledge of the local or regional social-economic or biophysical environment. With increased global pressures on resource development and urbanization, together with significant social and ecological data gaps, poorly understood social-ecological processes and the urgent need for solutions, xGIS can serve as a useful first filter of the spatial distribution of important marine locations and a starting point for stakeholder engagement in the planning of risk distribution from development activities. The results can be useful resources for environmental managers and planners at local or regional levels around the world.

Chapter Six. Environmental impact scenarios in coastal British Columbia: Opportunities for integrated analysis of social, ecological and economic effects

6.1 Introduction

Coastal communities can rely heavily on the services of marine environments for a range of social well-being and health benefits (see Assembly of First Nations 2003, Brotherston and White 2006, Molnar et al. 2009, CRIFC 2010a, Chan et al. 2011). Current projects and new proposals for development and use of coastal waterways (see examples for northern British Columbia, Canada in Carleton Ray and McCormick-Ray 2013) could cause significant impacts to marine ecosystems and, thus, the social and ecological benefits that they provide.

One of the challenges to better managing these impacts is that of the difficulties of measuring their effects on important ecosystem services in a spatial setting. A number of approaches to measurement have been proposed in the literature. Some are based on economic theory (see Hadley et al. 2011), while others on ecological principles (see Gregr et al. 2012) or socio-cultural data (see Stagl 2007). And some are attempted integrations of approaches (see examples of integrative approaches in Shmelev 2010, Chan et al. 2012, Saarikoski 2012). Integrative approaches can vary greatly in the complexity of application, the data and resources they require, and their degree of acceptance.

Approaches that are based on locally available data can be useful from the perspective of cost and time efficiency. One such approach involves the application of spatial statistical analyses to local economic, ecological and social data to determine locations of high value or high incidence clustering (see Chapter 4). The areas identified by this approach are considered important for the features being measured, and can be compared and interpreted in an integrated framework. Data often available for this purpose in many coastal communities include commercial fishing data (DFO 2013e), ecologically and biologically significant area (EBSA) data (Clarke and Jamieson 2006b) and local ecological knowledge (LEK) data (Booth et al. 2005-2008). These data have certain gaps and limitations. Yet they are the data often used by federal, provincial, municipal and First Nations agencies involved in coastal management and planning frameworks in BC.

Another approach to integrated analysis is expert-informed Geographic Information Systems (xGIS). The xGIS approach is based on the collection of local knowledge expertise to detect and value important social-ecological marine spaces based on a range of factors (see Chapter 5). The approach allows for the determination of the relative importance of each of the criteria considered (i.e. both use and non-use values of ecosystem services) (Mahboubi et al. 2015).

These approaches to integrated impact analysis have not been demonstrated in practice and their utility to environmental planners and managers is uncertain. Thus, the objective of this study was to create a modelled scenario of environmental impact and assess the broad economic, ecological and social implications of that impact by comparing it to the social-ecological maps produced by each of the two approaches to integrated analysis discussed above.

Any number of impacts could be used to suffice this objective including, natural phenomenon such as geological processes (an earthquake) or algal blooms, or anthropogenic impacts such as development activities or accidents at sea. The hypothetical impact selected for the purposes of this study was a modelled oil spill at sea. The oil spill scenario was selected as the impact of choice for two reasons: (1) during the period of this research, the EA process for the proposed Enbridge Northern Gateway Project (ENGP) was underway, thus an oil spill scenario was of particular relevance to the study area and (2) the oil spill scenario was deemed an event of appropriate scale for this study.

To achieve the objective above, three types of analyses were needed: (1) a determination of the quantity of oil to include in the spill model and, thus, an analysis of spill probability; (2) a determination of the geophysical forces to apply to the model (i.e. wind and ocean currents) in order to disperse the oil geographically, and (3) to overlay the hypothetical results produced by the oil spill model, with the spatial data produced by the two approaches to social-ecological mapping above (i.e. the local social-ecological-economic data analysis and the expert-informed GIS analysis). The two latter overlay analyses were then critically examined and compared for their strengths and limitations.

6.2 The Northern Gateway project

Among the projects recently proposed for northern BC, the ENGP has been a particular source of discussion and controversy due to the difficulties of forecasting the benefits and risks in a reliable and accurate manner. The project proposed construction of two 1170 km pipelines between the bitumen mines of Alberta and the coastal community of Kitimat, BC; one line to carry a variety of crude oil products[1] from Alberta to Kitimat, the other to pipe condensate (a gasoline-like mixture of light oil components usually obtained from natural gas production) to Alberta.

It was estimated that approximately 525,000 barrels or bbl (i.e. 83 million liters) of oil would reach Kitimat each day to be loaded onto tankers (Enbridge 2010). The ENGP is expected to use crude oil carriers with a carrying capacity of at least 160,000 DWT (deadweight tonnage) (Baker 2010). At an average of 6.7 bbl of cargo per DWT (UNCTAD 2006), a 160,000 DWT tanker could carry approximately one million bbl of oil. Approximately 220 tankers per year are expected to traverse the narrow passageways of the Douglas Channel and the Inside Passage bound primarily for Asia (Gunton and Broadbent 2013) (see Figure 6-1).

[1] Studies prepared for the project indicated that the majority of shipments would be diluted bitumen, which is a blend of light and heavy oil products (NEB 2014).

The project satisfied the requirements of both Federal and Provincial Environmental Assessment processes so long as it could meet 209 conditions (Canadian Environmental Assessment Agency 2014). Yet despite the approval and conditions, the risks of adverse impacts introduced by the project (be they to the social and ecological services of the marine environment or others), were heavily contended in a broad range of social, environmental, political and judicial arenas (Campbell 2006, Skuce 2010, BC Nature 2014, Coates 2014, Laanela 2014b, Moore 2014).

Figure 6-1. The North Coast of British Columbia displaying the proposed route of oil tanker travel to and from the community of Kitimat (the proposed site of the Enbridge Northern Gateway oil terminal). The locations identified for modelled oil spills are also shown.

6.3 Methods

The Pacific North Coast Integrated Management Area (PNCIMA), a 102,000 km^2 region of ocean, was selected as the broad context for this study. Particular focus was given to the northeast portion of PNCIMA referred to as the North Coast study area of the Marine Planning Partnership (MaPP), an area of approximately 25,000 km^2 (see Figure 6-2). Three oil spill scenarios were modeled within the study area. The sites were located along proposed tanker routes, one near the proposed Enbridge Northern Gateway loading terminal in Douglas Channel, a second on the southern route of proposed tanker travel and a third on the northern route (see Figure 6-1). The latter two locations were selected as moderate sites of impact: neither the worst-case (i.e. they do not fall in the most prolific areas of ocean values and narrow channels), nor the best-case scenarios. Causes of spills at these sites might include running aground in shallow waters, inclement weather conditions causing the ship to capsize, or collisions with other vessels due to human error.

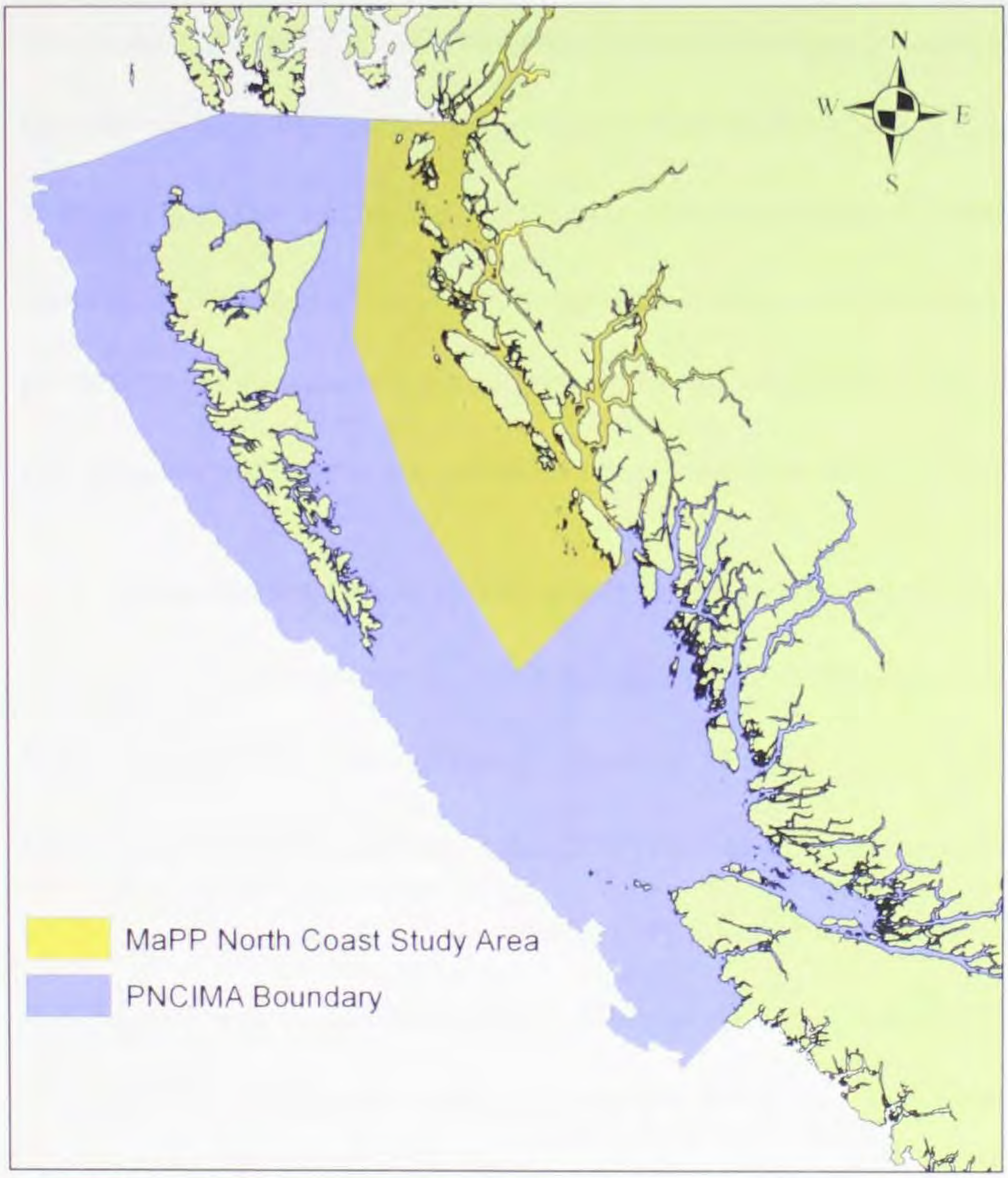

Figure 6-2. The North Coast study area of the Marine Planning Partnership (MaPP) for the North Pacific Coast, overlaid onto the Pacific North Coast Management Area (PNCIMA) LOMA.

Etkin (2009) reports that over the 10 year period, 1998-2007, oil tankers and barges travelling in American waters spilled on average 9,027 barrels (bbl) of oil per year, distributed across an average of 73 spills (>1 bbl). In relation to the quantity of oil carried and the distance transported, these spills equate to an average annual spillage of 5.28 bbl per billion bbl-miles (bbl per Bbbl-miles) transported. It is estimated that the ENGP would handle approximately 525,000 barrels of crude oil per day (annually 0.19 Bbbl loaded onto approximately 220 tankers). Tankers would travel from Kitimat, B.C. to Asian markets, such as Hong Kong, Taiwan and Xingang China, a mean approximate distance of 6700 miles. This equates to 1270 Bbbl-miles of oil transported. Based on the American spill rate of 5.28 bbl per Bbbl-miles transported, the expected spill rate of the ENGB would be 6700 bbl per year. This quantity was used in the modelled oil spill scenario discussed below.

Anderson and LaBelle (2000) report spills from Alaskan North Slope tankers, at sea and at port, to be 1 spill per 2.17 Bbbl handled (1985-1999 data). At 0.19 Bbbl of oil handled by the ENGP per year, this equates to 1 spill per 11 years (i.e. the spill return period). Alternatively, Gunton and Broadbent (2013) estimate the spill return period for the ENGP to be 23-196 years at sea and 15-41 years at port. Another tool for estimating spill risks is the US Oil Spill Risk Analysis (OSRA) model. The model estimates a 95.3-99.9% probability of spills >1000 bbl at sea with a spill return period of 7-17 years, and a 65.1-98.2% probability of spills >10,000 bbl with a spill return period of 13-48 years (Gunton and Broadbent 2013). These values conflict heavily with the spill return periods estimated by the ENGP EA: 250 years at sea and 62 years at port. This research assumed a spill return period of 14-years at sea (i.e. the mean of the lower values of the range estimates above, excluding the ENGP estimate).

Assuming a spill rate equivalent of 6700 bbl per year, occurring once in 14 years rather than yearly, then the one in 14-year spill would amount to 93,800 bbl. Historical data (1985-1999) show the size of larger spills (i.e. >1000 bbl) to range from 68,700-89,900 bbl at sea and 5600 bbl at port (Anderson and LaBelle 2000). ENGP similarly estimates average spills at sea and port at 56,700 bbl and 1,575 bbl, respectively (Gunton and Broadbent 2013). For the purposes of this research, the mean of the spill estimates above (i.e. 77,000 bbl) was used.

Modeled simulations of oil dispersal at sea were conducted using the General NOAA Operational Modeling Environment (GNOME™ 1.3.9); a standard spill–trajectory model supporting the NOAA standard for 'best guess' trajectories (i.e. where the spill is most likely to go) and 'minimum regrets' (i.e. the uncertainty bounds) (Beegle–Krause 2010). The best guess estimate assumes that (1) winds continue to blow steadily at the speed and from the direction input, and (2) the data in the location file accurately represent the current patterns.

Note: the GNOME model is intended for educational and planning purposes only. It is not intended to govern response decisions in an actual oil spill. Thus, the oil spills modeled in this study are entirely hypothetical. The actual movement of oil, should a spill occur, would be governed by the winds and currents existing at the time of the spill, as well as, the actual quantity and grade of oil spilled. Furthermore, the distribution of the oil may continue beyond the 72 hours modeled in this study. The spills modeled in this study were merely to illustrate how integrated analysis might be approached under conditions of impact. The modelled oil spill results should not be relied on for decisions.

Ocean current data, required for the model, were acquired by two methods. First, detailed ocean current measurements gathered during the ENGP EA process for the period November 12, 2010 were applied to the spill scenario located in Douglas Channel. These data did not, however, extend to the other two sites. Thus, ocean currents for the remaining sites were estimated using the Global Real-Time Ocean Forecast System (GOODS 2015) for the period February 15, 2015. Hourly wind data for the selected simulation dates were accessed from buoy C46181 and C46183; the closest climate stations to each spill site, available from Fisheries and Oceans Canada (DFO 2015). The spills were modeled over a 72 hour period with 77,000 bbl of medium crude oil (similar grade to that planned by the ENGP) spilled at each of the three sites.

The results of the GNOME model (i.e. the modelled dispersal of oil based on the physical and chemical inputs selected) provided a hypothetical representation of the area of impact (see Figure 6-3) and were compared to spaces identified as important within the study area based on the following analyses:

(1) Commercial fishing spatial economic analysis –see Section 4.3.1, page 114

(2) Ecologically and biologically significant areas (EBSA) analysis –see Section 4.3.2, page 116

(3) Local ecological knowledge (LEK) data analysis –see Section 4.3.3, page 117

(4) Expert informed GIS (xGIS) –see Section 5.2, page 144

6.4 Results

Figure 6-3 presents the output produced by GNOME depicting the hypothetical dispersal zones (best guess and minimum regrets) based on estimated inputs. Three oil spill scenarios are presented.

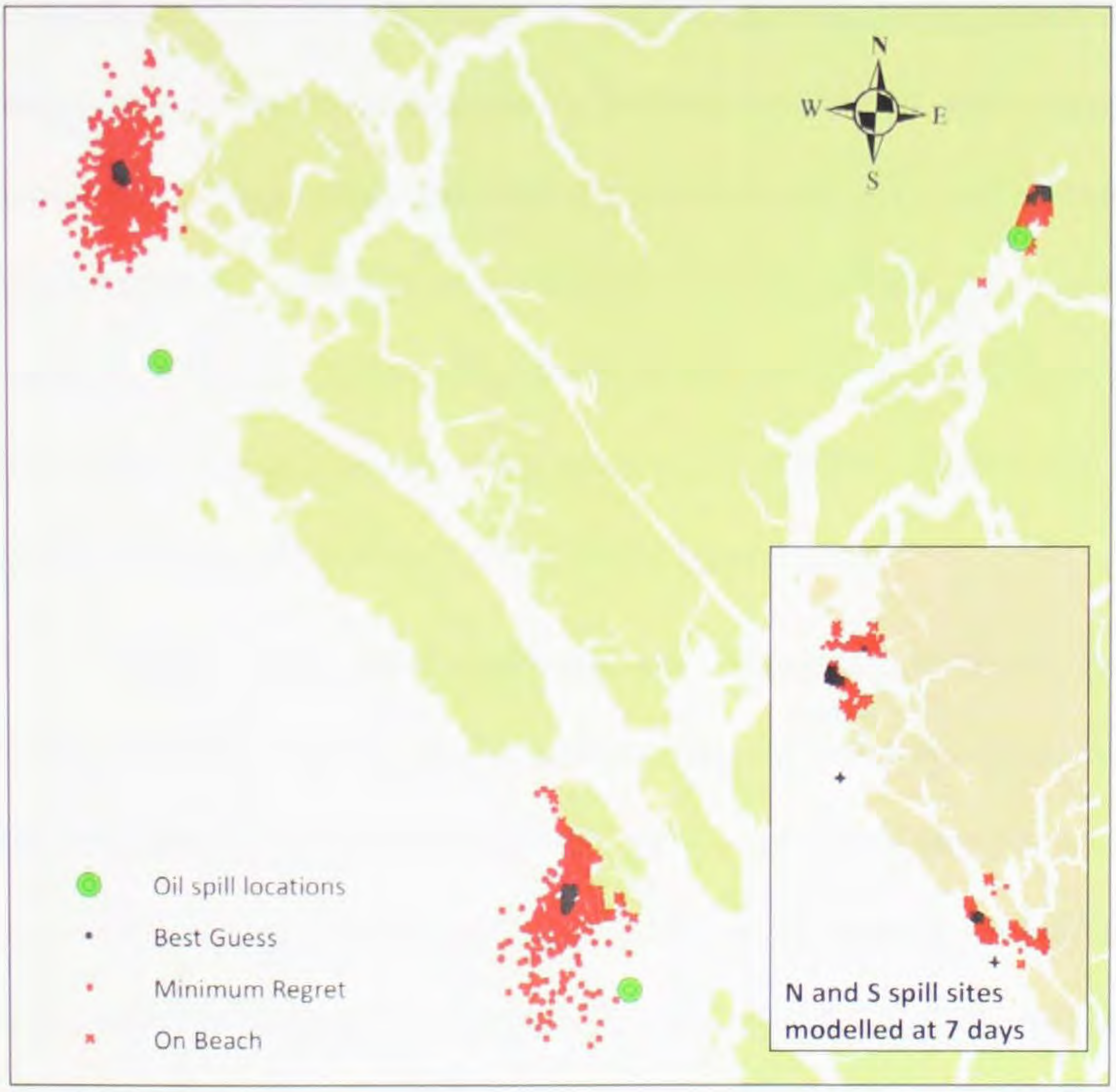

Figure 6-3. A hypothetical depiction of oil dispersal for three modelled oil spills using the General NOAA Operational Modeling Environment (GNOME™ 1.3.9) including best guess trajectories and minimum regret solutions (uncertainty bounds). The dispersal is based on the following estimated inputs: 77,000 barrels of medium crude spilled; 72 hour and 7-day spill dispersal; variable currents estimated by the Global Real-Time Ocean Forecast System for February 15, 2015; variable February winds estimated based on historic data from buoy C46181 and C46183. Note: these are hypothetically modelled spills intended to illustrate a scenario of environmental impact and should not be relied on for decisions related to actual oils spills.

The economic analysis conducted in Chapter 4 demonstrated the spatial distribution of commercial fishing harvest values within the PNCIMA. Examining these together with the modelled oil spills (Figure 6-4a) shows that the areas of ocean affected by the southern, northern and terminal spills overlap with areas supporting approximately \$445,000, \$216,000 and \$55,000 of annual commercial fishing, respectively (Table 6-1). In a worst case, a similar-sized spill (approximately 350 km^2) occurring on the highest value commercial fishing grounds (\$500,000 of harvest per 16 km^2) would be associated with approximately \$11 million of annual harvest. Commercial fishing hotspots (i.e. areas displaying statistically significant clustering of high harvest values, $p<0.05$) (Figure 6-4b) collectively comprised an area of approximately 30,000 km^2. There was no overlap between the modelled oil spills and commercial fishing hotspots.

Analysis of LEK incidence data conducted in Chapter 4 produced an LEK hotspot map depicting significant clustering of species occurrences. Four major concentrations of hotspots, ranging from 2500 to 4500 km^2, were detected (see Figure 6-4c). The LEK hotspot map shows the northern spill to affect (150 km^2) of LEK hotspots ($p<0.10$), while the southern spill is found to be nearly twice as large; affecting 280 km^2 of LEK hotspots ($p<0.10$) (see Table 6-1).

Statistical analysis of EBSA data conducted in Chapter 4 resulted in the EBSA hotspot map shown in Figure 6-4d. The modelled oil spills do not overlap with any EBSA hotspots. However, the northern spill does overlap with 168 km^2 of Phase II EBSAs as shown.

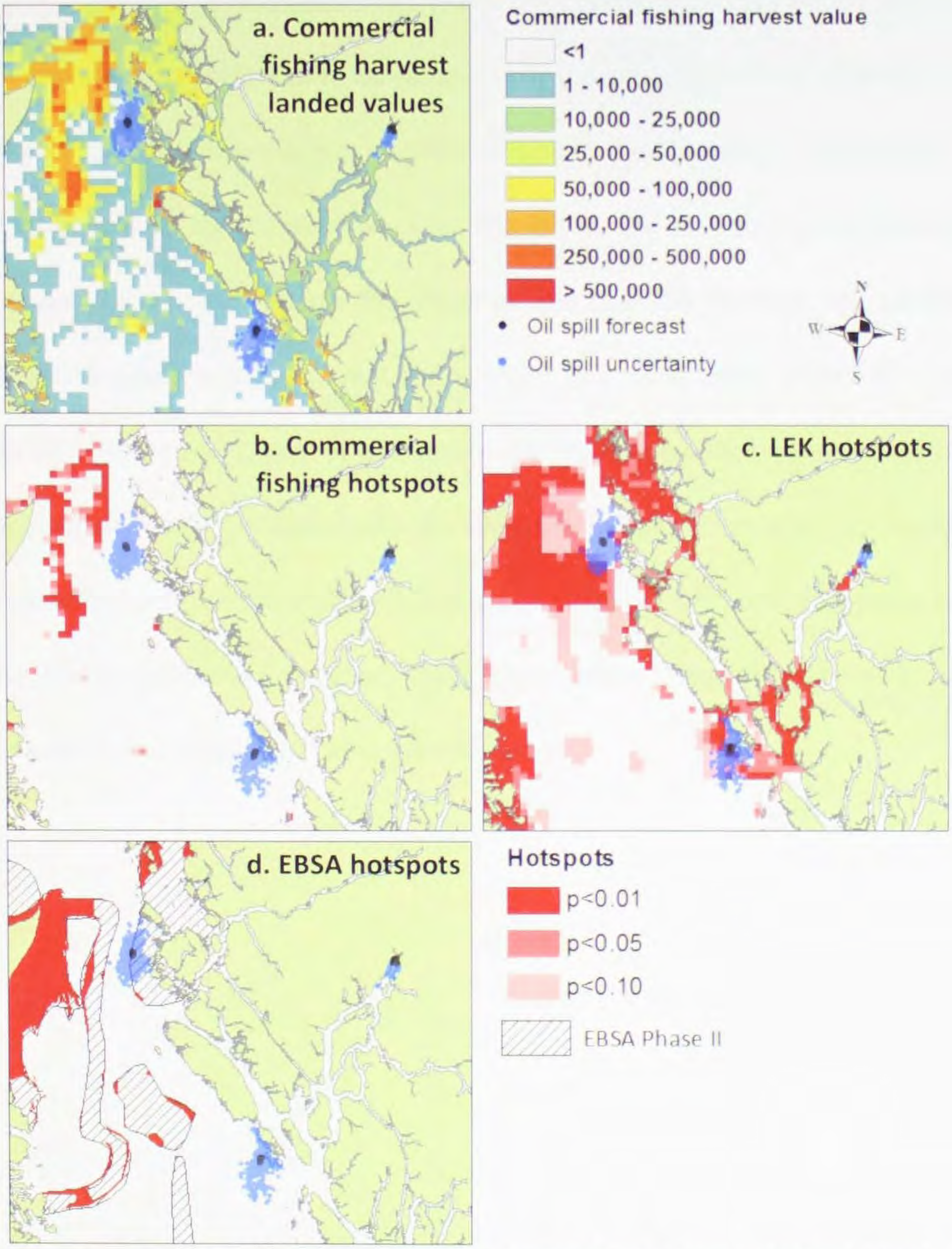

Figure 6-4. Analyses of data for the British Columbia North Coast including (a) landed values of commercial fishing spatial catch data; (b) commercial fishing economic hotspots; (c) species incidence hotspots derived from local ecological knowledge studies; (d) PHASE II Ecologically and Biologically Significant Areas (EBSAs) and EBSA hotspots. Modeled dispersal of three oil spill scenarios also shown, with black defining the best guess area for oil dispersion in 72 hours and blue providing a minimum regret (uncertainty) estimate for dispersion.

The maps produced using the xGIS tool (Chapter 5) showed the spatial distribution of social-ecological hotspots based on a series of landscape value attributes (Figure 6-5). The xGIS economic hotspot map, consistent with the LEK hotspot analysis and the commercial fishing economic analysis, shows that the southern spill will have the greatest overlap (i.e. 114 km^2), nearly four times greater than the northern spill (32 km^2) (Table 6-1). The xGIS map also suggests that the northern spill could affect 36 km^2 of ocean valued for its existence ($p<0.01$) and, to a lesser extent, areas valued for scientific, recreation, future and cultural uses. Comparatively, the southern spill affects an area that has been highly valued for its existence (119 km^2) and future value (28 km^2) and, to a lesser extent, its life sustaining and cultural services. Finally, the terminal spill could affect 47 km^2 of ocean valued for its scientific learning opportunities.

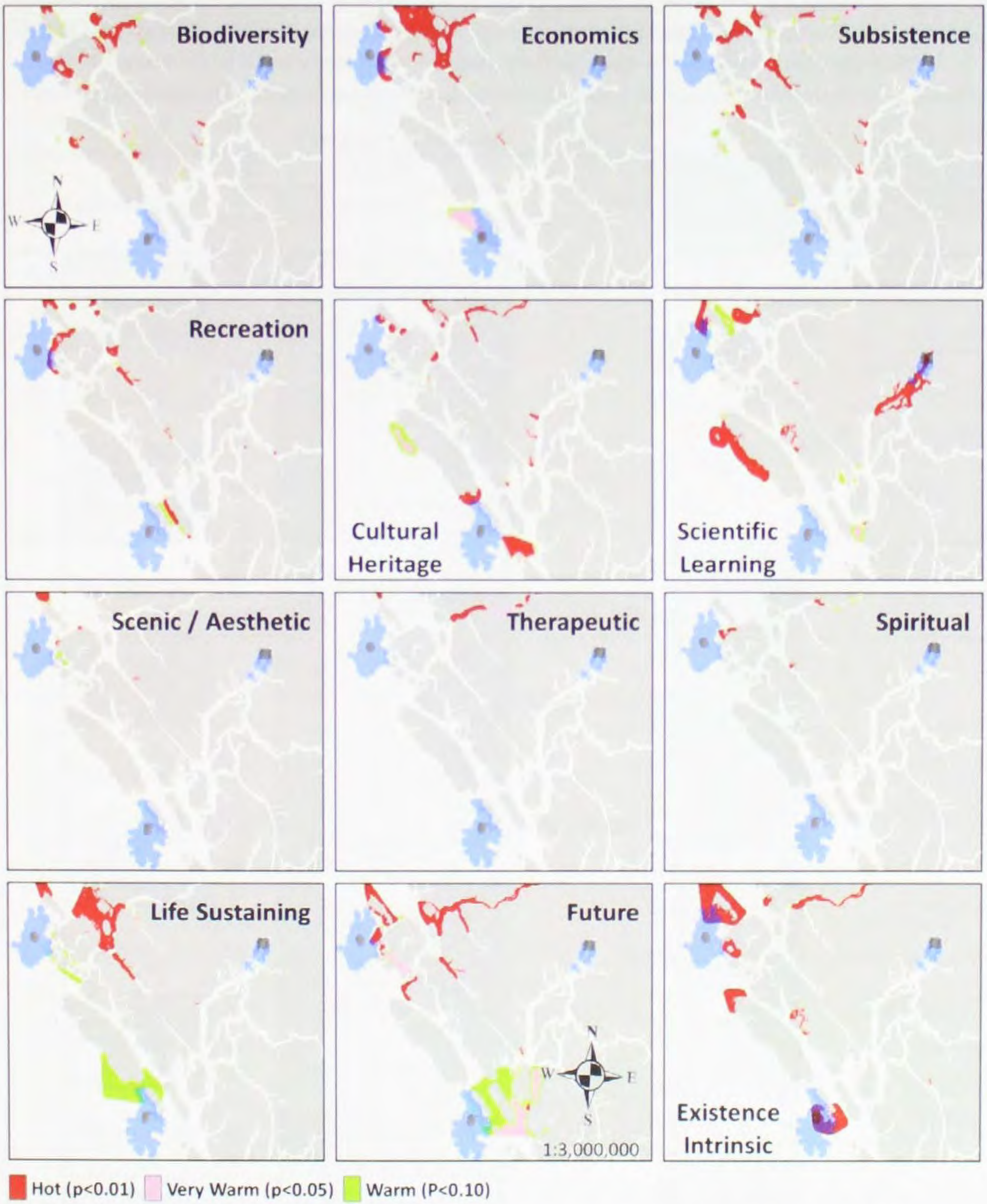

Figure 6-5. Social-ecological hotspot maps displaying hotspots (p<0.01, p<0.05 and p<0.10) for various social-ecological attributes. Three hypothetical oil spills also shown.

Table 6-1. Overlap of the dispersal zones of three modelled oil spill scenarios with harvest values from the commercial fishery and hotspots derived from local ecological knowledge (LEK), ecologically and biologically important areas (EBSA analysis) and expert informed GIS (xGIS) analysis.

			Commercial Fishing	LEK	EBSA	xGIS
Economic		NS	\$216,000	NS: 150 km^2 ($p<0.10$) SS: 280 km^2 ($p<0.10$) TS: Negligible		32 km^2 ($p<0.01$)
		SS	\$445,000			114 km^2 ($p<0.10$)
		TS	\$55,000			-
Ecological	Biodiversity	NS			NS: 168 km^2 ($p<0.01$) SS: - TS: -	-
		SS				-
		TS				-
	Scientific	NS				Negligible
		SS				-
		TS				47 km^2 ($p<0.01$)
	Life Sustaining	NS				-
		SS				Negligible
		TS				-
Social	Recreation	NS				Negligible
		SS				-
		TS				-
	Scenic	NS				-
		SS				-
		TS				-
	Therapeutic	NS				-
		SS				-
		TS				-
	Spiritual	NS				-
		SS				-
		TS				-
	Future	NS				Negligible
		SS				28 km^2 ($p<0.10$)
		TS				-
	Existence	NS				36 km^2 ($p<0.01$)
		SS				119 km^2 ($p<0.01$)
		TS				-
	Subsistence	NS				-
		SS				-
		TS				-
	Cultural	NS				Negligible
		SS				Negligible
		TS				-

NS – northern spill (328 km^2); SS – southern spill (312 km^2); TS – terminal spill (47 km^2)

6.5 Discussion

The objective of this chapter was to consider a scenario of marine environmental impact (in this case several hypothetical oil spills at sea) in order to evaluate the potential social, ecological and economic impacts of the scenarios in relation to the two integrative approaches to impact analysis developed in Chapters 4 and 5. This objective was shown to be a complex undertaking. Though spatial relationships could be described between the spills and a number of social, ecological and economic values associated with the marine environment, interpreting those relationships in a manner that would allow for their application to environmental planning and management processes was challenged. Yet, demonstrating these relationships is arguably an important and necessary underpinning to better planning and management. The social-ecological-economic and xGIS maps produced and presented above offer the backdrop on which both the spills and the identified ecosystem values affected by those spills could be viewed and interpreted. Interpreting these results is, therefore, the next key challenge.

One of the findings of Chapter 3 was that the integration of health concerns into the EA process may be constrained by poor collaboration among the actors involved, including government and non-government sectors, as well as the public. It was argued that the complexity of the social-ecological system necessitates collaborative approaches in order to overcome gaps in expertise. Without an effective process of collaboration to guide the integration process, the interpretation of the data and the solutions to follow can tend towards the status quo: biophysically focussed analyses with only nominal examinations of social, health and other factors (see section 3.5.1, page 89).

The need for collaboration becomes further evident when the individual analyses are examined critically for uncertainties. For example, the commercial fishing economic analysis above suggests that the three oil spills could potentially affect an area of ocean collectively responsible for approximately $716,000 per year of landed fish value[1]. In a best-case scenario, the spills may occur in the off-season and, therefore, not impact fishing. In a worse-case scenario, a complete fishing closure during the season could result in over $700,000 of losses to the commercial fishery. Determining appropriate compensation based on economic data can be challenging. Past experience with oil spills (e.g. the 2010 BP spill in the Gulf of Mexico, Brennan 2013) and mining impacts (e.g. the Taseko New Prosperity mine project, Leahy 2015), suggest that such compensation will likely fall substantially short of addressing the breadth of factors affected.

Similarly, an examination of the LEK analysis (Figure 6-4c) suggests that all three spills could affect important social spaces. Important LEK spaces were identified due to frequent reporting of species presence via the LEK interviews. The area affected by the southern spill, for example, was reported to have salmon, ground fish, sea urchins, sea lions, kelp and sport-fishing uses. It is not clear, however, why those species make those spaces important; whether they are valued economically (i.e. as a source of income or for subsistence), socially (for recreation or cultural practices) or ecologically (for biodiversity). This uncertainty limits opportunities to plan for impact management.

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[1] It is noteworthy that ascertaining the actual economic impact of oil spills at these locations to the commercial fishing industry will depend on the season of impact (i.e. with respect to the fishing season), the species of sea-life examined (i.e. shellfish may suffer greater contamination than pelagic species), as well as, social market perceptions (i.e. the willingness of the public to consume sea life from affected regions).

The EBSA hotspots (Figure 6-4d) suggest that the northern spill could have potentially important ecological consequences, affecting the Chatham Sound EBSA which is recognized for its extensive upwelling and coastal tidal mixing processes, and significant aggregations of species such as Dungeness crab and green sea urchins, and a large diversity of shrimp species (Clarke and Jamieson 2006b). However, the determination of how impacts to these ecosystem services should be integrated into an overall integrated analysis framework is uncertain.

The landscape value hotspots derived by the xGIS approach (Figure 6-5) provided a more detailed perspective of the potential social-ecological consequences resulting from the spills. For example, the northern spill was shown to potentially affect areas of economic importance that were also valued for their existence. The spill was also in proximity to important scientific, recreational and cultural spaces. Similarly, the southern spill affected economic areas valued for their existence, but also for future and cultural uses. The area affected by the terminal spill was important for the opportunity of scientific learning. Despite these advantages, the xGIS approach has not been adequately tested in practice and it is, therefore, uncertain whether the integrated analyses provided by xGIS will pass the a priori threshold of proof among stakeholders. Nevertheless, compared to the previous analysis, the xGIS approach was found to produce finer scale results over a broader range of attributes that were better integrated (see section 7.3 for a discussion on the strengths and limitations of the two approaches to spatial analysis).

What is clear from examining the range of analyses above is that the affected areas are complex - associated with various layers of ecological, social and economic values. Thus, risks posed to these areas would be expected to trigger layers of response from the related actors. In view of this complexity, the imperative for collaboration to interpret impacts becomes further evident. The question then is that of the process necessary to appropriately achieve integrated analysis in practice.

6.6 Integrated analysis in practice

As discussed above, the findings of Chapter 3 showed that approximately a third of the literature reviewed found that the institutionalization of the integration process, together with the leadership of government to advance the process, were key requirements to health considerations becoming better applied and integrated into the EA. These are viable options in the Canadian context as the EA process is already legally institutionalized (in the CEAA), and the mandate for integrating broader health consideration already embedded in that legislation. In view of these legal obligations, the responsibility and onus of leadership may be appropriately placed in the sphere of government.

Manuel-Navarrete et al. (2006) found sustainability outcomes to be unconvincing from either of two extreme approaches -use bans and science-based, top-down management on one end, and community-based, multidisciplinary management on the other. The challenge then is that of creating a process that is balanced: occurring within the government sphere, while facilitating consideration of social, health, economic and ecological factors in a collaborative environment among the actors. The factors considered in the two approaches applied in this research, as well as their associated data layers, disciplines and related actors are shown in Table 6-2.

Table 6-2. The main factors considered in this research, including the layers of spatial data available for each, associated disciplinary specialists, and related government agencies and stakeholders.

Factors	Data layers	xGIS	Local data[2]	Disciplinary specialists	Government sectors	Other sectors
Ecology	Biodiversity Life sustaining Scientific study EBSA[1] LEK[1]	X X X 	 X X	Ecologists Biologists LEK TEK[1]	Fisheries & Oceans Canada Environment Canada Parks Canada Ministry of Environment Ministry of Forests	Independent scientists NGOs[3]
Economics	Economic DFO[1] fish harvest	X 	 X	Economists		Proponent
Health/Well-being	Scenic Aesthetic Recreation Spiritual Therapeutic Cultural Subsistence	X X X X X X X	x x x x x x x	HIA practitioners Public health practitioners	Health Canada Local health departments	Affected public First Nations
Legislation	Protected areas		X	Policy makers Lawyers		

[1] May include commercial fishing data (Fisheries & Oceans Canada, DFO), local and traditional ecological knowledge (LEK and TEK) data, Ecologically & Biologically Significant Areas (EBSA) analyses, and legislated data.

[2] Lower case denotes limited or uncertain availability of data

[3] Non-governmental organizations (e.g. World Wildlife Fund, Ecotrust, Suzuki Foundation, etc)

Yet, as noted previously, examining multiple factors in complex social-ecological systems can present certain pragmatic challenges. Similar challenges were encountered by Parkes et al. (2008) in a study focused on the management of complex social-ecological watershed systems. They considered four factors (the ecosystem, social, health/well-being and watershed setting); placing them at the four vertices of a prism in order to illustrate the interactions between them. This allowed for relationships between any two factors to be discernible (as pairs of connected vertices), as were the relationships between any three factors (the faces of the prism) and the collective (integrated) relationships of the whole (the entire prism). The conceptual thinking of this approach is similar to that of the GIS overlay analysis which also considers multiple layers of data, but in a staged approach: (1) layers are compared two at a time, (2) each comparison produces a resultant output describing the spatial relationships between its source layers, (3) the process is repeated until the full suite of layers have been compared.

In the context of this research, notable complexities exist between any pair of factors considered; notwithstanding a complete examination of all factors. Thus, a staged approach to examination may be a useful method of staging the complexity. Figure 6-6 presents a proposed staging of the integration process. The approach entails a process of focus-group discussions among the major actors, including experts from government and non-government sectors and the public (Table 6-2). Stage 1 discussions focus on discipline-specific issues. Stages 2 to 4 progress through increasingly more complex integration analyses, comparing and interpreting any two factors, then three and finally all four factors to achieve a fully integrated analysis.

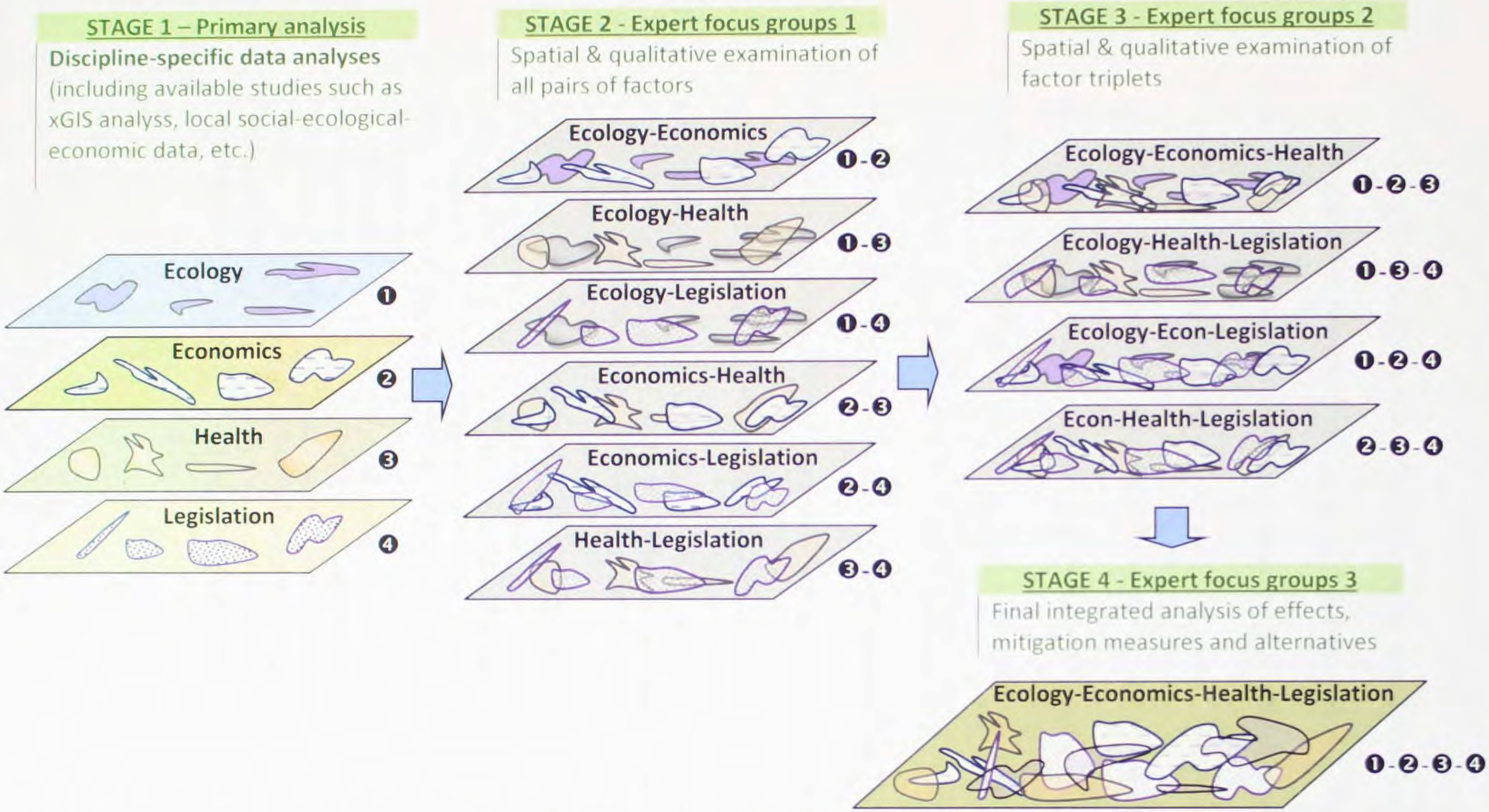

Figure 6-6. Proposed integration schema. A staged approach to data interpretation is proposed accounting for ecological, economic, health and legislated factors. In Stage 1, primary data and basic analyses demonstrate basic spacial relationships with the proposed activity or impact. In Stage 2, combinations of expert focus groups collaborate to interpret pairs of impacts in an integrated framework. In Stage 3, the process is repeated by each of the expert focus groups, now focusing on groups of three impacts. Stage 4 entails a final integrated analysis of all factors.

Despite the apparent practical advantages of collaboration, and the analytic simplification that may be achieved by the staged approach, in the field practice the process may be challenged by conflicts among the mandates and purposes of each of the actors involved. As discussed in Chapter 3, an authentic commitment to the core mandate of the CEAA 2012 (i.e. to protect the environment and human health) is required from the actors in order for the integration process to be effective. Managing the underlying conflicts is paramount to meaningful integration. Table 6-3 depicts the messy backstage factors amongst a number of actors that can underlie collaborative efforts. This commitment to a core human health-centered mandate, unbiased by backstage factors, will be of paramount importance regardless of the process to which the integrated analyses are applied (e.g. local marine use planning, selection of local protected areas, EAs, etc).

Table 6-3. A depiction of the 'messy backstage' of applying the proposed integration schema among the actors shown in Figure 6-6. The actors presented in Table 6-2 each have certain backstage mandates, responsibilities, legal commitments and interests that may influence their participation and, therefore, the process of integration.

Participants	Backstage challenges	Source
Fisheries & Oceans Canada	Serving 15 Acts and 42 Regulations	(DFO 2013a)
Environment Canada	Serving 19 Acts and 13 Regulations	(Environment Canada 2015)
Parks Canada	Serving 13 Acts and 32 Regulations	(Parks Canada 2015)
Ministry of Environment	Serving 10 Acts and 3 Regulations	(MOE 2013)
Ministry of Forests	Serving 53 Acts and 26 Regulations	(MOF 2015)
Health Canada	Serving 16 Acts and 103 Regulations	(Health Canada 2014)
First Nations	Unsettled land claims, proprietary	
Consulting scientists	Obligations to funder interests	
NGOs	Philanthropic funding obligations	
Affected public	Uncertainty interests	
Proponent	Obligations to shareholders	

6.7 Conclusions

The objective of this study was to consider a scenario of environmental impact in the Pacific waters of northern BC (where oils spills were selected as the impact of choice) and to use two integrative approaches to evaluate the social, ecological and economic impacts. Modeled dispersal of three hypothetical spill scenarios were found to overlap with areas of ocean associated with social, ecological and economic services. The effects were examined using several valuation approaches. A commercial fishing economic analysis provided a spatial perspective of the relationship of the impacts to commercial fishing harvest values. An ecological analysis of local EBSAs provided a spatial understanding of the spills with respect to important ecological spaces (species and habitats). Analysis of available social data (i.e. local ecological knowledge) helped to identify areas with significant clustering of species and ecosystem services. xGIS analysis provided a spatial perspective of a broad range of social-ecological values.

Quantitative comparisons of overlaps between the modelled spill dispersal zones and the spatial values described above provided limited opportunities for interpreting implications and managing the integrated effects of the spills. It was argued that an integrative process involving collaboration across a range of actors (i.e. government and non-government sectors and expertise from the public) could potentially bridge the gap. The analyses and resulting maps were presented as a medium through which both the spills and the ecosystem values identified could be viewed and interpreted by the actors. A staged approach of expert focus-groups engaged in analyzing increasingly complex integration questions was posed as an approach for further consideration.

Chapter Seven. Conclusions

Development opportunities on both land and marine-based ecosystems stand at the forefront of many Canadian communities, including those of the North Coast of British Columbia, currently being considered for over 20 major natural resource export and energy sector projects (for an overview of projects currently under consideration see Carleton Ray and McCormick-Ray 2013, District of Kitimat 2015, Prince Rupert and Port Edward Economic Development Corporation 2015). Despite discernible recognition among the actors involved that such projects can and do introduce significant social, health, economic and ecological effects (e.g. McCarthy et al. 2013, BG Canada 2015, Petronas 2015), common approaches and tools to assess these effects in an integrative manner are lacking (see section 3.4.3, page 75). This research was positioned to contribute to some of the important challenges encountered in the process of integrative environmental planning and management. The aim was to contribute new knowledge, insights and pragmatic solutions. In view of this overarching aim, this chapter summarizes and discusses the research questions posed and their significance, the approaches selected to address them, the strengths and limitations of the approaches, the resulting findings and the implications of the findings to future research.

7.1 Research questions and significance

Four research questions were posed in order to systematically address gaps in knowledge and application and, thus, address the aim above. In total, these questions focused on (1) the key issues preventing integration, (2) the importance of marine spaces based on integrated analysis of local spatial data, (3) the importance of marine spaces based on local expert knowledge, and (4) integrated analysis in practice. The significance of each of these questions is revisited here prior to synthesising key aspects of the research approach and findings.

The first research question sought to identify and understand the key issues believed to be responsible for the poor integration of human health and well-being considerations into environmental frameworks such as the EA process. This inquiry was further extended to the CEAA legislation in order to identify opportunities for improved integrative analyses. The imperative of this first research question is underscored by the urgency of integrative work in the environmental field. The issues identified also informed the remainder of this research as it endeavoured to put forward pragmatic (solution-oriented) responses in the form of tools and recommendations.

The second research question queried the utility of locally available social, ecological, economic and legislated spatial data for the purpose of integrated analysis, with the goal of achieving a holistic understanding of the spatial distribution of important marine spaces. The significance of this research question is particularly apparent under conditions of data, time and material resource limitations (a common condition in many coastal communities). In such instances, approaches that integrate available data to produce summary spatial results may serve as important initial inputs to environmental planning and management processes.

The third research question concentrated on the utility of a scoping tool based on local expert knowledge, to spatially detect and measure the social-ecological importance of marine spaces. The inquiry was focussed on measurements that could be fully integrated and were considerate of a broad range of attributes, ultimately producing an accurate understanding of the complete social-ecological importance of marine spaces. Given the imperative need for integrated analyses in the environmental field, this research question addresses core challenges faced by environmental managers and planners.

The final research question was concerned with the pragmatics of applying the learning and tools resulting from the inquiries above to a scenario of environmental impact (i.e. modelled oil spills at sea). This research question was posed to test the learning above - an imperative consideration when addressing the very pragmatic and practice-oriented field of environmental planning and management.

7.2 Approaches and findings

The first research question was addressed in Chapter 3. A scoping review of the literature identified the key issues preventing the integration of health concerns into environmental planning processes including the need for government intervention, gaps in methodology and tools, limitations of capacity and expertise, poor intersectoral and public collaboration / participation, challenges of data quantification, analytic complexity, and the need for process efficiency. It was found that despite decades of effort in Canada to integrate health concerns into the EA process, there has been limited progress in practice. It was argued that the status quo application of the CEAA 2012 has not been effective in overcoming the key issues in order to become an adequately integrative process, and that without a deliberate effort to reshape the process, EAs will continue to focus on biophysical impacts with some consideration of health and safety and HHRA, but will leave broader considerations of human health largely unaccounted. The ENGP EA was used as a case study to demonstrate the complexities and inefficiencies that could arise when the broader social effects of projects are not accounted for. Collaboration among the actors was presented as an important step in advancing the process of integration.

The second research question was addressed in Chapter 4 and used an approach based on locally available marine economic, ecological, social and protection legislation data (referred to as the tetrahedral of factors). Spatial statistical analyses of selected datasets pertaining to each of the four factors (i.e. commercial fishing harvest data, EBSA data, LEK data and marine protection legislation data) provided a spatial understanding of the distribution of high-value or high-incidence data clustering (hotspots); considered to be 'important' locations for the values measured. The data were also examined collectively, both for quantitative agreement (spatial overlaps) and qualitative (descriptive) relationships. This integrated analysis provided additional perspective with respect to spatial relationships showing areas of social-ecological-economic importance; areas with potentially significant contributions to certain determinants of health including income, nutrition, cultural practices and nature-based leisure.

The third research question was addressed in Chapter 5 and involved the development of a fully integrated framework for quantification and integration of spatial social-ecological expert knowledge, referred to as xGIS. The objective was to detect important social-ecological spaces in the marine ecosystem (as with the former approach) in a manner better oriented to integrated analysis. The research demonstrated that xGIS is an effective method for collecting and analyzing local knowledge expertise to identify and value a broad range of social-ecological values of marine spaces.

The final research question was addressed in Chapter 6. Several oil spills were modeled in areas with proposed tanker traffic. The modeled movement of oil was shown to interact spatially with areas associated with certain important social, ecological and economic values. It was argued that the quantitative analyses conducted could provide only limited opportunities for improved management. Instead the analyses and resulting maps were presented as a medium around which a collaborative process among the actors could occur, and the potential impacts of the spills could be interpreted and planned for.

7.3 Strengths and limitations

The scoping review conducted in Chapter 3 identified certain key issues that may be preventing the integration of health concerns into the environmental framework. This review is believed to be the first of its kind. The findings represent the learning achieved from a broad cross-section of case studies in Canada and internationally. Yet identifying the issues is only the first step. The more difficult and complex step is that of developing tools and approaches that are responsive to the issues.

Two approaches to integrated analysis were presented (see Chapters 4 and 5). Together these provided a broad and holistic perspective of the spatial distribution of important social-ecological spaces. The results of these two chapters are presented as a backdrop on which conversations about environmental planning may occur. The maps produced are a spatial representation of the social, ecological and economic values of a community. Yet, analyzing these results is associated with certain opportunities and challenges. These are elaborated below.

One, the social-economic-ecological-legislative analysis (the tetrahedral approach) is based purely on available data sources and does not require collection of additional primary data. It is, therefore, a relatively resource efficient process. The data have limitations including considerable gaps and uncertainties. Yet, these data were purposely selected for this research as they are the data that are in use by the actors –they are the data relied on by many agencies to make decisions in the study area.

Conversely, the xGIS approach does require the collection of some primary data. However, the xGIS analysis offers a significantly more detailed analysis including a broader set of values; thirteen xGIS landscape value attributes were included in this research, rather than four (the tetrahedral). Furthermore, the values included in the analysis are derived through an iterative selection process tailored to the research, rather than being defined by data availability. Conversely, the xGIS approach is largely untested. Though it is noteworthy that the final maps were presented in at least one public forum in Prince Rupert, BC and were well-received by those who were in attendance. Future research might consider a more systematic validation of the analyses and results.

Two, some of data sources employed in the tetrahedral approach offer certain advantages. First, despite gaps and uncertainties, the fish harvest data are expressed in familiar and widely accepted monetary units of measure. Second, despite certain assumptions, the Phase II EBSA analysis was partially based on peer-reviewed marine biological and ecological empirical studies (see Table 4-6).

Three, the xGIS approach was based on a relative valuation scheme, thus allowing the data produced to be seamlessly compared and the value attributes to be ranked. As a result, the total importance of a site could be described with respect to the proportional contribution of each attribute to that importance. Conversely, the tetrahedral approach could not be seamlessly integrated. That is, the integrated analysis of the four spheres considered could not be compared to determine the proportional contribution of each to the importance of a given location.

Four, the xGIS, Phase I EBSA and LEK approaches are all dependent on the reliability/accuracy of local expert knowledge related to the social-economic or biophysical environments of the study area. Clarke and Jamieson (2006b) contend that expert knowledge in the study area is accurate for ecologically and biologically significant areas. Given the data gaps, the complex and poorly understood social-ecological processes at play and the urgent need for solutions, these approaches do offer important additional insights, as has been demonstrated in this study.

A final limitation to note is that of participant selection bias – an inherent limit to the xGIS methodology, resulting in the exclusion of certain experts who may not have adequate 'spatial' knowledge of the study area. This bias was purposely accepted in order to create the spatial framework sought. However, future research may examine how non-spatial forms of knowledge may also be included in the methodology.

Chapter 6 examined the pragmatics of the approaches presented in Chapters 4 and 5. Three scenarios of environmental impact (i.e. modeled oil spills at sea based on hypothetical inputs) allowed for spatial relationships to be clearly visualized and described in relation to both the impacts (i.e. the modeled dispersal zones of the spills) and the social-ecological and economic values that had been identified in earlier analyses to be characteristic of those zones. Analyses of the relationships rapidly brought to light the inherent complexity and, thus, the necessity of collaboration among the actors to interpret the complexity from the perspective of social-ecological thresholds and tolerances.

7.4 Recommendations and implications for research and policy

Based on the findings of the scoping review of Chapter 3, it is asserted that an authentic commitment between the actors involved to the core mandate of the CEAA 2012 is required (i.e. to "exercise their powers in a manner that protects the environment and human health" [s5]). Without a genuine adherence to this common commitment, environmental planning processes may become subject to a range of complicating influences, such as the many other mandates served by the actors. To alleviate this challenge and maximize successful planning, the actors engaged in the integrated process need be permitted to set aside these 'backstage influences' and focus on the central goals established for the process as their primary directive (e.g. the core mandate of the CEAA in the case of the EA process).

The second recommendation focuses on collaboration as a means of bolstering the integrative process (see Boelen 2000, Bammer 2005, Pohl and Hadorn 2008). Significant progress is unlikely without a process that achieves appropriate collaboration among the actors; an assertion upheld by a broad range of literature including national and international case studies and theoretical analyses (Chapter 3). It is asserted that given the track record of action, the urgent need for progress and the legislation currently existing, the Canadian context provides a timely opportunity for a new era of leadership and innovation to address the challenge of integrating health considerations into environmental planning and management frameworks.

This research has maintained a pragmatic, application-oriented, view of scholarship. Boyer (1990) describes this view of scholarship as being grounded in engagement and service; seeking the application of knowledge to the problems of individuals and institutions, relying on the act of application as the medium from which new understandings will arise. Responding to calls for integration posed by Bammer (2005) and others, Chapters 4 and 5 provided new methodological insights and two pragmatic tools for valuing important social, ecological, economic and social-ecological marine spaces. Chapter 6 demonstrated that the maps produced by the tools developed could potentially serve as a useful medium of integrated analysis. Interpretation of the spatial relationships presented in the maps, in order to contribute to processes of planning and management, would still require a secondary process to determine whether the risks surpass social-ecological thresholds. It is proposed that a process that engages the actors in a staged approach to considering tiers of complexity may be an appropriate approach to interpreting and translating the results into management plans.

However, as discussed above, it is through application that the strengths and gaps of the tools of Chapters 5 and 6 will be most effectively revealed. The imperative of application arguably now exceeds the need for further theoretical methodological development. A collaborative process of engagement among the actors, as shown in Figure 6-6, to interpret the results (i.e. the maps produced in Chapters 4 and 5), ideally in the context of an environmental planning exercise, could potentially lay the groundwork for the most urgent areas of new research now required. Many coastal communities are in various stages of considering implementation of new projects or expansion of existing projects with potential risks to the services of the marine social-ecological system. It is not uncommon for these projects to become entangled in uncertainty and conflict (see BC Nature 2014, Coates 2014, Laanela 2014a, Moore 2014). Thus, there is ample opportunity and need to pilot applications in a research setting. The process of application should ideally be coordinated by the government sector in order to achieve the broadest system support (see section 3.4.2, page 73).

The MA (2005) demonstrates the extent of human exploitation of Earth's ecosystems. The warnings to follow establish the vital dependence of human well-being on better management of those ecosystems and asserted the need for integrative work in the field of environmental management as arguably the most pressing imperative. Responsive to these findings, this research has proposed approaches and tools to advance integrative work at the interface of social and ecological systems.

7.5 Conclusions and implications

Development opportunities on both land and marine-based ecosystems stand at the forefront of many Canadian communities, including those of the North Coast of British Columbia. This research was positioned to contribute new insight and solutions to some of the foremost challenges encountered in the process of integrative environmental planning and management. The key issues that may be preventing the integration of social and ecological factors were identified. Two methodologies responsive to the issues identified were developed; one based on typical datasets exiting in many coastal communities, the other on local expert knowledge. The two approaches provided several spatial representations of marine important areas based on social, ecological and economic factors.

These appear to be the first spatial representations of social-ecological values developed for the study area. The maps produced provide a medium on which questions of environmental impact (e.g. oil spills at sea) can take shape and be examined and discussed by the actors. An application-based approach to collaborative integrated analysis was proposed as a first step to better understanding the strengths and limitations of the two approaches. Despite the limitations, biases and inaccuracies of the results produced by the two approaches, this research has shown that there are ways to enable shared discussion. The approaches demonstrated the power of different spatial methods to present a diversity of information at the "same table", a table with room for everyone to sit at and develop a shared understanding of how others value space and how those values interact.

Fields related to the planning and management of social-ecological systems in Canada may have reached an important milestone: the convergence of learning from a range of past initiatives worldwide, the presence of specific legislation to guide and prod the process and a sense of urgency, in lieu of growing demands for ecosystem development, to achieve integrated management. Together these factors offer a timely opportunity for a new period of leadership in the Canadian context. This research offers a comprehensive approach for integrated analysis of social, ecological and economic factors; a potential starting point for many coastal communities. Systematic application of the approaches proposed, by the actors involved, is an important next step to help reveal new opportunities for advancing our understanding of integrated planning and management.

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Appendix A. Ocean classification systems

Ocean classification has a long history, and is rooted in the early works of both oceanographers and zoogeographers (Gregr et al. 2012). These systems of division (described in detail below) tend to create ocean units that are relatively large. Thus, though useful for large-scale oceanographic studies, they are typically too large for more localized community-based endeavors. Finer-scale divisions are needed in order to create more manageable study units for local marine research work. There are a variety of methods in use for sub-dividing large marine units into smaller and more manageable subunits (described in detail below).

A.1 Large Ocean Divisions

The earliest approaches to defining ocean units were based on the works of physical oceanographers such as Forbes (1856), Ekman (1953) and others who worked with limited ship-based observations of marine physical geography. Marine biologists had simultaneously started to recognize and describe distribution ranges among marine species (Hedgpeth 1957). The two approaches advanced largely independently (Gregr et al. 2012) until the early works of researchers such as Briggs (1974, 1995) focused on both the geography of marine coastal and shelf areas, as well as species inventory (e.g. the degree of species endemism). On these bases marine coastal and shelf areas were classified into provinces. The Briggs system eventually gained widespread adoption and is used to the present day. Other related systems have also emerged and, although the definitions and criteria used to draw the boundaries varied, they all focused on partitioning large areas into distinct (geographic) regions containing groups of organisms sufficiently distinct or unique from their surroundings (UNEP-WCMC 2007).

Today, the most significant challenge with biogeographic theory lies in our limited understanding of open-ocean and deep sea ecosystems, particularly with respect to the vulnerability, resilience and functioning of marine biodiversity in these areas (UNESCO 2009). Thus, one of the more commonly used biogeographic classification systems in these environments (the Longhurst Biogeochemical Provinces or BGCP ocean classification system) (Longhurst 1998, 2007) is based solely on oceanographic (Sverdrup) processes rather than species data.

Certain other classification systems attempt to combine biogeographic data with geo-political or socio-economic considerations. The Large Marine Ecosystem (LME) classification (Sherman and Alexander 1986) is an example of the former and perhaps the most widely used system for management purposes. The Large Ocean Management Area (LOMA) classification is an example of the latter (DFO 2011). Both share many of the same principles and criteria (Siron et al. 2008). These works have given way to other modern classification systems including the Global Open Oceans and Deep Seabeds (GOODS) biogeographic classification (UNESCO 2009) and the Marine Ecosystems of the World (MEOW) classification (Spalding et al. 2007).

The **Biogeochemical Provinces of the Ocean (BGCP)** classification system is essentially based on Sverdrup processes (the unit measure for volume transport in ocean currents) (DFO 2009a). These are used to determine marine biological processes and their influences on the rest of the food chain. The classification uses two spatial scales: biomes (based on the influence of winds and sunlight on Sverdrup mixing) and provinces (based on detailed Sverdrup processes within each biome). Biome boundaries follow latitudinal trends and seasonal changes in plankton composition, while provincial boundaries within biomes use a wider set of factors (e.g. regional circulation and stratification, bathymetry, river discharges, coastal wind systems, islands, and land mass distribution). BGCP establishes 4 biomes and 51 provinces (Figure A1) of which 6 are relevant to Canada (Longhurst 1998, 2007).

In 2008, at the 9th meeting of the Conference of the Parties (COP9) to the Convention on Biological Diversity (CBD), commitments were made by nations relating to the conservation and sustainable use of biodiversity in marine areas beyond national jurisdictions. The outcome of the meeting of expert panels was the adoption of a biographic classification system based primarily on oceanographic and bathymetric similarities. It was applied to each of Canada's three oceans: the Atlantic Ocean (the Scotian Shelf, the Newfoundland-Labrador Shelves, and the Gulf of St. Lawrence), Arctic Ocean (Hudson Bay Complex, the Arctic Archipelago, the Arctic Basin, the Eastern Arctic, and the Western Arctic), and Pacific Ocean (the Northern Shelf, the Strait of Georgia, the Southern Shelf, and the Offshore Pacific Zone) (Figure A2) (DFO 2009a).

Large Marine Ecosystems (LME) are relatively large regions of ocean, on the order of 200,000 km^2 or greater. The LME system is rooted in the 1982 United Nations Law of the Sea Convention which granted coastal states sovereign rights to explore, manage and conserve the natural resources of their Exclusive Economic Zone (EEZ). As a result, LMEs tend to focus on coastal and shelf areas (open ocean and deep sea areas beyond national jurisdiction are not covered). The system establishes 64 LMEs worldwide (see Figure A3) distinguished on the basis of bathymetry (bottom depth), hydrography (temperature, salinity, Sigma T, tides and currents), productivity (chlorophyll, dissolved oxygen, total zooplankton), and trophic linkages (informed using plankton, demersal and pelagic surveys) (Sherman and Alexander 1986). Today, the LME classification system is one of the most widely used (UNESCO 2009).

Large Ocean Management Areas (LOMAs) are used in Canada for the implementation of ecosystem-base integrated-management plans (Siron et al. 2008, DFO 2011). Canada has 5 Large Ocean Management Areas (LOMAs) off its west, north and east coasts. These include one LOMA on the Pacific North Coast (also known as the Pacific North Coast Integrated Management Area or PNCIMA), one LOMA for the Central and Arctic (Beaufort Sea) and three LOMAs on the Atlantic (Gulf of St. Lawrence, Eastern Scotian Shelf and Placentia Bay/Grand Banks) (see Figure A4).

Global Open Oceans and Deep Seabed Biogeographic Classification (GOODS) is rooted in the vision of establishing an international network of marine protected areas; a vision that emerged from the Johannesburg Plan, was furthered at the 2002 World Summit on Sustainable Development, and again following adoption of targets set at the 7th meeting of the Conference of the Parties (COP7) to the Convention on Biological Diversity (CBD) in 2004. In response to this vision, a series of three multidisciplinary expert workshops resulted in the creation of GOODS. GOODS is a hypothesis-driven system that uses both the geographic and physical characteristics of the benthic and pelagic environments to identify regions of homogeneous habitat and associated biological characteristics (i.e. similar temperature, salinity, depth profiles and species complements) (DFO 2009a). GOODS establishes pelagic bioregions (29 provinces of which 5 are relevant to Canada) and a deep-sea benthic classification with 3 depth zones and 29 biogeographic provinces (6 of which are relevant to Canada) –see Figure A5. UNESCO (2009) recommends GOODS be considered in conjunction with finer scale biogeographic classifications adopted or developed for the region of use.

Marine Ecosystems of the World (MEOW) is among the newest classification systems, covering coastal areas and continental shelves to the 200 m depth contour (i.e. does not extend to the open ocean, deep sea or beyond national jurisdictions) (UNESCO 2009). It distinguishes ecosystems based on both biogeographic and practical utility considerations. The MEOW system consists of 12 realms, nesting 62 provinces and a further

232 ecoregions; 15 of which are relevant to Canada (Figure A6). This system provides considerably better spatial resolution than earlier global systems (Spalding et al. 2007). The GOODS and MEOW systems are compatible in terms of approaches and definitions and can therefore be nested relatively well; though 'fuzzy boundaries' do invariably lead to some mismatching (UNESCO 2009).

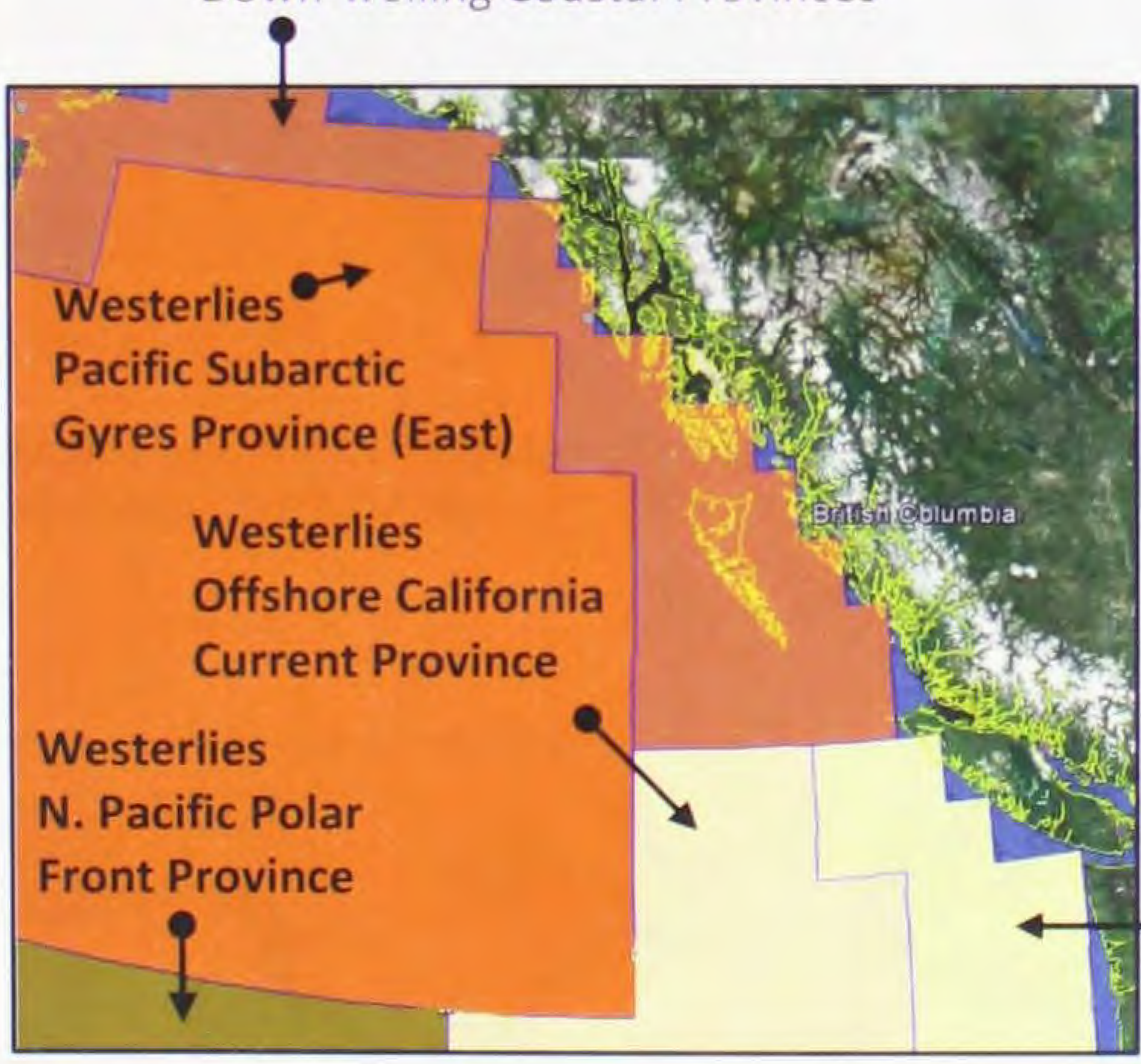

Figure A-1. The biogeochemical provinces (BGCP) of the Pacific North Coast of BC. Map adapted from Watson (2008).

Figure A-2. Major biogeographic units for the Canadian Pacific Ocean established at COP9. Map from DFO (2009a).

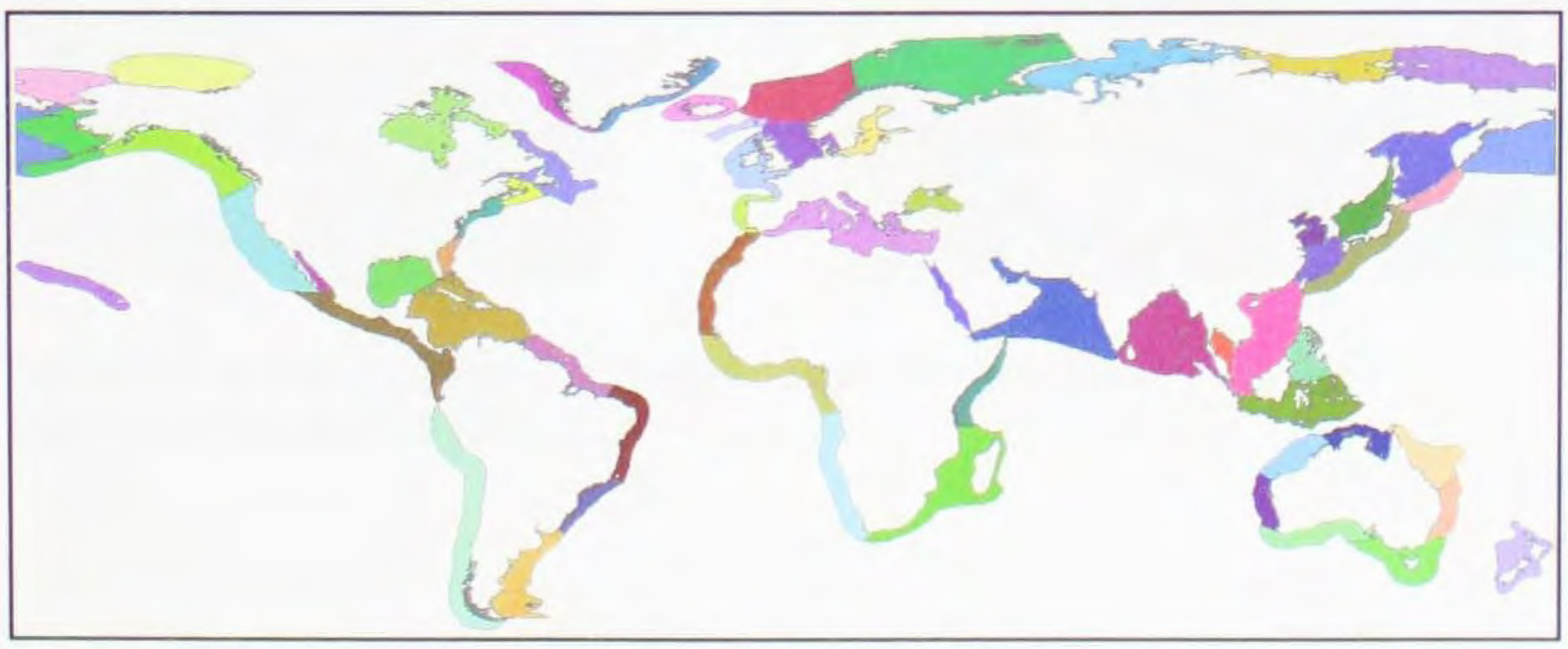

Figure A-3. A depiction of the global distribution of the Large Marine Ecosystems (LME). Source data from U.S. LME Program (2013).

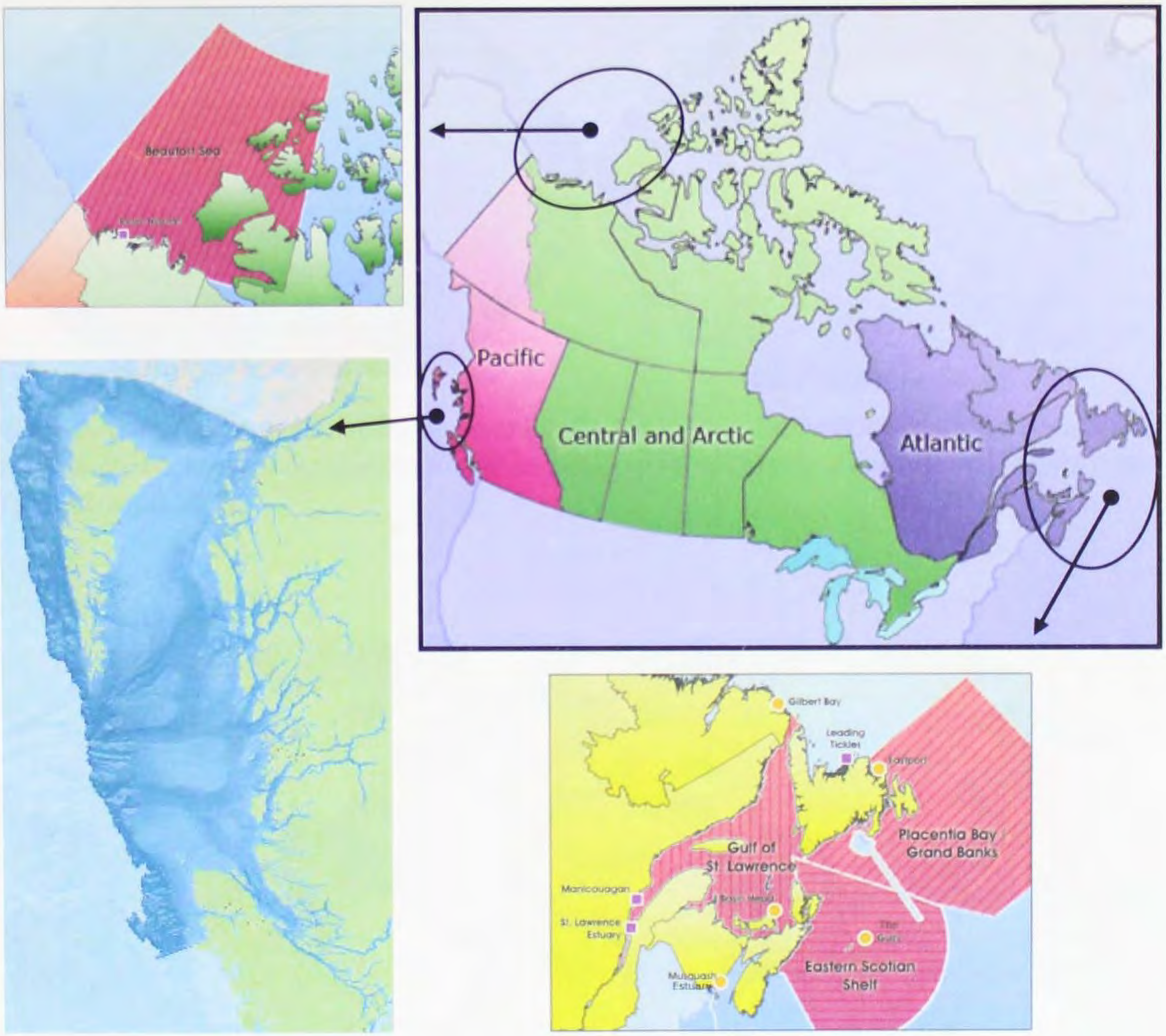

Figure A-4. A map of Canada's five large ocean management areas (LOMA). Map adapted from DFO (2011).

Figure A-5. The abyssal (3500-6000m depth) (left) and bathyal (800-3000m depth) (right) provinces of the Global Open Oceans and Deep Seabed Biogeographic Classification System (GOODS). Map from (UNESCO 2009).

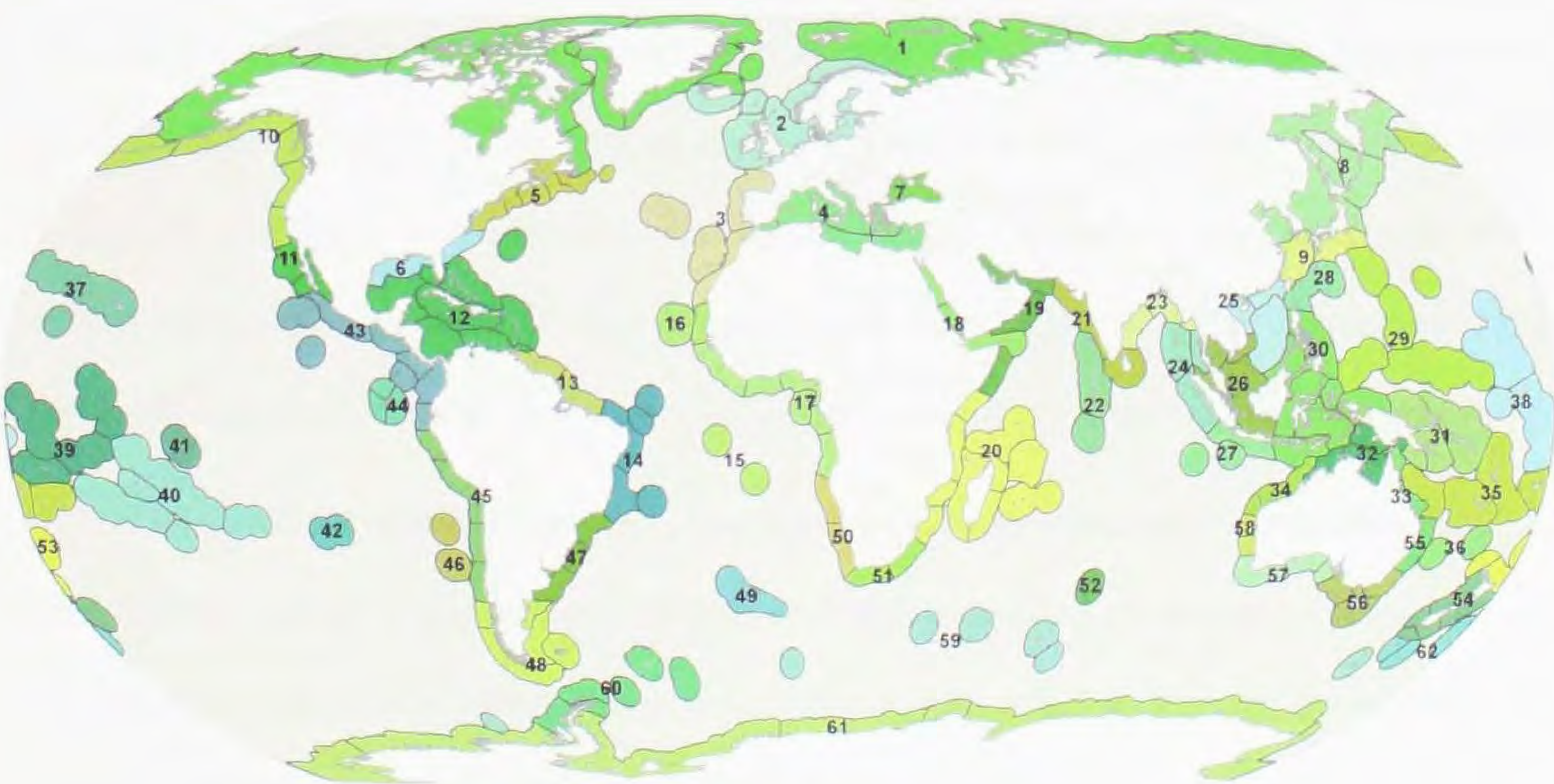

Figure A-6. Provinces and ecoregions of the Marine Ecosystems of the World (MEOW). Map from Spalding et al. (2007).

A.2 Smaller ocean units

Sub-dividing large marine units into smaller and more manageable subunits is achieved using a number of methods. For example, a simple grid system can be applied to the waterscape, where each cell is treated as a spatial unit (Figure A7). The grid system has been used by Fisheries and Oceans Canada to record catch data for various commercial fisheries.

Other methods can be much more complicated, and equally demanding of data that may not be universally available for all areas of the ocean. One of a number of examples that might serve to illustrate this include models developed by Kostylev and others (Kostylev 2004, Kostylev et al. 2005, Kostylev and Hannah 2007) which incorporate a number of physical oceanic attributes to describe ocean units by two characteristics: Disturbance and Scope for Growth. Gregr (2007) applied these theoretical models to Pacific marine waters (Figure A8) and found that marine species groups were preferentially associated with the different regions of the disturbance–scope for growth grid. Gregr (2007 p.15) contends that this work represents "the limit of what can be achieved with the data currently available".

The **British Columbia Marine Ecosystem Classification (BCMEC)** is a hierarchical classification system first developed in 1995 that delineates Provincial marine areas into four nested divisions (Ecozones, Ecoprovinces, Ecoregions and Ecosections) based on 1:2,000,000 scale physiographic and oceanographic properties and a fifth nested division (Ecounits) based on more detailed (1:250,000 scale) ocean current, depth, substrate class, bottom relief, salinity, temperature, stratification and wave exposure data (see Figure A9). The BCMEC is used for marine and coastal planning, resource management and a Provincial marine protected areas strategy (GeoBC 2012).

Ecologically and Biologically Significant Areas (EBSAs) are identified through a process developed by Fisheries and Oceans Canada (DFO 2014c). EBSAs are marine areas considered worthy of enhanced management or risk aversion because they rank highly on one or more of three dimensions, *uniqueness, aggregation* and *fitness* consequences, and can be weighted by two other factors, *naturalness* and *resilience*. On the Pacific Coast of BC, a two-phase process was used to identify EBSAs. In Phase I, regional scientific experts were surveyed using a modified Delphic approach to identify areas of PNCIMA that met the five criteria. The areas identified were called Important Areas (IAs). Experts were also asked to rank each IA they identified according to each of the five EBSA criteria. This produced 132 species-related biological IAs (41,838 km^2 or 41% of PNCIMA); simplifying to 10 IA polygons once overlaps were accounted for (Clarke and Jamieson 2006b).

In Phase II, EBSAs were delineated based on the findings of the Gulf of St. Lawrence Integrated Management Area review which concluded that major oceanographic, bathymetric and topographic constraining of species distributions to specific areas (i.e. bottleneck areas) could also provide a basis for the identification of EBSAs (DFO 2006). In other words, just as areas of high biological productivity (typically corresponding to aggregations of many species) are considered to have ecological significance, oceanographic, bathymetric and bottleneck areas are also areas of high ecological significance (Clarke and Jamieson 2006b). The IA's from Phase I and II were collectively analyzed for overlaps and congruency. In total, 95 of the 132 biological IAs were captured by the Phase II approach. Based on this analysis Clarke and Jamieson (2006b) proposed a final list of 15 EBSAs in the PNCIMA (i.e. 42.7% of the LOMA) (Figure A10).

These methods of habitat typing, along with a suite of others that are currently in use (e.g. Howes 2001, Pickrill and Kostylev 2007, Ban et al. 2010) do provide certain insights into the distribution of smaller-scale ecological units. However, they also typically suffer from various critical data gaps. For example, the disturbance map (Figure A8) lacks coverage of near shore waters due to insufficient ocean substrate data and good resolution wave data which Gregr (2007) suggests unlikely to bec[redacted] Furthermore, these methods are essentially biophysical in nature and, thus, largely ignore the powerful forces of social phenomena that can heavily contribute and shape the overall designati[redacted]

Another approach to marine sub-division is the **Pacific Fisheries Management Areas (PFMA)**, also called DFO Statistical Areas (Figure A11). This method was applied by the Canadian government to the 27,000 km of coastline along the west coast of Canada, extending out 200 nautical miles (about 370 km) into the Pacific Ocean. The method was originally designed for the management of the salmon fishery (Marine Matters 2006) and was based on geographical and management-based criteria, rather than ecological criteria (PNCIMA 2011). PFMAs are then further sub divided by sight lines (rhumb lines) connecting known geographic reference points[1] resulting in a variable number of varying sized Subareas. The boundaries of PFMAs and their Subareas are very exactly delineated in the Pacific Fishery Management Area Regulations (Government of Canada 2007).

[1] Reference points used to date include physical features, navigation aids, co-ordinates and boundary signs.

Today this system is used by Fisheries and Oceans Canada for the purpose of managing most of the commercial, recreational and Aboriginal communal fishing activities on the Pacific Coast. Specifically, it manages openings, closures and fishing quotas (including size and weight limits), and monitors catch and catch effort for various stocks or species by Subarea. Note: some fisheries are managed using other management areas, some of which do not align with the PFMAs (Government of Canada 2007, PNCIMA 2011).

Another approach applied in the BC context to delineating ocean spaces, is referred to as the **Marine Planning Partnership for the North Pacific Coast (MaPP).** MaPP boundary delineation is achieved based on a range of social and ecological parameters including: ecological values, cultural and Aboriginal use values, current uses and activities, future economic opportunities, adjacent land uses, Marxan analyses, buffer zones, and ease of identification, navigation and management (Marine Planning Partnership Initiative 2015) (See Figure A12).

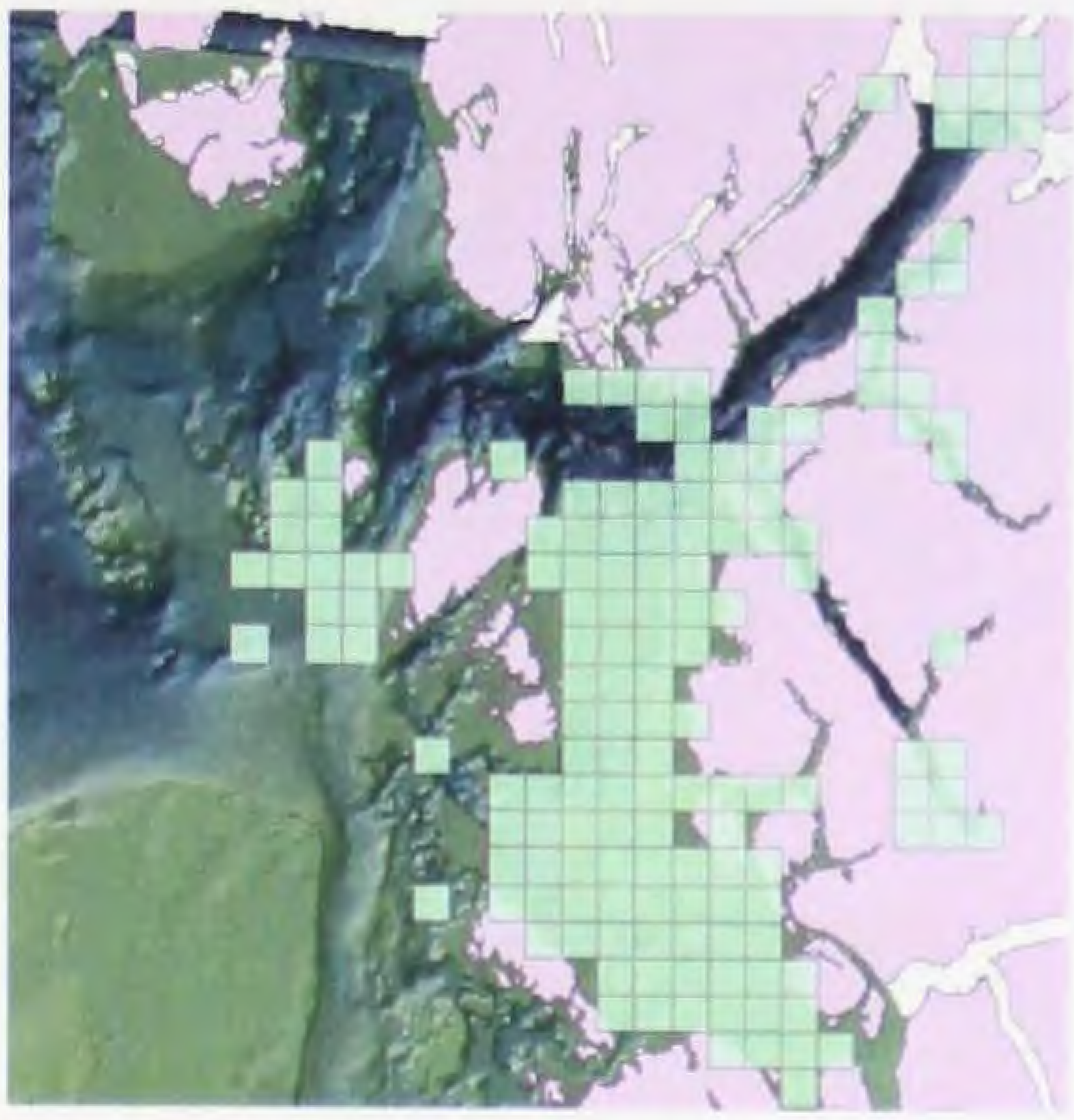

Figure A-7. A 4x4 km gridded map of a portion of the Pacific North Coast of BC; used by Fisheries and Oceans Canada as a spatial unit for summarizing fish-catch data.

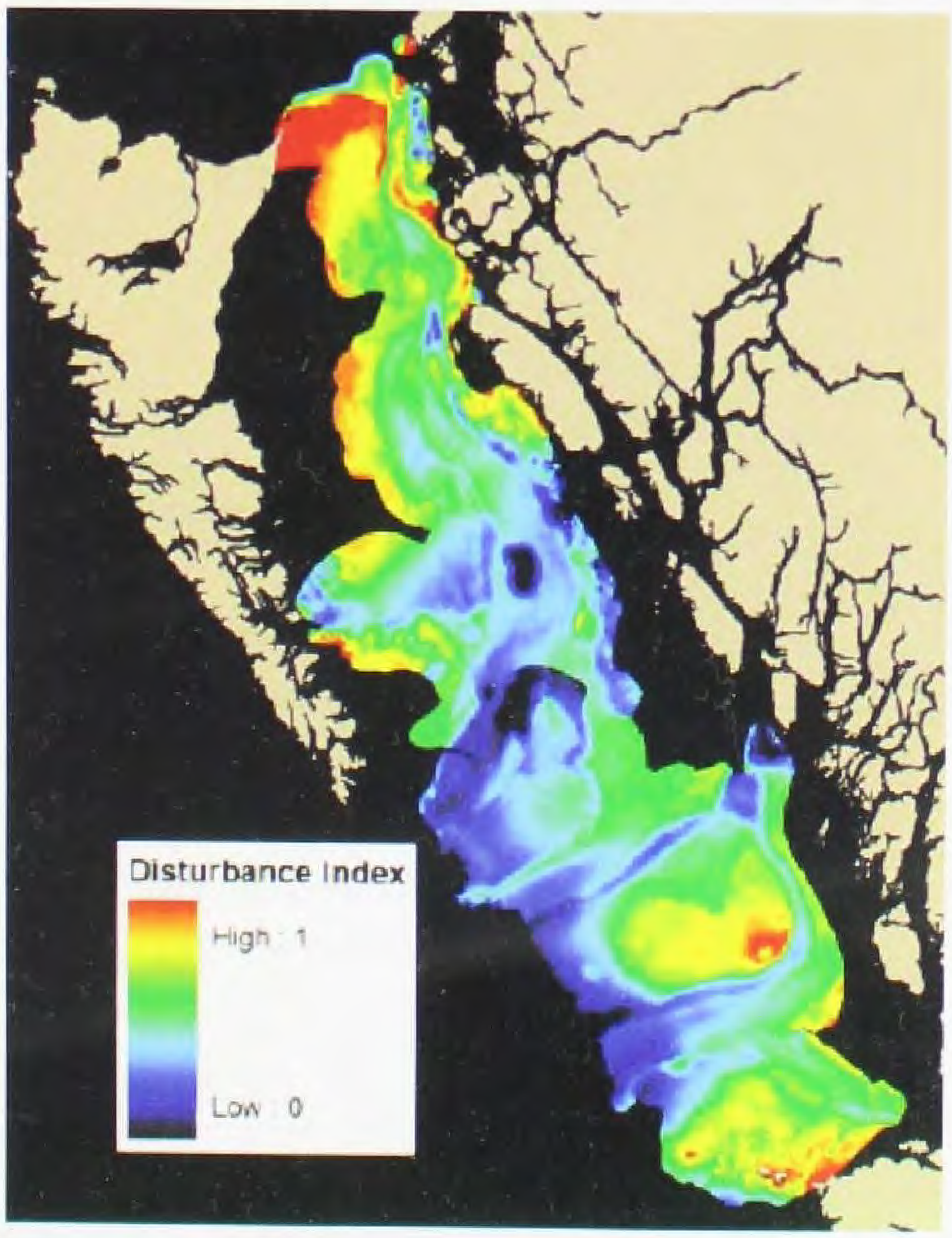

Figure A-8. Disturbance map based on a model developed at NRES Canada demonstrating large data gaps in near-shore waters. Map from Gregr (2007).

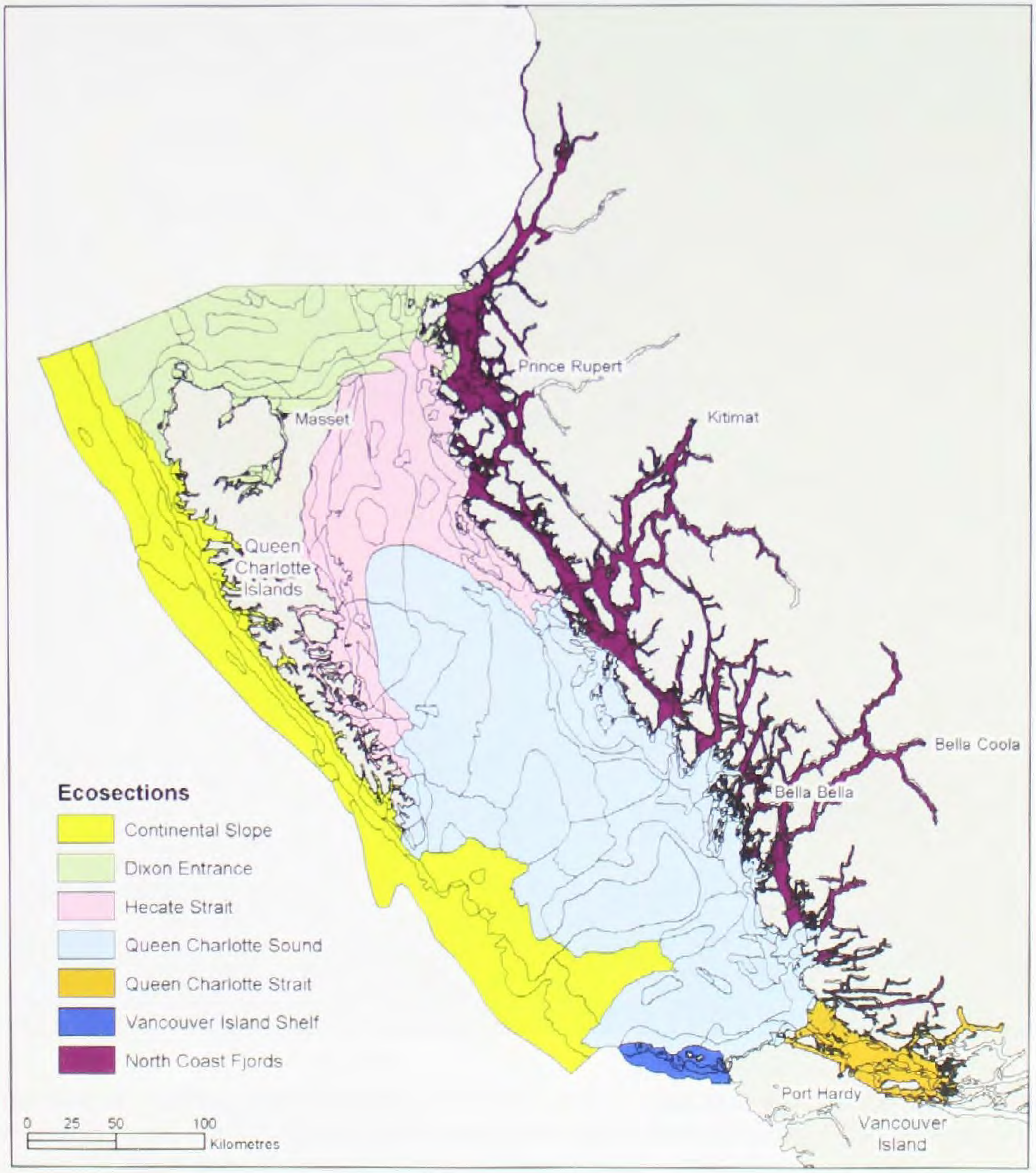

Figure A-9. Map of the BC Marine Ecosystem Classification (BCMEC) system displaying the deliniation of Ecosections (colors) and Ecounits (lines). Map from LGL Limited Environmental Research Associates (2004).

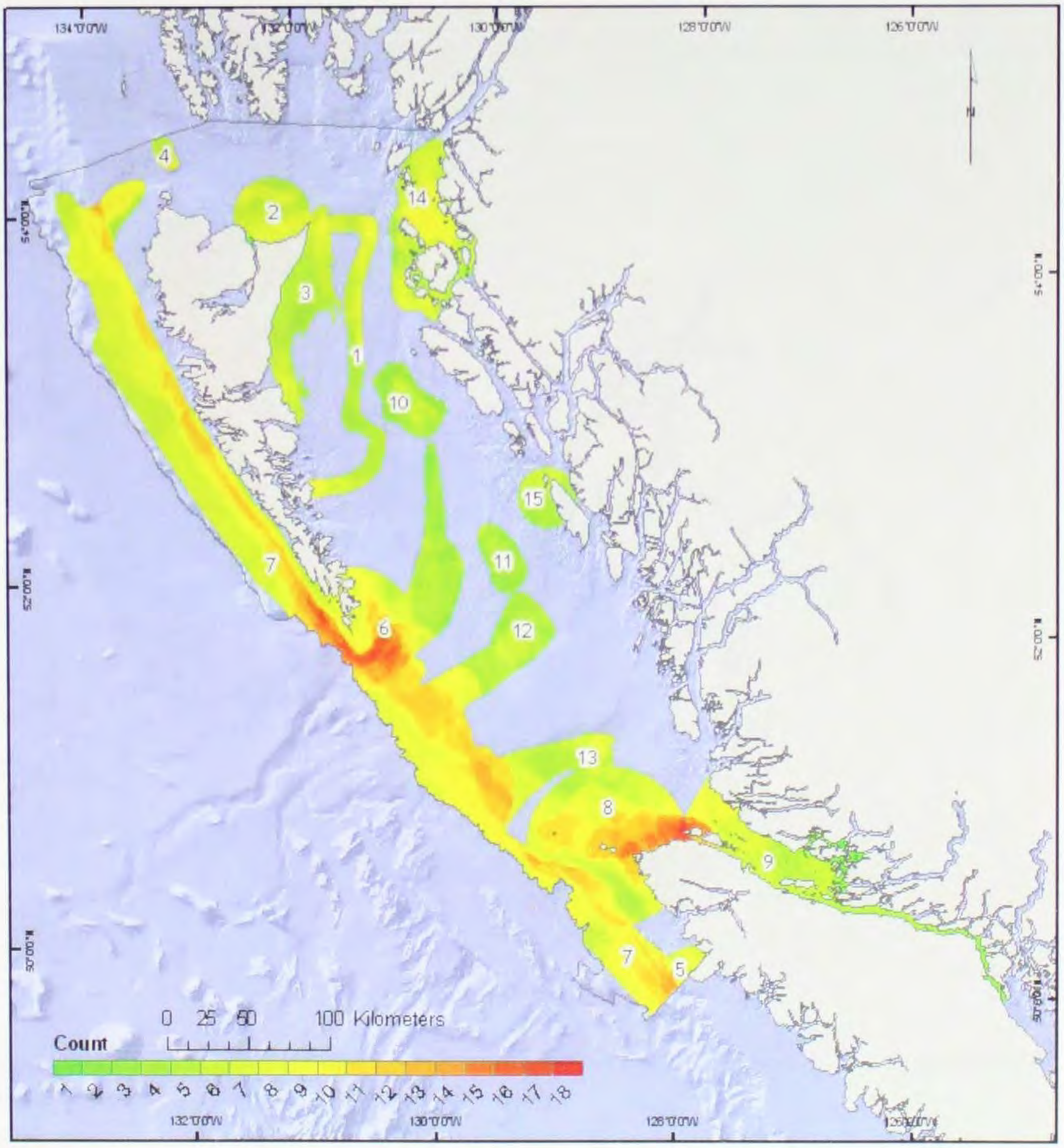

Figure A-10. The final EBSAs identified for the PNCIMA. (1) Hecate Strait Front, (2) McIntyre Bay, (3) Dogfish Bank, (4) Learmouth Bank, (5) Brooks Peninsula, (6) Cape St James, (7) Shelf Break, (8) Scott Islands, (9) North Island Straits, (10-13) Sponge reef bioherms, (14) Chatham Sound, and (15) Caamano Sound. Counts represent the number of overlaid Important Areas in each EBSA. Map from Clarke and Jamieson (2006b).

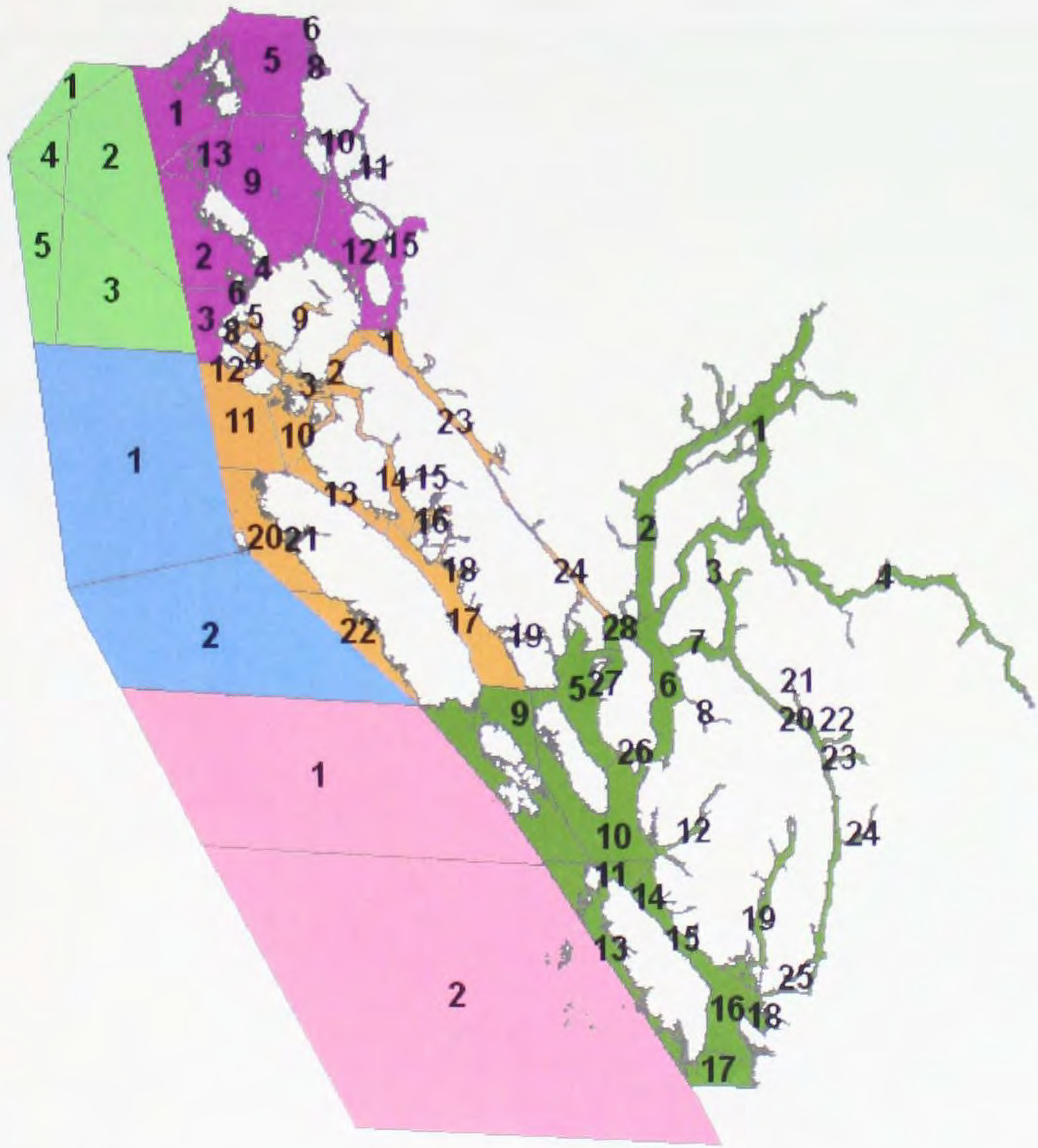

Figure A-11. Pacific Fisheries Management Areas (differentiated by colors) and Subareas (differentiated by numbers).

Figure A-12. A boundary map of the North Coast study area of the Marine Planning Partnership (MaPP) and the Pacific North Coast Management Area (PNCIMA) LOMA. Selected communities of the Regional Districts of Skeena-Queen Charlottes and Kitimat-Stikine are shown, including the municipalities of Prince Rupert, Kitimat and Port Edward, the settlements of Oona River and Dodge Cove, and the First Nations communities of Kitimat Village (Haisla), Hartley Bay (Gitga'ata), Kitkatla (Gitxaala), Lax Kw'alaams and Metatakla.

Appendix B. The governance and protection of Canada's oceans: Acts and agencies

The Canadian federal government has specific authority as legislated in the Canadian Constitution Act of 1867 to govern and protect the nation's oceans [s. 91.10], sea coasts and inland fisheries [s. 91.12] (Government of Canada 1867). To implement the Constitution, a number of Acts were brought into force including, the Oceans Act, Fisheries Act, Species at Risk Act, Canada Wildlife Act, Migratory Birds Convention Act, Canada National Marine Conservation Areas Act, and the Canada National Parks Act (see Table B-1). The Canadian federal government has established three key agencies to carry out the work of implementing the legislation, as well as, number of other agencies with related responsibilities. Table B-2 outlines the ocean-related responsibilities of each of the government agencies of Canada.

Table B-1. Canadian Acts related to the management of Canada's oceans and their natural resources.

The ACT	Authority and Protection measure
Oceans Act	Authorizes the Minister of Fisheries and Oceans Canada to develop and implement a national oceans management strategy and to recommend the establishment of Oceans Act Marine Protected Areas in order to conserve and protect commercial and non-commercial fishery resources and their habitats, endangered or threatened marine species and their habitats, unique habitats and areas of high biodiversity or biological productivity. The Act is based on the principles of sustainable development, integrated management and the precautionary approach.
Fisheries Act	Authorizes the Minister of Fisheries and Oceans Canada to manage freshwater and marine fisheries throughout Canada, including licensing and enforcement, provisions for closing areas to fishing, prohibiting the harmful alteration, disruption or destruction of fish habitat, and to regulate the use of substances deleterious to fish.
Species at Risk Act	Authorizes three ministers (the Minister of Fisheries and Oceans Canada, Minister of Environment, and Minister of Parks Canada) to protect wildlife (flora/fauna) that are nationally listed as 'at risk' from becoming extinct or lost from the wild. The Act also provides for the recovery of endangered and threatened species and encourages the management of other species to prevent them from becoming at risk. Lastly, the Act allows for the creation of prohibitions for the purpose of protecting listed threatened and endangered species, their residences and their critical habitat. The Act is based on achieving conservation through stewardship and cooperation.
Canada Wildlife Act	Authorizes the Minister of the Environment to acquire nationally significant habitats for the purposes of wildlife research, conservation and interpretation. The Act provides for the establishment and management of National Marine Wildlife Areas based on defensible biological values, for the purpose of ensuring the conservation and protection of key breeding, feeding, migration, and over wintering sites for birds, species-at-risk and other wildlife of national importance.
Migratory Birds Convention Act	Contains a Sanctuary Regulations subsection which authorizes the Minister of Environment to establish and manage Migratory Bird Sanctuaries (MBS) on federal lands. These are areas that are important for major migratory bird populations, such as seabird breeding colonies or the critical habitat of migratory birds at risk. The purpose of these sanctuaries is to protect birds and their nests from harm or harassment.
Canada National Marine Conservation Areas Act	Authorizes the Minister of the Environment to establish National Marine Conservation Areas; the purpose being to protect and conserve marine areas that are representative of the country's ocean environments and Great Lakes, and to encourage public understanding, appreciation and enjoyment of this marine heritage.
Canada National Parks Act	Authorizes the Minister of Environment to establish and manage National Parks (which may include a marine component) in a manner that leaves them unimpaired for the benefit, education and enjoyment of future generations. (DFO 2005)

Table B-2. The ocean-related responsibilities of the various agencies of the federal and provincial governments of Canada.

Government Agency	Responsibility
Fisheries and Oceans Canada (DFO)	Oceans Act Fisheries Act Species at Risk Act
Environment Canada	Canada Wildlife Act Migratory Birds Convention Act Species at Risk Act
Parks Canada	Canada National Marine Conservation Areas Act Canada National Parks Act
Transport Canada	Regulate navigation and shipping including prevention of pollution from ships via ballast water
Natural Resource Canada	Management of non-renewable resources
Indian & Northern Affairs Canada	Manage non-renewable resources of the Arctic Ocean
Foreign Affairs Canada	International issues in Canada-U.S. boundary waters
Foreign Affairs + Justice Canada	Maritime boundary disputes
Foreign Affairs + DFO	Review foreign requests to conduct marine scientific research in offshore waters under Canada's jurisdiction
Provinces of Canada	Business licenses [s. 92.9]. Incorporate companies with provincial objects [s. 92.11] Property and civil rights within the province [s. 92.13]

Appendix C. The ArcGIS hotspot analysis tool and the Getis-Ord Gi* statistic

The ArcGIS 10.1 Hot Spot analysis tool calculates the Getis-Ord Gi* statistic for each feature in a dataset (see algorithm Figure C1). The tool analyzes features in a dataset to detect the presence of clustering of both high and low values. It reports the statistical significance of the clustering. Statistical significance is determined by proportionally comparing the local sum of a feature and its neighbors to the sum of all features. When the local sum is very different from the expected local sum, and that difference is too large to be the result of random chance, a statistically significant z-score and p-value results (ESRI 2012). The null hypothesis is complete spatial randomness, either of the features themselves or of the values associated with those features. The p-value is the probability that the observed spatial pattern was created by random process while the z-score is the standard deviation. Both z-scores and p-values are associated with the standard normal distribution as shown in Figure C2. The relationship between z-values, p-values and confidence is shown in Table C1.

The Getis-Ord local statistic is given as:

$$G_i^* = \frac{\sum_{j=1}^{n} w_{i,j} x_j - \bar{X} \sum_{j=1}^{n} w_{i,j}}{S \sqrt{\frac{\left[n \sum_{j=1}^{n} w_{i,j}^2 - \left(\sum_{j=1}^{n} w_{i,j} \right)^2 \right]}{n-1}}} \quad (1)$$

where x_j is the attribute value for feature j, $w_{i,j}$ is the spatial weight between feature i and j, n is equal to the total number of features and:

$$\bar{X} = \frac{\sum_{j=1}^{n} x_j}{n} \quad (2)$$

$$S = \sqrt{\frac{\sum_{j=1}^{n} x_j^2}{n} - (\bar{X})^2} \quad (3)$$

The G_i^* statistic is a z-score so no further calculations are required.

Figure C-1. The Getis-Ord statistic used in hotspot analysis. From ESRI (2012).

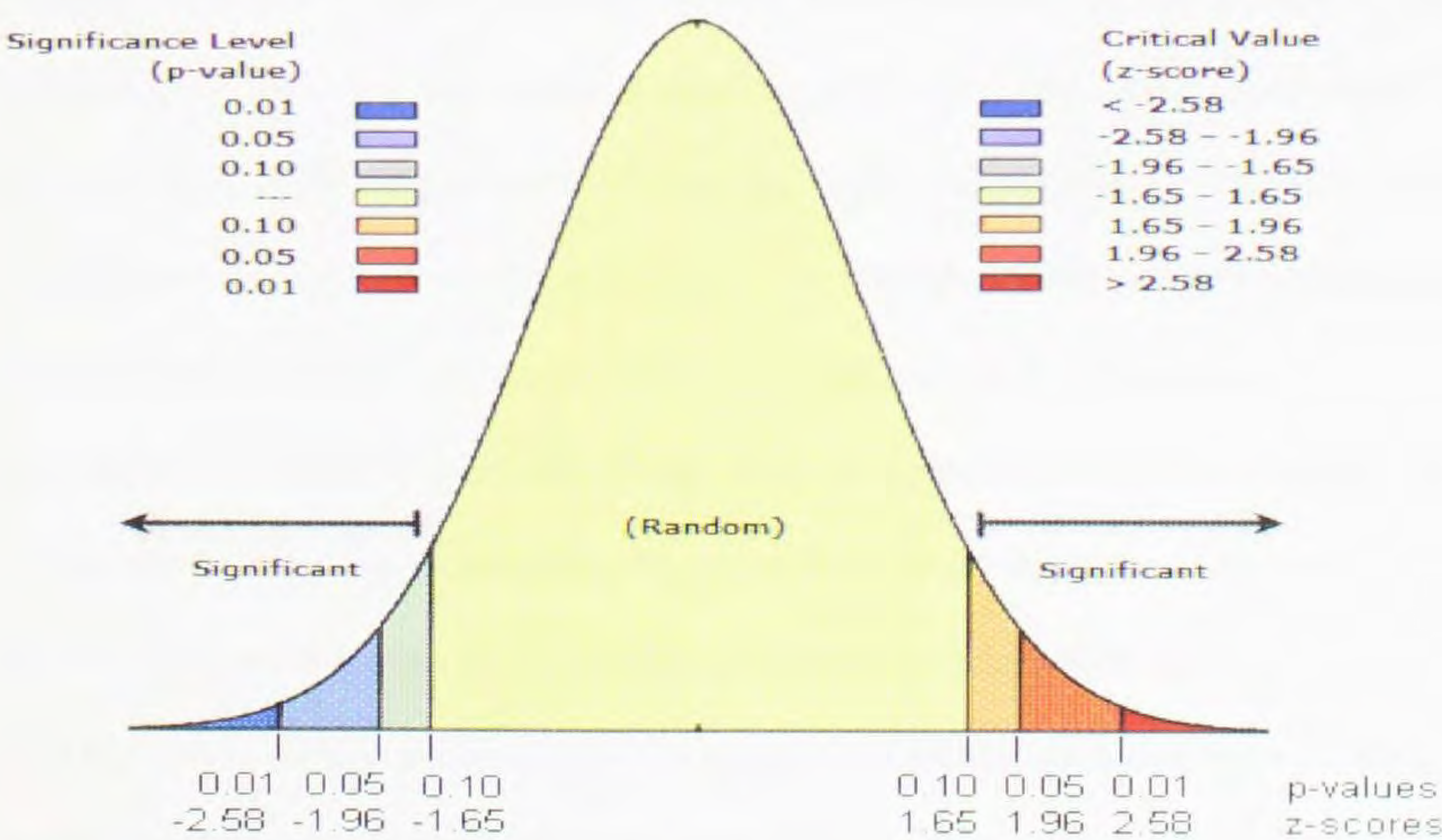

Figure C-2. Z-scores and p-values shown in association with the standard normal distribution. From ESRI (2012).

Table C-1. The relationship between z-scores, p-values and confidence

z-score (Standard Deviations)	p-value (Probability)	Confidence level
< -1.65 or > +1.65	< 0.10	90%
< -1.96 or > +1.96	< 0.05	95%
< -2.58 or > +2.58	< 0.01	99%

Another consideration in the application of the hotspot analysis tool is that of selecting the **conceptualization of spatial relationships**. The objective is to inform the model with respect to how features interact with each other in order to maximize the accuracy of results. According to ESRI (2012) **inverse distance methods** are most appropriate with continuous data or to model processes where the closer two features are in space, the more likely they are to interact/influence each other. The **fixed distance band method** works well for point data and is the default option used by the Hot Spot Analysis (Getis-Ord Gi*) tool. It is also a good option for polygon data with large variation (e.g. very large polygons at the edge of the study area and very small polygons at the center) and the need to ensure a consistent scale of analysis. The **zone of indifference conceptualization** works well when a fixed distance method is appropriate but the imposing of sharp boundaries on neighborhood relationships is not an accurate representation of the data. The **polygon contiguity conceptualizations** are effective when polygons are similar in size and distribution, and when spatial relationships are a function of polygon proximity (the idea that if two polygons share a boundary, spatial interaction between them increases). The **K nearest neighbors** option is effective in analyses requiring a minimum number of neighbors and when the values associated with features are not normally distributed.

The models above also require selection of a 'distance band' or 'threshold distance' value. The default distance threshold will be the minimum distance that ensures every feature has at least one neighbor (ideally 8). There is no absolute method for selecting distance bands. One approach to selecting an appropriate distance band is to apply the ArcGIS 10.1 Spatial Autocorrelation (Morans I) tool in distance increments to determine peak z-values preceding z-value dips, see ESRI (2012) for details.

Appendix D. Determining expert richness and the completeness of sampling

A useful estimate in studies involving sampling is that of determining the completeness of sampling (i.e. the number of species detected at current sampling effort as a proportion of the total species richness). The challenge of measuring sampling completeness has been well studied in ecology and biodiversity research. Generally, as sampling effort increases, the rate at which new species are added to the inventory declines asymptotically (Jiménez-Valverde et al. 2006). The asymptotic value of the species accumulation curve theoretically represents the total species richness that would be achieved with infinite surveys. In some cases, the asymptotic condition is not evident in the species accumulation curve. This may be resolved with additional sampling or, alternatively, the use of estimators. Longino et al. (2002) summarized three categories of estimators:

(1) estimating the hidden (unsampled) portion of the curve by fitting the data to a lognormal distribution and then estimating species richness (the area under the fitted curve including the portion hidden behind the 'veil line');

(2) Fitting asymptotic equations to the species accumulation curve;

(3) Using non-parametric estimators.

In instances when the distribution of the data deviates markedly from normal (i.e. skew and kurtosis measurements >1), the asymptote of the curve may not become apparent. In such instances, non-parametric estimators show promise for species richness estimation (Colwell and Coddington 1994, Longino et al. 2002). Non-parametric estimators are based on rarefaction -the number of rare or infrequent species appearing in the collection during species sampling (i.e. the number of singletons and doubletons). They extrapolate from the rarefaction data to estimate the likelihood that there are still more undiscovered species (Chao et al. 2009, Vavrek 2011). In other words, they attempt to estimate the true number of species in the wild (Colwell and Coddington 1994).

A number of non-parametric estimators are available for measuring true species richness. These include the Chao 1, Chao 2, ACE, ICE, first-order jackknife, second-order jackknife and bootstrap. The Chao estimator has several variations. Chao 1 was proposed by Chao (1984) and relies on abundance data. When only occurrence or incidence data are available, the Chao 2 estimator is appropriate (Chao 1987, Colwell and Coddington 1994). Zhang (2010) argues that the Chao 2 estimator is most stable after the cumulative sample size reaches 10 samples, and prior to that, considers it 'second-best' to the Bootstrap estimator. The second order Jackknife has been shown to be one of the most effective estimators for highly sparse collections as it is less susceptible to sampling bias (Chazdon et al. 1998, Vavrek 2011). However, it tends to underestimate if sample size is larger than 40-50 samples (Zhang 2010). The Incidence-based Coverage Estimator (ICE) shows considerable promise (Longino et al. 2002). It calculates both rare and common species, considering as 'common' those that occur in more than 10 samples (Basualdo 2011). Zhang (2010) assessed seven non-parametric estimators in a study aimed at estimating plant species richness and found the Bootstrap estimator developed by Smith and van Belle (1984) to yield the least absolute and relative bias and to be insensitive to cumulative sample size. Conversely, Chazdon et al. (1998) argue the Bootstrap estimator to be one of the poorest species estimators.

Given the conflicting conclusions as to the strengths of the various estimators, Vavrek (2011) recommends using a number of them in concert "as concurrence between their individual values can lend support to their results." Software packages, such as EstimateS™ (Colwell 1994–2011), report multiple species estimators, including data variance and confidence intervals. Table D1 demonstrates the general variability observed among the various estimators when applied to data from a study by Zhang (2010) to estimate the total richness of plant species in the study region.

Table D-1. Estimates of plant species richness using seven non-parametric models. From Zhang (2010).

Estimator	Species Richness	95% lower limit	95% upper limit
Chao 1	49.00	47.42	50.58
Chao 2	48.35	47.77	48.92
Jackknife 1	50.94	29.94	71.94
Jackknife 2	42.59	-	-
Bootstrap	51.18	48.20	54.16
Chao 3	51.76	49.80	53.72
Chao 4	52.19	50.12	54.26
Average	49.28	45.54	55.59

Appendix E. Ethics Approval

UNIVERSITY OF NORTHERN BRITISH COLUMBIA

RESEARCH ETHICS BOARD

MEMORANDUM

To:	Pouyan Mahboubi
CC:	Margot Parkes
From:	Michael Murphy, Chair Research Ethics Board
Date:	August 7, 2015
Re:	**E2012.0314.032.03** **Development and critical assessment of a scoping tool to characterize the social–ecological importance of coastal marine locations on the North Coast of BC**

Thank you for submitting a request for renewal to the Research Ethics Board (REB) regarding the above-noted proposal. Your request has been approved.

We are pleased to issue renewal approval for the above named study for a period of 12 months from the date of this letter. Continuation beyond that date will require further review and renewal of REB approval. Any changes or amendments to the protocol or consent form must be approved by the REB.

Good luck with continuation of your research.

Sincerely,

Dr. Michael Murphy
Chair, Research Ethics Board