MULTIPLE PERSPECTIVES FOR ENVISIONING MARINE PROTECTED AREAS

by

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Abstract

This thesis provides the first direct comparison between – and integration of – community-based and science-based approaches to the establishment of marine protected areas (MPAs). MPAs are one potentially effective conservation tool, but are being established very slowly. My research shows that community involvement in placing MPAs can help meet many ecological goals, although biophysical data improve the conservation value of sitings.

To assess the need for MPAs in British Columbia (BC), Canada, I mapped stressors resulting from human activities. This produced a powerful rationale for MPAs: very little of the ocean, and almost none of the continental shelf of BC, lies beyond the reach of human stressors.

My work helps reconcile differing perspectives about the efficacy of community-based vs. science-based MPA selection. I explored and analyzed these approaches, separately and together, in two areas in BC. First, I generated a community-based plan for MPA placement through partnerships with two First Nations (indigenous peoples) in BC. They offered strong support for spatial protection measures, and individuals nominated overlapping areas. Second, I applied a decision support tool (Marxan) to determine MPA placement under scientific precepts. Conservation planning usually lacks detailed ecological information but the Marxan approach was robust to some missing data; in such cases, it was best to use available abiotic and biotic data to ensure that both habitats and species were represented. Third, I integrated community-based and science-based approaches, to find that they verified and complemented each other. Indeed, an integration of the two was preferred by participants and also achieved all conservation objectives.

Finally, I took a novel and pragmatic approach to ocean zoning. I used spatial data for thirteen commercial fisheries on Canada's west coast to select areas where fishing should be permitted, rather than prohibiting fishing under a MPA paradigm. The results revealed that small reductions in fisheries yields, if judiciously selected, could allow creation of large unfished areas that embraced diverse biophysical regions and habitat types. Such a pragmatic approach could achieve remarkable conservation gains.

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Co-authorship statement

With the exception of Chapters 1 and 8, all chapters have been prepared as stand-alone, peer-reviewed publications. Chapters 2 to 7 are either published, in press, accepted, or in review at peer reviewed journals or conference proceedings. I am the senior author on all papers. I took primary responsibility for design, implementation, analysis, and writing of all co-authored chapters. I am the sole author on Chapter 2 and 5, although Dr. Amanda Vincent provided guidance. Details of co-authorship contributions for Chapters 3, 4, 6 and 7 are outlined below.

Chapter 3: My co-author Dr. Jacqueline Alder developed the original idea of mapping marine wilderness areas, in order to compare the intactness of the marine environment to a similar terrestrial assessment being undertaken by the Province of British Columbia. I embraced the opportunity to transform this idea into reality, developed a much more complex methodology than originally envisioned, and shaped it into a context chapter for my thesis. I expanded on the original idea, developed and carried out all analyses, and drafted the text. My co-author contributed the initial idea and commented on multiple versions of the manuscript. She also facilitated access to the data.

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1. Introductory chapter

Rationale

The impact of humans on wild ecosystems, particularly common-pool resources such as the oceans, is tremendous and growing (Halpern et al. 2008; Kennedy 2003; Newton et al. 2007; Palmer et al. 2004; Weinstein et al. 2007). On land, between one third and half of the landscape has been transformed by human action (Vitousek et al. 1997). In the marine environment, we are confronted with collapsing fish stocks (Jackson et al. 2001; Myers & Worm 2003; Newton et al. 2007) and an increasingly evident loss of biodiversity and degraded areas such as dead zones (Sala & Knowlton 2006; Schiermeier 2002). Such changes are altering ecological communities (Vitousek et al. 1997) and impacting humans through the loss of ecosystem goods and services (Millennium Ecosystem Assessment 2005). These changes are perpetuating poverty and are affecting food security (Millennium Ecosystem Assessment 2005). For the benefit of biodiversity and people, action must be taken to prevent further degradation, and recover degraded ecosystems.

This thesis focuses on marine protected areas (MPAs) as one of the strategies for marine conservation. I address the major challenge of placing MPAs in such a way that they are both socially acceptable and ecologically sound. Below I draw upon the experiences of terrestrial and marine environments, outline the rationale for and shortcomings of MPAs, and summarize the debate on park selection.

Terrestrial and marine conservation strategies

Different conservation strategies can be used to reduce impacts on ecosystems and minimize or halt the loss of biodiversity. Biodiversity conservation approaches fall into four categories: protection and management, law and policy, education and awareness, and changing incentives (Salafsky et al. 2002). The first of these – protection and management – is an in situ strategy, whereas the others can be in situ or ex situ. The establishment of parks and protected areas is one of the most common conservation strategies to stem biodiversity declines (Vanclay et al. 2001), and hence is the focus of this thesis. Other specific strategies include restoration (Dobson et al. 1997), re-introductions or relocations (Donlan et al. 2005; Rosenzweig 2003), direct payment for conservation (Ferraro & Kiss 2002), active management (*e.g.,* invasive species removals (Krajick 2005), culling (Koenig 2007), fire management (Keeley et al. 1999)), and ex-situ conservation (Cohen et al. 1991), amongst others. The most effective strategy will depend on the threats to the ecosystem, and the vulnerability of species and ecosystems to those threats (Brooks et al. 2006).

Marine conservation has also employed multiple strategies, although the difficulty of working in the marine environment has meant that fewer strategies have been applied than on land. The main marine conservation strategies are MPAs and fisheries management, although others such as management of invasive species in ballast water and restoration of degraded habitats exist as well. The most commonly used definition of a MPAs is that of the World Conservation Union: "any area of the intertidal or subtidal terrain, together with its overlying water and associated flora, fauna, historical and cultural features, which has been reserved by law or other effective means to protect part or all of the enclosed environment" (Kelleher & Kenchington 1992). This definition encompasses fully protected areas, partial protection, or some combination thereof through zoning. Fisheries management strategies have included single-species stock assessments, multiple species assessments, gear restrictions, area closures (including MPAs), and some combination thereof termed as ecosystem-based management or integrated coastal zone management (Leslie 2005; Pikitch et al. 2004; Walters & Martell 2004). International agreements and accords prompt marine conservation action, but implementation is primarily carried out through MPAs and fisheries management. For example, the international listing of marine species has become a conservation strategy to encourage or require action, through the IUCN Red List (Vincent & Hall 1996) and the Convention on International Trade in Endangered species (CITES). The listing of seahorses as the first fully marine fish on CITES initiated CITES as a marine conservation tool (Foster & Vincent 2005). The management after such a listing, however, is done primarily through the main marine conservation strategies listed above. Some conservation strategies that are common on land are much less common in the marine environment, such as reintroductions and habitat restoration.

Terrestrial parks and insights for the marine environment

Parks, while not perfect, have been effective in helping stem destruction of habitats and biodiversity loss (Bruner et al. 2001; Vanclay et al. 2001). Terrestrial parks have been a successful strategy because they address the most significant terrestrial threat of habitat degradation (Bruner et al. 2001). However, parks are only as effective as compliance by people permits, with poaching and illegal clearing potentially undermining biodiversity benefits (Bruner et al. 2001). Terrestrial parks were found to be in better condition than their surroundings with respect to land clearing, but were still impacted by logging and hunting, although at reduced rates from non-park areas. Support by communities surrounding parks, or affected by parks, is a key to their success. Despite some shortfalls, parks are one of the most effective biodiversity conservation tools, and therefore should remain a central component of conservation strategies (Bruner et al. 2001).

The history of terrestrial parks may provide some insight into the current challenges of MPA design and establishment. Traditional marine tenure systems aside (Asafu-Adjaye 2000), the establishment of MPAs is a relatively recent phenomenon; terrestrial parks have a longer history (Lovejoy 2006). In Europe, hunting reserves were some of the first parks established. Their primary purpose was to allow hunting opportunities for the elite (Evans 1997), while in North America early national parks were selected primarily for their scenic beauty (Shafer 1999). Planning for national parks was initially non-existent, although later the focus was shifted to creating systems of parks (Shafer 1999). The shift from establishing single parks, to focusing on systems of parks that represent biodiversity and habitats seems to have preceded a similar shift in the marine environment. On land, there are limited options for park expansion due to ownership of land (Vitousek et al. 1997); in the marine environment, the options for establishing MPAs may be more flexible because oceans are predominantly common property (Costanza 1999).

One of the big debates in the terrestrial park literature is about the role of people within national parks (Locke & Dearden 2005; Martino 2005). Expropriations and expulsions of people from parks were common in the early days in North America (Searle 2000), and are still occurring in developing countries (Barume & Jackson 2000). Some authors argue that using parks for the benefits of local people to alleviate poverty will result in further biodiversity losses. Instead, they insist, strict protected areas need to be maintained, created, and expanded, if biodiversity is to persist (Locke & Dearden 2005). Others argue that the idea of wilderness is no longer a reality, that most biodiversity occurs outside of parks (Martino 2005), and that parks will only be effective with the support of people (Phillips 2003). In the marine environment, people are also excluded from some forms of MPAs (*i.e.,* no-take MPAs), but because people do not live within MPAs, such exclusions pertain to their uses of the marine environment and not their residences. As on land, the debate about what uses are admissible within MPAs is likely to occur whenever the type of protection is open for discussion.

The terrestrial parks literature acknowledges that social factors are crucial to park effectiveness (Bruner et al. 2001). MPA practitioners and researchers are hearing the message: Social factors are often highlighted as the primary determinants of MPA success (*e.g.,* Drew 2005; Kessler 2004; Mascia 2003; Morin Dalton 2001). In the 1990s concerns for incorporating social factors into MPA planning and establishment started being voiced more prominently (*e.g.,* Agardy 1994, Ballantine 1995, Kelleher & Kenchington 1992). For example, the Fenner Conference of 1991 discussed a strategic approach for MPA establishment in Australia that included social factors (Ivanovici et al. 1991).

Recent case studies of MPA selection experiences repeatedly confirm the importance of engaging those affected by the designation (Drew 2005; Helvey 2004; Kessler 2003; Lundquist & Granek 2005; Morin Dalton 2001). Indeed, human communities and their willingness to abide by conservation regulations are a major determinant of the effectiveness of MPAs. Yet many MPA studies pay lip-service to the human component of conservation (*e.g.,* Roberts et al. 2003), instead placing most emphasis on ecology (Christie 2004). Clearly, both the ecology of an area and social considerations are crucially important in designing effective MPAs (Roberts 2000).

Some conceptual developments about parks incorporate both terrestrial and marine environments (Garnett et al. 2007). In particular, many integrated conservation and development projects (ICDP) include coastal and marine areas (Brown 2002). Such projects were prevalent in developing countries starting in the 1980s and 1990s (Alpert 1996, Garnett et al. 2007). As the name implies, ICDP marries conservation and development, largely because neither had done well on its own (Alpert 1996, Sinclair et al. 2000). The applicability of ICDP to the marine environment indicates that concern about social factors in MPA establishment is not very new. Similar concepts also encompass terrestrial and marine environments, such as integrated coastal zone management, ecosystem-based management, and community-based conservation. The results of these endeavours have been mixed, with some successes and some failures (Berkes 2004, Garnett et al. 2007, Inamdar et al. 1999).

While there are similarities, there are also some fundamental differences between terrestrial and marine parks, especially the expectations of parks. The primary goal of terrestrial parks is understood to be biodiversity protection (Bruner et al. 2001). The emphasis in the marine environment has likewise been biodiversity protection, yet a core argument for the establishment of MPAs has been that MPAs will also provide fisheries benefits (Alcala et al. 2005; Roberts et al. 2001). Proponents of the establishment of MPAs, such as conservation organizations, routinely "sell" MPAs to stakeholders by emphasizing their fisheries benefits (*e.g.,* Gell & Roberts 2003). The expectation is thereby set that MPAs would fail if they do not provide such benefits in the form of larval and/or adult spillover. The main impetus for the establishment of terrestrial parks was the romantic notion of wilderness. However, in some cases the establishment of terrestrial parks began partly due to a concern about the effect hunting was having on wildlife (Lubchenco et al. 2002). Yet on land there is no documented expectation for existing or newly established parks to result in spillover of animals to improve hunting outside of parks. More recently, there have been arguments for establishing corridors, but their purpose is to connect protected areas, not to enhance spillover (Rouget et al. 2006). While differences in connectivity might in part explain the expectation of fisheries benefits from MPAs, it will be harder to meet the expectations set for MPAs compared to terrestrial parks.

Rationale for marine protected areas

Parks in the ocean, or marine protected areas (MPAs) are becoming particularly popular. Given the main marine threats of fisheries and habitat destruction (primarily by fishing gear) (Pauly et al. 2003; Pauly et al. 2002), coupled with the past failures of traditional fisheries management (Wappel 2005), MPAs have received a lot of focus to advance marine conservation (Sala et al. 2002), and are increasingly being promoted as a solution to protecting marine biodiversity and slowing the decline of fish stocks (Lubchenco et al. 2003; Sladek Nowlis & Roberts 1999). It is frequently noted that MPAs should be an addition to – not a replacement of – traditional fisheries management (Lubchenco et al. 2003; Sladek Nowlis & Roberts 1999). Many countries have signed international agreements, committing to establishing networks of MPAs (CBD 2006; Wood et al. 2007; World Summit on Sustainable Development 2003).

MPAs are popular for many reasons: From an ecological perspective, they can be effective at conserving biodiversity (Dayton et al. 2000; Micheli et al. 2004); from a social perspective, they may provide economic benefits, including some evidence for enhancement of fisheries (Halpern & Warner 2002; Roberts et al. 2001; Tetreault & Ambrose 2007), and they can promote community engagement in conservation (Russ & Alcala 1999) and thereby improve compliance.

Scientific claims that MPAs, especially marine reserves, work in terms of biodiversity conservation and replenishment is growing (Halpern 2003; Halpern & Warner 2002; Micheli et al. 2005; Micheli et al. 2004; Mosquera et al. 2000; Tetreault & Ambrose 2007). In a review of 89 separate studies of the effectiveness of marine reserves, four biological measures were significantly higher inside reserves compared to outside (Halpern 2003), showing that depleted areas rebound when fishing pressure is removed. Regardless of size, marine reserves led to increases in density (double), biomass (triple), individual size (20-30% higher), and diversity in all functional groups (20-30% higher) (Halpern 2003). The performance of reserves is even better if one considers only long-term studies of well-managed reserves where enforcement is high (*e.g.,* review by Gell & Roberts 2003). For example, in the Leigh Marine Reserve in New Zealand, densities of fishable size individuals of an exploited stock were 5.8 to 8.7 times higher in the reserve compared to fished areas nearby (Babcock et al. 1999). As another example, densities of large predatory reef fish increased 7-fold in 11 years of protection in the Apo Island reserve in the Philippines (Russ & Alcala 1996), and mean biomass was 1000% higher for the largest size classes of legal-sized targeted fishes in reserves in southern California (Tetreault & Ambrose 2007).

Empirical data and modeling suggest that marine reserves would increase yield in overfished areas through spill-over (Guenette et al. 1998; Manriquez & Castilla 2001; Neubert 2003; Sladek Nowlis & Roberts 1999). The reported ecological benefits of marine reserves are export of biomass to fished areas, increase in spawning stock biomass within the reserve in turn leading to larval dispersal outside of reserve boundaries (Sladek Nowlis & Roberts 1999; Tetreault & Ambrose 2007), and restoration of more natural size-frequency distributions of the protected populations (Chiappone & Sullivan Sealey 2000). Marine reserves may also increase juvenile survivorship (Lindholm et al. 2001). Reserves can therefore provide an insurance mechanism against the failure of conventional single-species fisheries management (Agardy 2000; Guenette et al. 1998; Walters 1998), and against natural and man-made catastrophes (Allison et al. 2003; Ballantine 1997). Many of these studies, however, are based on the results of models rather than empirical observation (see review by Russ 2002), or assume no fisheries regulations outside of MPAs (Hilborn et al. 2006).

MPAs can empower communities to engage in conservation and provide other economic benefits. In many developing countries, communities have been embracing MPAs as a conservation tool (Russ & Alcala 1999; Russ et al. 2004). In addition to potential fisheries benefits discussed above, MPAs can attract tourists by becoming diving destinations (Russ & Alcala 1999), thereby providing economic benefits. It is not clear, however, whether economic benefits through fisheries and tourism are able to offset lost fishing opportunities through the establishment of the MPA. Also, while some communities unite to support MPAs, in others MPAs can be a divisive issue (Lien 1999).

In recent years, research interest on community engagement and MPA effectiveness has been growing (Aswani & Hamilton 2004; Christie 2004; Christie et al. 2003; Cinner & Aswani 2007; National Centers for Coastal Ocean Science 2007). This research highlights the importance of understanding motivations for interests in marine conservation, and reactions to establishment of MPAs (Christie 2004; Christie et al. 2003; Christie et al. 2002; Pollnac et al. 2001). Indeed, MPAs can be biological successes (*i.e.*, result in increases in biomass, diversity, etc.) while being social failures (Christie 2004). Long-term sustained successful MPAs will require the engagement of, and benefits to, local communities.

For community participation to lead to well-managed MPAs, participation has to be meaningful. Community engagement can be carried out in many different ways, with varying degrees of involvement by communities. Arnstein's ladder of citizen participation is commonly invoked when describing community participation (Arnstein 1969). Of the eight rungs of participation on the ladder, the top three provide power to citizens or communities (in descending order: Citizen control, delegated power, partnership), whereas the other rungs are either tokenism or manipulation (Arnstein 1969; Carlsson & Berkes 2005).

Shortcomings of MPAs

MPAs are not a panacea for all that ails the ocean (Allison et al. 1998). For fisheries in particular, it is not clear whether MPAs can provide enough benefits through spillover to make a substantial difference to fisheries – the biomass that spills over may not be enough to compensate for effort displacement, and MPAs therefore should augment, not replace, fisheries management (Sale et al. 2005). Not all ecosystem components benefit equally from protection, either; trophic cascades have been observed in several cases (Mumby et al. 2007; Pinnegar et al. 2000; Shears & Babcock 2003), and highly mobile species may benefit less from protection (Hilborn et al. 2004). Also, MPAs can do little to mitigate some threats. Pollution and invasive species, for example, ignore MPA boundaries (Boersma & Parrish 1999).

While evidence is mounting that MPAs replenish depleted populations, these increases may not be enough to sustain fishing. A ten-fold increase in biomass might sound impressive, but if this entails an increase from 1kg to 10kg, one fishing event could wipe out the increase. For MPAs to sustain fisheries in areas where accurate stock assessment is not possible, we would need to reverse our thinking and consider the ocean closed to fishing, with only a few areas open for fisheries (Walters 1998). Also, the effectiveness of MPAs compared to fished areas depends on the level of depletion and management outside of the MPA (Hilborn et al. 2006). If depletion is high outside of the MPA, then replenishment inside the MPA is going to be relatively higher than a similar MPA where outside depletion is slower. Nevertheless, studies on MPAs confirm that the removal of fishing pressure does allow the areas to recover compared to fished areas.

Successful establishment and management of MPAs is contingent upon a range of social factors: compliance and/or enforcement, political support, and economic incentives. Without effective management and compliance of MPAs, little ecological benefit will result from protection (Jameson et al. 2002). Effective management and compliance is facilitated by the support of user groups and communities affected by the MPA (Walmsley & White 2003). Enforcement is expensive in the marine environment, especially in offshore areas, and hence compliance through peer pressure is the preferred option. Political support at all levels involved in ocean management is essential for MPAs to be established, and for subsequent effective management (Guenette & Alder 2007; Jessen & Ban 2003). Sufficient funding is also necessary, and political and public support can facilitate the availability of funds. Economic incentives, in particular to compensate user groups for lost fishing and other revenues, can be essential to building support for a MPA. Because there are many social factors necessary for the success of MPAs, the potential for failure exists if one or more of these issues are not met.

Despite these shortcomings, MPAs remain one of the most practical and effective ways to conserve marine biodiversity. MPAs provide biodiversity benefits, and they can be socially feasible.

Debate about approaches to selecting parks and MPAs

At the heart of discussions about MPAs lies the question of how to select socially acceptable and ecologically viable places for protection. There is an ongoing debate in the peer-reviewed literature about the benefits of systematic conservation planning (as defined by Margules & Pressey 2000) versus "ad-hoc" or "opportunistic" methods of selecting protected areas (Knight & Cowling 2007; Pressey 1994; Roberts 2000; Stewart et al. 2007; Stewart et al. 2003). On the one hand, scientists argue that representing biodiversity is of utmost importance, and that systematic approaches to selecting MPAs are more efficient – and hence better – at selecting such areas (Pressey 1994; Pressey et al. 1996). Because existing, opportunistically selected MPAs are theoretically less efficient, a much larger total area for protection is required when such existing parks are built upon to represent biodiversity than if they are excluded (Stewart & Possingham 2005; Stewart et al. 2003). Efficiency is defined as achieving biodiversity objectives at the lowest possible cost, where the cost can be foregone revenue, commercial fisheries, area, etc. Because the area needed for selection is larger when existing parks are included, in theory the cost of representing biodiversity is higher for stakeholders. Thus the argument is made that it would be preferable to identify all new parks and MPAs using systematic conservation planning. On the other hand, existing MPAs have been shown to be effective (Halpern & Warner 2002), and they have been established within the context of socio-economic-political realities (Fernandez & Castilla 2005; Roberts 2000). Because the establishment of protected areas will always take place given such realities, the counter argument is that we may as well embrace such areas as opportunities and encourage their establishment (Knight & Cowling 2007; Roberts 2000).

In both terrestrial and marine parks, the challenge is to balance conservation with human need. The contemporary tendency, especially in the MPA literature, has been to focus on the scientific imperative, assuming that such an approach will accomplish conservation goals. But what about stakeholder preferences, which clearly determine conservation effectiveness, in the absence of armed guards? And if we do incorporate these, how do they affect our capacity to achieve ecological integrity? My thesis considers just such matters

Research objectives

This thesis addresses the major challenge of placing MPAs in such a way that they are both ecologically sound and socially acceptable. The core research question of my thesis was as follows: how do different approaches to prioritizing places for marine protection compare? My first objective was to enhance our capacity for conservation planning in the marine environment. The second objective was to compare and integrate science-based and community-based approaches of MPA selection. The following specific questions guided this research:

1. How can anthropogenic stressors be mapped in the marine environment? (Objective 1)
2. What are community-based perspectives and objectives for marine protection? (Objective 2)
3. What kinds of data, and how many datasets, are necessary to make science-based systematic conservation planning practical? (Objective 1 and 2)
4. How do community-based and science-based approaches compare, and how can they be integrated? (Objective 1 and 2)
5. Are there spatial marine conservation approaches beyond MPAs that are promising? (Objective 1)

Research framework

Because ecological and social perspectives are important in successful MPAs, the placement of MPAs is most suitably studied using an interdisciplinary approach (Newell 2001). I investigate a multi-faceted and complex problem – protecting the ocean – from several angles, using interdisciplinary techniques from conservation biology.

Conservation biology aims to be a synthetic field that applies the principles of ecology, biogeography, population genetics, economics, sociology, anthropology, philosophy, and other disciplines to the maintenance of global biological diversity (Meffe & Carroll 1994). The discipline of conservation biology recognizes that diverse and functioning ecosystems are critical not only to the maintenance of the few species we harvest, but also to the survival of the little-known and yet-to-be-discovered life forms. Conserving biodiversity is important to ensure the life-support system for the planet, which is critical for our own continued survival and well-being as a species (Balvanera et al. 2001; Daily 1997; Meffe & Carroll 1994). The goal of conservation biology is thus to ensure that humans do not extinguish genes, species, or ecosystems, as these may be important for future adaptability (Norse 1993). In the context of this thesis, the work of conservation biologists informed the ecological objectives needed for prioritize MPAs.

While conservation biology purports to incorporate some of the social sciences, in reality much of the emphasis has been on the ecological perspectives of conservation (Christie 2004; Robinson 2006). I used the framework of community-based conservation to inform the people-centric component of this thesis. Community-based conservation arises from within the community, or at least at the community level rather than internationally or nationally (Western & Wright 1994). Community-based conservation reverses top-down driven conservation by focusing on the people who bear the costs of conservation. Therefore, in theory community-based conservation includes natural resources or biodiversity protection by, for, and with the local community (Campbell & Vainio-Mattila 2003; Western & Wright 1994). Experience suggests that community-based conservation has been successful in some areas (*e.g.,* Forgie et al. 2001; Pollnac et al. 2001; Salafsky et al. 2001), but less successful in others (Berkes 2004; Campbell & Vainio-Mattila 2003).

To carry out the community-based approach to envisioning MPAs, I developed partnerships with two indigenous groups (called First Nations) in British Columbia (BC), Canada. I focused on indigenous people because of their recently reaffirmed rights by the United Nations' adoption of the Declaration on the Rights of Indigenous Peoples (United Nations General Assembly 2007). In Canada, First Nations are considered a level of government and have constitutional rights to fish for food, social and ceremonial purposes and to be meaningfully consulted on resource management issues (Constitution Act 1982; Harris 2002). In addition, generations of First Nations peoples have lived in the study areas, therefore providing a more engrained knowledge of the marine environment (Ayers 2005), and marine resources are an integral part of indigenous cultures in coastal Canada (Garibaldi & Turner 2004; Turner et al. 2000). I also focused on indigenous communities because they are becoming increasingly proactive in planning their marine areas. To get the perspectives of indigenous people in the two study areas, I carried out semi-structured interviews with individuals, and held community meetings.

To implement the science-based approach to prioritizing MPAs, I use the theory of systematic conservation planning (Margules & Pressey 2000), a sub-field of conservation biology. Systematic conservation planning is typically facilitated by computer programs called site selection algorithms, which provide options for achieving conservation objectives. I use the site selection algorithm Marxan (Ball & Possingham 2000; Possingham et al. 2000). I chose Marxan over other algorithms for several reasons. First, Marxan is the most commonly used reserve selection algorithm in the world (The Ecology Centre 2004), and its popularity ensures that my conclusions will be applicable to most of the world's MPA planners using systematic techniques. Second, Marxan has the ability to provide multiple solutions to meet planning objectives, which provides much-needed flexibility when addressing conservation problems. Third, Marxan has the ability to allow the clustering of conservation solutions into sizes that are realistic for conservation planning. And finally, the cost option within Marxan is flexible, and allows for the inclusion of a variety of costs.

Thesis outline

In total there are eight chapters in this thesis, consisting of five data-based research chapters. The present opening introductory chapter (Chapter 1) provides the context, research questions and framework for the thesis. A more thorough literature review of MPA selection approaches follows (Chapter 2). In Chapter 3, I develop a methodology for mapping human stressors in the marine environment, while simultaneously providing an overview of the current condition of the marine environment in BC. This chapter provides a rationale for the need for MPAs in BC. Chapter 3 is one contribution to my first objective of enhancing the field of systematic conservation planning in the marine environment.

In the next three chapters (Chapters 4, 5, and 6), I address my second objective of comparing and integrating science-based and community-based approaches of MPA selection. In Chapter 4, I focus on a community-based approach to envisioning MPAs in order to explore indigenous perspectives on spatial approaches to marine conservation, using the same case studies. In Chapter 5, I carry out a science-based MPA prioritization scheme in two case study areas in BC, in order to assess the data needs for such an approach. In Chapter 6, I bring together the previous chapters. I do this by integrating the community-based approaches (Chapter 4) and science-based (Chapter 5), while also incorporating the stressor mapping into MPA prioritization (Chapter 3). I use gap analysis to gauge the ecological effectiveness of the community-based approach, and I ask community members to rate the science-based approach. I then combine the two approaches in the selection tool Marxan, and again ask community members to rate the outcome. I also use two proxies for incorporating social perspectives into MPA prioritization: The human stressor map, and the relative importance of areas to commercial fisheries.

In Chapter 7, I reach beyond the conventional approach of selecting protected areas. I reverse the reserve selection approach by instead using Marxan to select permitted fishing areas. I assess this approach for its ability to meet the ecological objective of representation of species and habitats. Finally, I end with a synthesis chapter (Chapter 8) that summarizes the findings presented in this thesis, their limitations, and some recommendations for future work.

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2. Siting marine reserves: stakeholder-based vs. science-driven approaches[1]

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Introduction

Marine protected areas (MPAs), especially no-take MPAs (also referred to as marine reserves), are an emerging tool for marine conservation and management (Gell & Roberts 2003; Lubchenco et al. 2003), and many countries have committed to establishing such areas. Numerous international conventions and agreements encourage the establishment of MPAs. The latest of these is the Plan of Implementation from the World Summit on Sustainable Development (World Summit on Sustainable Development 2003), where signatory countries committed to establishing a representative network of MPAs by 2012. A set of MPAs is considered a network when connectivity has been taken into account. However, so little is known about marine connectivity that most common uses of the term "network" refer to a set of MPAs. Because many countries are moving towards establishing networks of MPAs, there is a growing need to provide guidance on how to implement such networks.

Background on MPA site selection

A range of approaches to site selection is emerging in the MPA literature, with most emphasis on either of two extremes: (1) stakeholder-based site selection (Ballantine 1997, 1999; Salmona & Verardi 2001; Walmsley & White 2003), and (2) science-based, systematic selection of sets of MPAs (Leslie et al. 2003; Roberts et al. 2003a; Stewart et al. 2003). In many cases, a combination of these site selection approaches is used, incorporating both stakeholder and science approaches, with differing levels of emphasis on either end of the spectrum. Here I review the main arguments for stakeholder-driven and science-driven MPA site selection approaches, and provide examples of such approaches for single MPAs and sets of MPAs.

The argument for stakeholder-driven MPA selection

Most MPAs had a champion (*e.g.,* a community, stakeholder group, individual, government agency) that helped bring designation to fruition (Pollnac et al. 2001). The location of most of these MPAs was determined by social criteria and opportunistic action (Roberts 2000). This one-at-a-time approach is often labelled as *ad hoc* (Stewart et al. 2003) because selection tends to be made in the absence of clearly identified selection criteria.

To date, most MPAs have been selected on a case-by-case basis, not as networks or sets. While systematic approaches to designing MPAs usually refer to science-based approaches, stakeholder-based processes can be used to create sets of MPAs. For example, multi-criteria analysis can be used to enhance stakeholder decision-making in the MPA process (Brown et al. 2001), a semi-quantitative approach for MPA selection has been proposed (Levings & Jamieson 1999), and participatory socioeconomic analysis has been used to elicit fishers' knowledge for site selection (Scholz et al. 2004).

Four main arguments support stakeholder-driven MPA selection approaches: (1) opportunistically chosen MPAs have performed better biologically than areas without protection; (2) MPAs will only be effective with community support; (3) stakeholder-driven MPA selection is capable of incorporating a multitude of factors in deciding on placement of MPAs; and (4) the placement of the early MPAs may not be important, as gaps can be filled later.

First, it is argued that opportunistically chosen MPAs have performed better biologically than adjacent areas open to fishing. Such opportunistically chosen reserves have shown an increase in biomass, biodiversity, and size of fish inside the reserves (Halpern & Warner 2002), have provided habitat-quality improvements (Rodwell et al. 2003), and have enhanced fisheries (Russ 2002). Opportunistically chosen MPAs can therefore be considered effective in achieving conservation and fishery objectives. At this time it is not known whether areas identified using science-driven selection would perform better than opportunistically chosen sites, or vice versa.

Second, it is frequently stated that MPAs will only be effective with community and stakeholder support, most commonly achieved through citizen participation in decision-making. This argument is supported by several case studies. For example, high levels of community participation led to increased levels of success in MPAs in the Philippines (Pollnac et al. 2001). Support by communities in proximity to the MPA, education and enforcement are considered important to the establishment and effectiveness of MPAs (Chiappone & Sullivan Sealey 2000; Roberts et al. 2003b; Walmsley & White 2003). While the contribution of participation to effective environmental decisions is primarily anecdotal, a few studies do provide quantitative evidence of improved effectiveness because of stakeholder participation (Beierle & Konisky 2001; Brody 2003; Pollnac et al. 2001; Salafsky et al. 2001). Yet support by stakeholders does not guarantee an effective MPA if support is not universal. For example, if MPAs are stakeholder-driven in the absence of consensus, they may end up as paper parks: Legally protected but in reality ineffective because of a lack of compliance and enforcement. But because stakeholder-driven approaches by definition involve at least some stakeholders in siting discussions, they may have a greater chance of consensus and support. If a set of MPAs is designed using a science-driven, systematic site selection approach, and communities are not supportive, then any such area may not protect the marine environment even if considerable enforcement expenses are incurred (Levings & Jamieson 1999; Walmsley & White 2003).

Third, stakeholder-driven selection approaches often use a multitude of information that is not incorporated into science-driven selection. People make the choice of where to place sites based on their own experience, their own knowledge of the region (*e.g.,* local ecological knowledge), available biological information, economic importance of the area, and much more, depending on the individual. For example, while science-driven approaches do not currently account for a system's historical productivity or biodiversity, people may be capable of selecting such sites. Areas that were once very diverse before being degraded may have the potential to recover, but would be missed by a science-driven approach that does not include such information. However, with improved information, such elements could be incorporated into the science-driven approach.

Fourth, the placement of the initial marine reserves may not be important, as gaps can be filled later. Therefore, arguably if there is an opportunity to establish a marine reserve, action should be taken, rather than delaying implementation. Terrestrial experience has shown that historically the development of a system of parks can be divided into five steps: (1) ad hoc selection, (2) scientific basis for systematic selection of sites, (3) gap-filling by selecting additional sites using systematic methods, (4) scientific basis for a connected network, (5) completion of networked system of protected areas (Willison 2001). Most countries have barely begun step 1 in the marine environment, whereas in many countries terrestrial parks are at step 3 (Willison 2001), and the opportunity will remain to complete a system of marine reserves later by filling gaps. This does indicate, however, that the stakeholder-based approach will need to be combined with a science-driven approach in the later stages of the steps outlined above to identify the gaps that need to be filled.

<u>The argument for science-driven MPA site selection</u>

Systematic site selection is meant to ensure that conservation targets are met when designing a set of reserves (Margules & Pressey 2000). With increasing complexity in biological, ecological and biophysical information, and multiple conservation goals (*e.g.,* representation, redundancy, resilience), systematic conservation planning can assist with the selection of a set of sites by using explicit methods for locating reserves. Systematic site selection is particularly helpful when selecting sets of sites because of the complexity involved. It is science-driven because the best available conservation information and spatial datasets (*e.g.,* species distributions, habitats, ecoregions) are used to select the set of sites.

Mathematical models, or algorithms, have become a particularly popular method for science-driven site selection over the past 10 years (McDonnell et al. 2002), and this section will focus on such an approach. A science-driven approach has been used extensively in terrestrial conservation (Margules & Pressey 2000; Pressey et al. 1996), and is increasingly being adopted in the marine environment. Inherent in most site selection algorithms is the assumption that competing uses limit the number of protected areas that can be established, and therefore the minimum area required to achieve the objectives is the most efficient solution. Efficient in this context refers to achieving the objectives in the smallest possible area (Leslie et al. 2003).

Three main arguments are used for science-based site selection approaches: (1) sets of MPAs selected are more efficient; (2) sites selected achieve representation targets; and (3) because the approach is based on scientific data, the decision can withstand public pressure.

First, it is argued that science-driven approaches to site selection of sets of reserves are more efficient than *ad hoc* approaches (Leslie et al. 2003; Stewart & Possingham 2002). Minimizing the area incorporated into a set of reserves is advantageous in a situation where stakeholders or politicians are only willing to set aside a small proportion of the available area, or in terrestrial situations where the cost of acquiring land has to be minimized (McDonnell et al. 2002). Costs other than area can be used in the marine environment as well, such as the importance to fisheries, or polluted areas. The site selection algorithm would then find areas that meet biodiversity objectives while minimizing the areas of importance to fisheries, or minimizing polluted areas in the selected reserve system. Usually, however, area gets used as the cost in the marine environment. While it has been shown that site selection algorithms indeed select the smallest area to satisfy objectives, this may not be the best outcome from a conservation perspective. Generally, more and larger areas are preferable for ensuring the long-term persistence of biodiversity (Cabeza & Moilanen 2001).To a certain extent this can be compensated by adjusting the inputs into the algorithm so that a given area (*e.g.,* a certain percentage of each habitat or of the total area) is incorporated into the outcome. It is then up to the algorithm user to determine what percentage or target has to be incorporated into the set of reserves.

Second, in order to protect biodiversity, MPAs must be representative of biodiversity. The science-based approach is able to ensure that all components of biodiversity for which spatial data exist are included in the reserve system. Because science-driven site selection approaches can be used to target sites of highest biological value, such an approach is effective at capturing high biodiversity sites at the species level (Pressey et al. 1997). Some even argue that biological criteria must precede socioeconomic evaluation (Roberts et al. 2003b). Conversely, when socioeconomic criteria are given equal or greater weight than ecological considerations in the design of a system of MPAs, this can lead to the selection of reserves of little biological value or that are not representative (Roberts et al. 2003a; Roberts et al. 2003b), and possibly damage rather than enhance fisheries (Crowder et al. 2000).

Third, because systematic site selection uses biophysical data, it is often considered to be more objective than stakeholder-based site selection (Bedward et al. 1992; Leslie et al. 2003; McDonnell et al. 2002; Moore et al. 2003; Pressey et al. 1997), and can therefore be used to counter public pressure by providing a scientific rationale for the placement of MPAs. Stakeholders, especially fishers, may resist the establishment of marine reserves in the most productive habitats – commonly because they coincide with fishing grounds – and therefore the less productive habitats often get chosen as marine reserves if socioeconomic pressures are allowed to prevail in the site selection process (Crowder et al. 2000; Gell & Roberts 2003). The same may apply to other stakeholders – resistance will be higher if areas chosen for protection coincide with areas that are of current or future potential economic importance for that stakeholder group. Without scientific data about the underlying habitat quality and fish population structure, there may be no reason to defend placing reserves in any particular habitat patch (Crowder et al. 2000). Yet it needs to be acknowledged that more than science goes into human decision-making and behaviour in response to such decisions, and that the setting of objectives which are entered into the site selection algorithm is a subjective activity.

MPA site selection in practice

While the theoretical MPA literature portrays a dichotomy of site selection approaches – stakeholder-driven and science-driven, this dichotomy rarely exists in practice. Site selection takes place either through (1) the selection of individual MPAs, or (2) the selection of sets of MPAs. A gradient of approaches seems to exist, with individual MPAs most commonly being selected through stakeholder selection, and sets or networks of MPAs increasingly being selected through science-based approaches. A range of approaches has been used for the siting of individual MPAs, from predominantly stakeholder-based to science-focused. Although not as common, in some cases of stakeholders select sets of MPAs.

Studies that examine MPA management success emphasize the importance of community and stakeholder participation in selecting MPAs and making them viable (Walmsley & White 2003; White & Courtney 2002). For example, experience in the Philippines indicates that the decentralization of planning and management of local government units is a key to the establishment of MPAs in that country, because it allows communities (municipalities) to designate MPAs (White & Courtney 2002). Similarly, the participation of stakeholders in the MPA process is part of the reason why the Philippines has been able to designate so many no-take MPAs (White & Courtney 2002). Unfortunately, in many parts of the world documentation on selection criteria and factors leading to successful establishment and management is sparse (Pollnac et al. 2001), and it is therefore difficult to generalize about factors that lead to MPA success.

Systematic science-driven site selection has been applied primarily in planning or academic exercises. Mostly this approach has been applied in developed countries, presumably because it tends to be time-demanding (and therefore expensive), and requires reliable and current biological data, which can be non-existent or sparse, especially in developing countries. Conservation groups in particular have been quick to adopt science-driven systematic site selection to advocate certain sites: (1) the Nature Conservancy uses a site selection program in its ecoregional planning processes (The Ecology Centre 2004); (2) World Wildlife Fund and the Scripps Institute of Oceanography used an optimization model as a quantitative approach to recommend a set of MPAs design to protect rocky reef habitat in the Gulf of California, Mexico (Sala et al. 2002); (3) the Living Oceans Society in British Columbia, Canada, is using a site selection algorithm (Marxan) to suggest areas of high conservation utility (Living Oceans Society 2004); and (4) World Wildlife Fund Canada and the Conservation Law Foundation are applying a similar approach on the Scotian Shelf/Gulf of Maine (The Ecology Centre 2004). None of the above examples have to date resulted in the establishment of a set of MPAs.

Site selection algorithms are occasionally used in a consultative process leading to MPA establishment, combining systematic site selection approaches with stakeholder input. The site selection tool MARXAN was originally developed as a decision support tool for the Great Barrier Reef Marine Park Authority (Lewis et al. 2003), and has been applied in developing the new zoning plan along with extensive public consultations for Australia's Great Barrier Reef (Australian Government 2003; Lewis et al. 2003). The Channel Islands (California, USA) science panel used a site selection algorithm (Sites version 1) in an iterative process with stakeholders to suggest a set of reserves within the Channel Islands National Marine Sanctuary (Airamé et al. 2003). The Florida Keys Marine Sanctuary (Florida, USA) also used a site selection algorithm in combination with consultations, and a similar approach has been taken in the Galapagos Islands marine reserve in Ecuador (The Ecology Centre 2004). Time will tell the effectiveness of this approach, although to date there have been mixed results. For example, the process in Florida – while faced with challenges – appears to have been successful (Florida Keys National Marine Sanctuary 2003). The rezoning of the Great Barrier Reef Marine Park was recently completed with more than 10,000 submissions by the public and stakeholders (Day 2002). The process in the Channel Islands National Marine Sanctuary angered the sports fishing lobby, who sued the state government over the process (Recreational Fish Association 2002). In the Galapagos Islands, public demonstrations ensued, including vandalism to park facilities (Friends of Galapagos 2004).

Very few cases of stakeholder-based site selection of sets of MPAs are documented in the literature. In some cases, stakeholders assist in the design of zoning within MPAs, which may be considered a set of marine reserves on a smaller scale. For example, extensive stakeholder input was used to make recommendations for zoning the Seaflower Biosphere Reserve in Colombia (Friedlander et al. 2003) and the Asinara Island National Marine Reserve of Italy (Villa et al. 2002). Perhaps the spatial disconnect between local areas and larger geographic zones (*e.g.,* sets of MPAs) make it difficult to apply this approach to sets. Other countries are using the site-by-site approach to eventually achieve a set of MPAs (*e.g.,* New Zealand, Philippines).

Conclusion

Site selection processes – be they stakeholder- or science-driven – that have meaningful community and stakeholder participation appear to be most successful (Elliott et al. 2001; Gladstone 2000; Pollnac et al. 2001). The experience of countries successful in setting up no-take MPAs (*e.g.,* New Zealand, Australia, the Philippines) illustrates that communities and stakeholders turn into MPA advocates once they have had personal experience with MPAs. Countries intent on establishing networks of MPAs might therefore want to start designating some individual sites, thereby potentially making the designation of future sites faster and easier.

Increasingly scientists are calling for the establishment of networks of MPAs rather than single sites (*e.g.,* Roberts et al. 2003a). To adequately assess connectivity, larval dispersal, juvenile and adult migration patterns would have to be taken into account (*e.g.,* Gaines et al. 2003). Site selection algorithms are currently not able to incorporate connectivity into the selection process. Even if they could, our current lack of knowledge of marine connectivity is such that we do not even know the order of magnitude of dispersal (Palumbi 2004). As our understanding of connectivity improves, selection algorithms may be modified to account for connectedness amongst sites. While it is unlikely that stakeholder-selected MPAs will account for connectivity, some stakeholders, such as experienced fishers, may have a good understanding of movement patterns of species they are most familiar with. Stakeholders could use this information, as well as biological information as it becomes available, to select sites that are connected.

The question is not whether science should dominate the site selection process, or whether stakeholders should do the selection; rather, the question is when to bring in science or stakeholders. Each country and case may be different, but better documentation and analysis of experiences throughout the world, even if descriptive, will help in providing guidance to those places just starting to embark on the site selection process.

From the experience with single MPAs and sets, those that involve stakeholders early and often appear to be most successful in generating support for the protected area. Because funds for enforcement are typically limited, such support is crucial in eliminating or reducing non-compliance. The first step in involving stakeholders early may be to discuss the goals and objectives of the network or set of MPAs, including the issues and problems affecting the area in question. Stakeholder involvement and science-aided site selection can then be undertaken simultaneously. If stakeholders identify sites they would like to see protected, those can be incorporated into the selection algorithm, and additional areas identified that would complete the network. Similarly, if the science-driven selection process identifies sites that stakeholders find unacceptable, the process can be repeated excluding those sites to determine whether other areas can also achieve the objectives. Depending on the situation, the approach used should be adaptive and flexible to meet local needs.

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3. How wild is the ocean? Assessing the intensity of anthropogenic marine activities in British Columbia, Canada[2]

[2] A version of this chapter has been published: Ban, N., and J. Alder. 2008. How wild is the ocean? Assessing the intensity of anthropogenic marine activities in British Columbia, Canada. Aquatic conservation: marine and freshwater ecosystems 18:55-85.

Introduction

Most human activities in the ocean have a direct and/or indirect impact on marine life and habitat (Jackson et al. 2001; McIntyre 1995). The oceans are fished for economic benefit, subsistence, and recreation (Botsford et al. 1997; Cooke & Cowx 2006; Costanza 1999). Oceans serve as a major transportation network, coastal areas continue to have among the highest population densities, and people enjoy the ocean for pleasure and relaxation. All of these pressures on the oceans are having an impact (Kappel 2005; Roberts & Hawkins 1999; Solan et al. 2004; Verity et al. 2002; Vincent & Hall 1996). We are fishing down trophic levels (Pauly et al. 1998; Pauly et al. 2002; Pauly et al. 2005; Watson et al. 2004), predatory fishes have declined significantly (Devine et al. 2006; Myers & Worm 2003; Myers & Worm 2005) and pollution is prevalent (McIntyre 1995; Siboni et al. 2004).

The effect humans are having on the ocean has been well documented. For example, more than 70 percent of fisheries are fully exploited or overexploited (FAO 2004). Overfishing has occurred worldwide for many centuries (Jackson et al. 2001), and causes cascading effects on the pelagic food web (Scheffer et al. 2005) and other structural and functional changes (Bascompte et al. 2005; Hutchings 2000; Jackson et al. 2001; Myers & Worm 2003; Scheffer et al. 2005; Solan et al. 2004). Certain fishing techniques, such as bottom trawling, are known to damage benthic structures (Ardron 2005; Collie et al. 2000; Kaiser et al. 2005; Thrush & Dayton 2002; Thrush et al. 1998; Watling & Norse 1998). Recreational fishing may have similar effects as commercial fishing (Coleman et al. 2004; Cooke & Cowx 2006). Finfish aquaculture contributes to habitat destruction, introduces species and diseases, and further depletes wild fish stocks (Auditor General of Canada 2000; Krkošek et al. 2005; Milewski 2000; Naylor et al. 2003; Naylor et al. 2000; Naylor et al. 2001), while shellfish aquaculture can enhance algal growth rates, reduce food supply for other herbivores, and bias community composition towards fast-growing species (Broekhuizen et al. 2002; Gibbs 2004; Jamieson et al. 2001). Shipping, cruise ships and recreational boating affect marine fauna through noise (Commoy et al. 2005; Foote et al. 2004; Moore & Clarke 2002; Richardson & Malme 1995; United States General Accounting Office 2000), pollution and the introduction of non-natives in ballast water (Hampton et al. 2003; United States Environmental Protection Agency 2002), and alter shorelines and habitats through erosion and the water column through sedimentation (Stevens & Ekermo 2003). Infrastructure, such as ferry docks, marinas, anchorages, boat launches, docks, piers and moorages in the marine environment contributes to pollution (Backhurst & Cole 2000b; Nightingale & Simenstad 2001; Stevens & Ekermo 2003; Turner et al. 1997; Wendt et al. 1996), noise (Foote et al. 2004; Nightingale & Simenstad 2001), habitat damage (Backhurst & Cole 2000a; Milazzo et al. 2004; Stamski 2005), and reduces light levels (Macfarlane et al. 2000; Sanger & Holland 2002). Land-based activities also impact nearby coastal and marine areas. For example, the biggest source of marine oil pollution is urban and industrial run-off (Government of British Columbia 2006), and fauna in marine environments close to urban centres have elevated heavy metals in their tissues (Bolton et al. 2004). Mines result in elevated levels of heavy metals in the coastal environment as far away as 40 km from the coast (Hines et al. 2000), and acid mine drainage is toxic to marine flora and fauna (Grout & Levings 2001; Levings et al. 2004).

The impacts caused by human activities can be divided into three categories: physical, chemical, and biological change. Physical change is comprised of direct alterations to habitats, and includes damage from fishing gear, dredging, etc. (Nightingale & Simenstad 2001; Watling & Norse 1998). Noise from shipping and boating is also considered a physical change. Increased noise has been shown to cause marine mammals to change their feeding, diving and swimming habits (Croll et al. 2001; Foote et al. 2004). Chemical change includes the effects of pollution, such as introduction of nutrients and toxic materials (Costanzo et al. 2001; Je et al. 2004; United States General Accounting Office 2000). Biological change is effected through fishing, potentially resulting in trophic cascades (Jackson et al. 2001; Pauly et al. 1998), and also includes the introduction of disease and exotic species (Gibbs 2004; Naylor et al. 2003; Naylor et al. 2001).

A first step in managing marine resources effectively is to understand the influence humans are having on the ocean, which activities are having an impact, where those activities are taking place, and how far the stressors from those activities extend. Yet very few comprehensive analyses of the extent and spatial patterns of human activities in the ocean exist (but see Lumb et al. 2004). Previous studies have focused on identifying the impacts of activities such as fishing (Collie et al. 2000; Cooke & Cowx 2006; Jackson et al. 2001), mining (Levings et al. 2004), shipping (Hong et al. 2005; Stevens & Ekermo 2003), and aquaculture (Krkošek et al.

2005; Milewski 2000), but have rarely examined the collective contribution of multiple activities (but see Lumb et al. 2004).

Human impact is commonly mapped on land (Foley et al. 2005; Hannah et al. 1994; Sanderson et al. 2002), in ways that may be instructive in marine environments. On land, roads are routinely used as a proxy for human impact – areas distant from roads are considered intact – and advances in remote sensing and GIS facilitate such analyses (*e.g.,* Government of British Columbia 2006; Lee et al. 2003; Nelson & Hellerstein 1997). However, in marine environments it is more difficult to identify areas affected by humans because of the ephemeral and episodic nature of many activities. Also, while many of the human impacts on land are clearly visible from space (*e.g.,* logging, industrial development, urban centres), marine habitat impacts occur below the water, and are therefore not detectable using current remote sensing technology.

This paper explores the impact of human activities in the ocean, using the exclusive economic zone (EEZ) of British Columbia (BC) as a case study. The human use of the marine environment is mapped in order to identify patterns and intensity of use, providing an approximation of possible damage to marine life and habitats.

Methods

A geographic information systems (GIS) approach was used for data analysis (ESRI 2004, ArcGIS Version 9.0). The Albers Equal Area projection (NAD83) was used throughout the analysis, because it holds constant the areas on the maps.

Spatial data for marine activities from 1992 to 2005 were collated, for a total of thirty-nine data layers of human uses affecting the ocean. Data layers include commercial and recreational fishing areas, transportation and infrastructure uses, aquaculture, and land-based activities in the coastal area (see Table 3.1 for a complete list of datasets used). Data were obtained from federal and provincial government agencies; much of the infrastructure data were provided by the Province of British Columbia through the Terrain Resource Information Management (TRIM) data. Only very few spatial data were available for the period prior to 1992, and thus historical uses were not considered in this analysis. Spatial data were not available for all human uses.

Table 3.1: Data used, relative stressors beyond the location of occurrence, impact factor and calculation, and categories of impact

Type of data	Source of data	Stressor beyond location	Severity and duration of impact (0=least, 10=greatest)[3]		Impact= direct + 0.3 * indirect	Impact value (Jamieson and Levings 2001)	Category of impact			References
Baseline and protected areas			Direct	Indirect			Physical	Chemical	Biological	
BC coastline	Province of BC	N/A								
Marine Ecoregions	Province of BC	N/A								
Provincial protected area designations	Province of BC (ftp://ftp.gis.luco.gov.bc.ca/pub/)	N/A								
National parks	Parks Canada	N/A								
Rockfish conservation areas	Fisheries and Oceans Canada (http://www-heb.pac.dfo-mpo.gc.ca/maps/themesdata_e.htm)	N/A								
Marine Protected Areas	Fisheries and Oceans Canada (http://www.pac.dfo-mpo.gc.ca/oceans/mpa/Info_e.htm)	N/A								
Aquaculture										
Finfish aquaculture	Province of BC (ftp://ftp.gis.luco.gov.bc.ca/pub/coastal/)	High	9	6	10.8	Severity and duration of impact habitat rating	√	√	√	Auditor General of Canada 2000, Milewski, 2000, Naylor et al. 2000, Jamieson and Levings 2001, Naylor et al. 2001,Naylor et al. 2003,Krkošek et al. 2005

[3] Based on impact weighting scheme devised by Jamieson and Levings Jamieson GS, Levings CO. 2001. Marine protected areas in Canada - implications for both conservation and fisheries management. *Canadian Journal of Fisheries and Aquatic Sciences* **58**: 138-157.
* Activities not weighted by Jamieson and Levings (2001)

Type of data	Source of data	Stressor beyond location	Severity and duration of impact (0=least, 10=greatest)		Impact= direct + 0.3 * indirect	Impact value (Jamieson and Levings (2001))	Category of impact (physical, chemical, biological)			References
Shellfish aquaculture	Province of BC (ftp://ftp.gis.luco.gov.bc.ca/pub/coastal/)	Low	1	2	1.6	Severity and duration of impact habitat rating	√		√	Jamieson et al. 2001, Jamieson and Levings 2001, Broekhuizen et al. 2002, Gibbs 2004
Commercial Fisheries										Pauly et al, 1998, Thrush et al. 1998, Jackson et al. 2001, Jamieson and Levings 2001
Bottom trawling	Fisheries and Oceans Canada (DFO) 1996 to 2005 groundfish trawl data (# of sets; no data if less than 3 distinct vessels fished in a grid) in 10km by 10km grid	(None)	8 (destruction of substrate structure or structural, epibenthic species)	4 (sediment plume; loss of habitat for other species, bycatch)	9.2	Jamieson and Levings (2001), average of severity and duration of impact habitat and species rating	√		√	Watling and Norse 1998, Collie et al. 2000, Thrush and Dayton 2002, Ardron 2005, Kaiser et al. 2005
Commercial urchin	Province of BC (ftp://ftp.gis.luco.gov.bc.ca/pub/coastal/)	(None)	4 (some destruction of structural, epibenthic species)	3 (loss of habitat for other species, bycatch)	4.9	Average of "severity and duration of impact habitat" and "species" rating	√		√	

Type of data	Source of data	Stressor beyond location	Severity and duration of impact (0=least, 10=greatest)		Impact= direct + 0.3 * indirect	Impact value (Jamieson and Levings (2001))	Category of impact (physical, chemical, biological)			References
Commercial shrimp	Province of BC (ftp://ftp.gis.luco.gov.bc.ca/pub/coastal/)	(None)	4 (some destruction of structural, epibenthic species)	3 (loss of habitat for other species, bycatch)	4.9	Average of "severity and duration of impact habitat" and "species" rating	√		√	
Commercial sea cucumber	Province of BC (ftp://ftp.gis.luco.gov.bc.ca/pub/coastal/)	(None)	4 (some destruction of structural, epibenthic species)	3 (loss of habitat for other species, bycatch)	4.9	Average of "severity and duration of impact habitat" and "species" rating	√		√	
Commercial scallop	Province of BC (ftp://ftp.gis.luco.gov.bc.ca/pub/coastal/)	(None)	4 (some destruction of structural, epibenthic species)	3 (loss of habitat for other species, bycatch)	4.9	Average of "severity and duration of impact habitat" and "species" rating	√		√	
Commercial salmon troll	Province of BC (ftp://ftp.gis.luco.gov.bc.ca/pub/coastal/)	(None)	4 (some destruction of structural, epibenthic species)	3 (loss of habitat for other species, bycatch)	4.9	Average of "severity and duration of impact habitat" and "species" rating	√		√	
Commercial salmon net	Province of BC (ftp://ftp.gis.luco.gov.bc.ca/pub/coastal/)	(None)	4 (some destruction of structural, epibenthic species)	3 (loss of habitat for other species, bycatch)	4.9	Average of "severity and duration of impact habitat" and "species" rating	√		√	

Type of data	Source of data	Stressor beyond location	Severity and duration of impact (0=least, 10=greatest)		Impact= direct + 0.3 * indirect	Impact value (Jamieson and Levings (2001))	Category of impact (physical, chemical, biological)			References
Commercial groundfisfh (other than bottom trawling)	Province of BC (ftp://ftp.gis.luco.gov.bc.ca/pub/coastal/)	(None)	4 (some destruction of structural, epibenthic species)	3 (loss of habitat for other species, bycatch)	4.9	Average of "severity and duration of impact habitat" and "species" rating	√		√	
Commercial squid	Province of BC (ftp://ftp.gis.luco.gov.bc.ca/pub/coastal/)	(None)	4 (some destruction of structural, epibenthic species)	3 (loss of habitat for other species, bycatch)	4.9	Average of "severity and duration of impact habitat" and "species" rating	√		√	
Commercial prawn	Province of BC (ftp://ftp.gis.luco.gov.bc.ca/pub/coastal/)	(None)	4 (some destruction of structural, epibenthic species)	3 (loss of habitat for other species, bycatch)	4.9	Average of "severity and duration of impact habitat" and "species" rating	√		√	
Commercial octopus	Province of BC (ftp://ftp.gis.luco.gov.bc.ca/pub/coastal/)	(None)	4 (some destruction of structural, epibenthic species)	3 (loss of habitat for other species, bycatch)	4.9	Average of "severity and duration of impact habitat" and "species" rating	√		√	
Commercial herring	Province of BC (ftp://ftp.gis.luco.gov.bc.ca/pub/coastal/)	(None)	4 (some destruction of structural, epibenthic species)	3 (loss of habitat for other species, bycatch)	4.9	Average of "severity and duration of impact habitat" and "species" rating	√		√	

Type of data	Source of data	Stressor beyond location	Severity and duration of impact (0=least, 10=greatest)		Impact= direct + 0.3 * indirect	Impact value (Jamieson and Levings (2001))	Category of impact (physical, chemical, biological)			References
Commercial herring roe	Province of BC (ftp://ftp.gis.luco.gov.bc.ca/pub/coastal/)	(None)	4 (some destruction of structural, epibenthic species)	3 (loss of habitat for other species, bycatch)	4.9	Average of "severity and duration of impact habitat" and "species" rating	√		√	
Commercial gooseneck barnacle	Province of BC (ftp://ftp.gis.luco.gov.bc.ca/pub/coastal/)	(None)	4 (some destruction of structural, epibenthic species)	3 (loss of habitat for other species, bycatch)	4.9	Average of "severity and duration of impact habitat" and "species" rating	√		√	
Commercial crab	Province of BC (ftp://ftp.gis.luco.gov.bc.ca/pub/coastal/)	(None)	4 (some destruction of structural, epibenthic species)	3 (loss of habitat for other species, bycatch)	4.9	Average of "severity and duration of impact habitat" and "species" rating	√		√	
Commercial geoduck	Province of BC (ftp://ftp.gis.luco.gov.bc.ca/pub/coastal/)	(None)	4 (some destruction of structural, epibenthic species)	3 (loss of habitat for other species, bycatch)	4.9	Average of "severity and duration of impact habitat" and "species" rating	√		√	
*Recreational fisheries**										Coleman et al. 2004, Cooke and Cowx 2006
Recreational squid	Province of BC (ftp://ftp.gis.luco.gov.bc.ca/pub/coastal/)	(None)	3	2 (bycatch)	3.6	Less than commercial fishing because presumably a lower volume is extracted	√		√	

Type of data	Source of data	Stressor beyond location	Severity and duration of impact (0=least, 10=greatest)		Impact= direct + 0.3 * indirect	Impact value (Jamieson and Levings (2001))	Category of impact (physical, chemical, biological)			References
Recreational scallop	Province of BC (ftp://ftp.gis.luco.gov.bc.ca/pub/coastal/)	(None)	3	2 (bycatch)	3.6	Less than commercial fishing because presumably a lower volume is extracted	√		√	
Recreational prawn	Province of BC (ftp://ftp.gis.luco.gov.bc.ca/pub/coastal/)	(None)	3	2 (bycatch)	3.6	Less than commercial fishing because presumably a lower volume is extracted	√		√	
Recreational groundfish	Province of BC (ftp://ftp.gis.luco.gov.bc.ca/pub/coastal/)	(None)	3	2 (bycatch)	3.6	Less than commercial fishing because presumably a lower volume is extracted	√		√	
Recreational crab fishing areas	Province of BC (ftp://ftp.gis.luco.gov.bc.ca/pub/coastal/)	(None)	3	2 (bycatch)	3.6	Less than commercial fishing because presumably a lower volume is extracted	√		√	
Recreational fish (not species-specific)	Province of BC (ftp://ftp.gis.luco.gov.bc.ca/pub/coastal/)	(None)	3	2 (bycatch)	3.6	Less than commercial fishing because presumably a lower volume is extracted	√		√	

Type of data	Source of data	Stressor beyond location	Severity and duration of impact (0=least, 10=greatest)		Impact= direct + 0.3 * indirect	Impact value (Jamieson and Levings (2001))	Category of impact (physical, chemical, biological)			References
Transportation and infrastructure										
Shipping lane*	Coast Guard	Medium	5 (pollution, noise)	1	5.3	Higher impact than commercial fishing because of the concentration of ships using the same route, but causes less habitat destruction than bottom trawling or permanent structures such as ferry docks	√	√		Moore and Clarke 2002, Hampton et al. 2003
Cruise ship routes*	Oil and gas commission website	Medium	5 (noise, discharge of effluent)	2	5.6	Same as shipping lane, but higher indirect impact due to black and greywater discharges	√	√		United States General Accounting Office 2000, Commoy et al. 2005
Anchorages	Province of BC (ftp://ftp.gis.luco.gov.bc.ca/pub/coastal/)	Low	1	2 (fishing and litter)	1.6	Severity and duration of impact habitat rating	√	√		Backhurst and Cole 2000, Jamieson and Levings 2001, Milazzo et al. 2004

Type of data	Source of data	Stressor beyond location	Severity and duration of impact (0=least, 10=greatest)		Impact= direct + 0.3 * indirect	Impact value (Jamieson and Levings (2001))	Category of impact (physical, chemical, biological)			References
Boat launches*	Province of BC (ftp://ftp.gis.luco.gov.bc.ca/pub/coastal/)	Medium-low	1	3 (noise, litter, spills)	1.9	Considered same as anchorages, but higher indirect impact because of the permanent structures	√	√		Turner et al. 1997
Disposal sites	Province of BC (ftp://ftp.gis.luco.gov.bc.ca/pub/coastal/)	High	10	4 (possible toxins, leaching)	11.2	Severity and duration of impact habitat rating		√		Jamieson and Levings 2001, Savage 2005
Moorage*	Province of BC (ftp://ftp.gis.luco.gov.bc.ca/pub/coastal/)	Low	1	2 (fishing and litter)	1.6	Considered same as anchorages	√	√		Nightingale and Simenstad 2001
Ferry dock*	TRIM – Province of BC	Medium	8 (destruction of habitat, alteration of currents)	4 (litter, toxins)	9.2	Less severe (-2 points) than "loading areas and terminals" as only people are loaded	√	√		Nightingale and Simenstad 2001, Stevens and Ekermo 2003
Marina*	TRIM – Province of BC	Medium	8 (destruction of habitat)	4 (litter, toxins)	9.2	Less severe (-2 points) than "loading areas and terminals" as only people are loaded	√	√		Turner et al. 1997, Nightingale and Simenstad 2001
Ferry route*	TRIM – Province of BC	Medium-low	5 (noise)	1	5.3	Same as shipping lane	√	√		Stevens and Ekermo 2003
Docks*	TRIM – Province of BC	Low	1	3 (litter, fishing)	1.9	Considered same as anchorages, but higher indirect impact because of the permanent structures	√	√		Wendt et al. 1996, Macfarlane et al. 2000, Nightingale and Simenstad 2001

Type of data	Source of data	Stressor beyond location	Severity and duration of impact (0=least, 10=greatest)		Impact= direct + 0.3 * indirect	Impact value (Jamieson and Levings (2001))	Category of impact (physical, chemical, biological)			References
Pier*	TRIM – Province of BC	Low	1	3 (litter, fishing, noise)	1.9	Considered same as anchorages, but higher indirect impact because of the permanent structures	√	√		Macfarlane et al. 2000, Nightingale and Simenstad 2001
Terrestrial uses										
Terrestrial Mining	TRIM – Province of BC	Medium-Low	0	3 (waste and chemical dispersion, sediment plume)	0.9	Some indirect impact due to pollution, metals, acid rock drainage		√		Hines et al. 2000, Jamieson and Levings 2001, Levings et al. 2004
Built-up area*	TRIM – Province of BC	Medium-high	2 (increased sedimentation discharge, disturbance of natural vegetation)	2 (disturbance of estuarine functions, removal of detrital sources)	2.6	Some direct impact due to seaside structures, indirect impact due to urban run-off	√	√		Nightingale and Simenstad 2001, Kennish 2002, Bolton et al. 2004
Industry*	Province of BC (ftp://ftp.gis.luco.gov.bc.ca/pub/coastal/)	High	8	7 (possible toxins)	10.1	Based on an average of "industrial outfall", "groins and breakwaters", "logbooming", and "dredging"	√	√		Colodey and Wells 1992, Khan 1997, Roberts et al. 1998, Bolton et al. 2004
Lighthouse*	TRIM – Province of BC	Low	0	1	0.3	Some indirect impact due to structure and human presence	√			Stamski, 2005

Given that marine activities do not affect the marine environment equally, a measure of the impact of marine activities at the location of occurrence was incorporated. Ranking impacts can be contentious, as conflicting evidence can lead to differing interpretations of relative impact. Therefore an existing classification method was sought, and one which ranks 27 out of the 39 uses mapped in this study was applied (Jamieson & Levings 2001). This scheme uses a qualitative ranking of the direct and indirect impact of human activities for British Columbia (high impact = 10, least impact = 0), developed through focus groups of regional experts representing habitat managers and field biologists (Jamieson & Levings 2001). Only the "severity and duration of impact" values are applied; the "extent of impact" category included in Jamieson and Levings (2001) is superfluous as the geographic extent of activities are included in this analysis through spatial data. The median value of 0.3 for weighting indirect impacts is used, for a maximum possible impact value of 13. The impact value is calculated using this formula:

$$\text{Impact} = \text{direct} + (0.3 * \text{indirect})$$

Table 3.1 contains an explanation of all the impact values used.

For most marine activities, little is known about the geographic extent of the impact beyond the location of the activity. A table was compiled referencing the measured impacts of activities (Table 3.2). Observed impacts vary an order of magnitude in many cases, and therefore assumptions about the extent of the impact were applied to each data layer. These are termed stressors, as the impact on species and habitats is inferred. First, the assumption was that the stressors resulting from activities are localized, and a uniform 1 km buffer around point and line data is applied (Figure 3.1). Second, a medium extent of stressors beyond the site of occurrence is assumed, with buffers out to a maximum of 5 km. Third, a larger extent of stressors is assumed, and buffers are assigned out to a maximum of 25 km. To apply the last two assumptions, a qualitative ranking of high to low to rate the extent of stressors beyond the site of occurrence is designated based on a review of the literature (Table 3.2). Diminishing buffers were then assigned based on the assumptions above, using the multiple buffer option with 1 km increments for the medium buffers, and 5 km increments for the large buffer assumption (Figure 3.2). Each marine use and its associated buffers was mapped on a raster (1 km^2 grid), and then the impact of each activity was multiplied by its relative weighting (stressors) and the appropriate buffer distance (Figure 3.2). Where activities overlap in a grid cell, the values were added. Because of the variability in the fishing polygons, buffers were not used for fishing areas. The bottom trawling data were summarized at a coarse scale in a 10 km by 10 km grid, such that buffering these data could exaggerate the impact of this activity. The polygon data for the other fisheries were of unknown completeness and accuracy, and so were also handled without buffers.

Table 3.2. Impact beyond location of occurrence from the literature.

Impact beyond site	Type of data	Type of impact	Maximum distance of observed impact	Location of study	References
High	Finfish aquaculture	Transmittance of furunculosis	24 km	Puget Sound	Quoted in EVS Environment Consultants, 2000
		Sea lice infections exceeded ambient levels	30 km	British Columbia	Krkošek et al. 2005
		Second generation of lice that re-infected juvenile salmon exceeded ambient levels	75 km	British Columbia	Krkošek et al. 2005
		Escaped Atlantic Salmon	100s of kms	British Columbia	Naylor et al., 2001
		Dead zone created by accumulated organic matter	100 to 500 feet	British Columbia	Quoted in Naylor et al. 2003
	Disposal sites	Sewage-derived nitrogen traced to 24 km from outfall; sewage influence most pronounced within 10 km	10-24 km	Baltic Sea	Savage 2005
	Industry	Trace metal contaminants found ~50 km distant from Vancouver harbor	~50 km	British Columbia	Bolton et al. 2004
		Structural changes in benthic communities along a presumed pollution gradient	~20km	British Columbia	Je et al. 2004
		Traces of bark, fiber and wood chips observed 12 km upcurrent from a pulp and paper mill	12 km	Newfoundland	Khan 1997
Medium-high	Built-up area	Structural changes in benthic communities along a presumed pollution gradient	~20km	British Columbia	Je et al. 2004
Medium	Shipping lane	Responses of feeding humpback whales to vessels	2-4 km	British Columbia and Alaska	Richardson and Malme 1995
		Illegal dumping of oily wastes	80 km	California	Hampton et al. 2003
		Boat noise could impair communication between killer whales over a range of 1 – 14 km	1-14 km	Washington and British Columbia	Foote et al. 2004
	Cruise ship routes	Volume of greywater plume with detectable levels of tracer dye	6 – 45 billion litres	Florida	United States Environmental Protection Agency 2002

Impact beyond site	**Type of data**	**Type of impact**	**Maximum distance of observed impact**	**Location of study**	**References**
Medium		Boat noise could impair communication between killer whales over a range of 1 – 14 km	1-14 km	Washington and British Columbia	Foote et al. 2004
	Marina	Dredging for marina development and vessel navigation, water quality issues creating conditions for dinoflagellate blooms	At least extent of the marina and vessel channels	Washington	Nightingale and Simenstad 2001
		Sedimentation and erosion due to ship traffic	Erosion areas represent 56% of the total 1,149,000 m^2 mapped	Sweden	Stevens and Ekermo 2003
	Ferry dock	Increased heavy metal contamination, differences in biological communities and settlement rates	1.4 km	New Zealand	Turner et al. 1997
Medium -low	Ferry route	Boat noise could impair communication between killer whales over a range of 1 – 14 km	1-14 km	Washington and British Columbia	Foote et al. 2004
		During construction, pile driving noise would be heard by salmonids within a radius of at least 600 m from the noise	600 m	Washington	Nightingale and Simenstad 2001
	Boat launches	Increased heavy metal contamination, differences in biological communities and settlement rates	1.4km	New Zealand	Turner et al. 1997
	Terrestrial Mining	Increased levels of mercury 40km from abandoned mercury mine	40 km	Gulf of Trieste, Slovenia and Italy	Hines et al. 2000
		Acid mine drainage had a deleterious effect on mussels at least 2.1 km north and 1.7 km south of the mine	At least 2.1 km	British Columbia	Grout and Levings 2001
Low	Shellfish aquaculture	(Very little quantified information on impacts beyond shellfish farms is available, but see (Broekhuizen *et al.*, 2002)p. 7 for an overview)			Broekhuizen et al. 2002
		Introduction of exotic species	100s of kms	British Columbia	Naylor et al. 2001
	Moorage	During construction, pile driving noise would be heard by salmonids within a radius of at least 600 m from the noise	600 m	Washington	Nightingale and Simenstad, 2001, Stevens and Ekermo 2003
	Anchorages	Anchor damage to benthos	Scale of whole embayments	New Zealand	Backhurst and Cole 2000

Impact beyond site	Type of data	Type of impact	Maximum distance of observed impact	Location of study	References
Low	Docks	Shading from the average dock adversely affects 87 m^2 marsh grass	87 m^2	South Carolina	Sanger and Holland 2002
		Light levels reduced 2-4 orders of magnitude	2400 feet	Seattle	Macfarlane et al. 2000, Nightingale and Simenstad 2001
	Pier	During construction, pile driving noise would be heard by salmonids within a radius of at least 600 m from the noise	600 m	Washington	Nightingale and Simenstad 2001
	Lighthouse	Physical habitat alteration due to structures and erosion control	Unknown	California	Stamski 2005

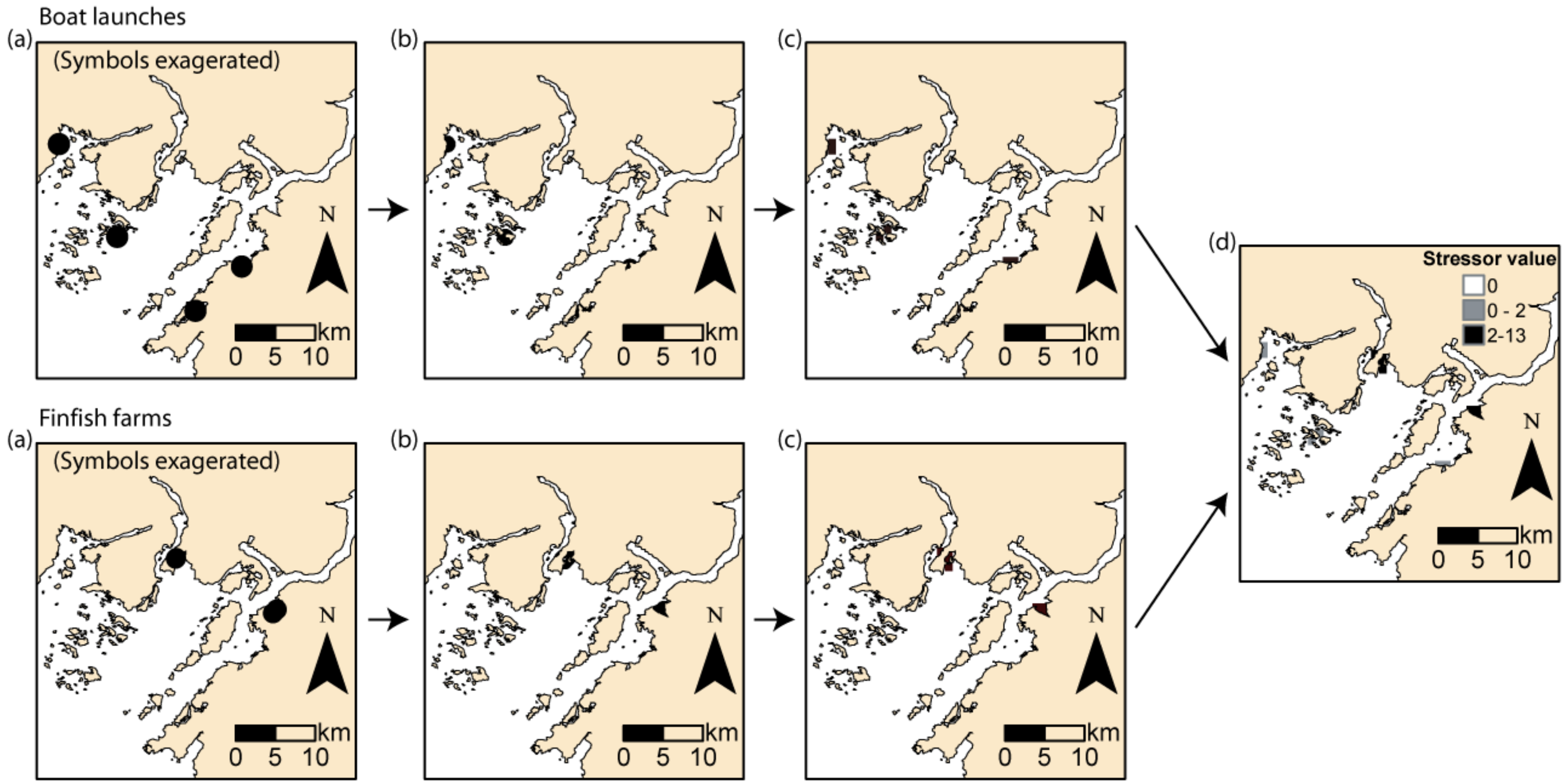

Figure 3.1. Schematic of the methodology used to generate the small buffer (1 km) data layers and analysis, highlighting two activities in one part of British Columbia. (Additional activities take place in this particular part of BC). (a) the activities are mapped. (b) 1 km buffers are added to mapped activities (see Table 3.1). (c) the maps are converted to a 1 km2 raster grid, assigning the stressor value associated with each activity (Table 3.1). (d) the stressor values for all layers are added.

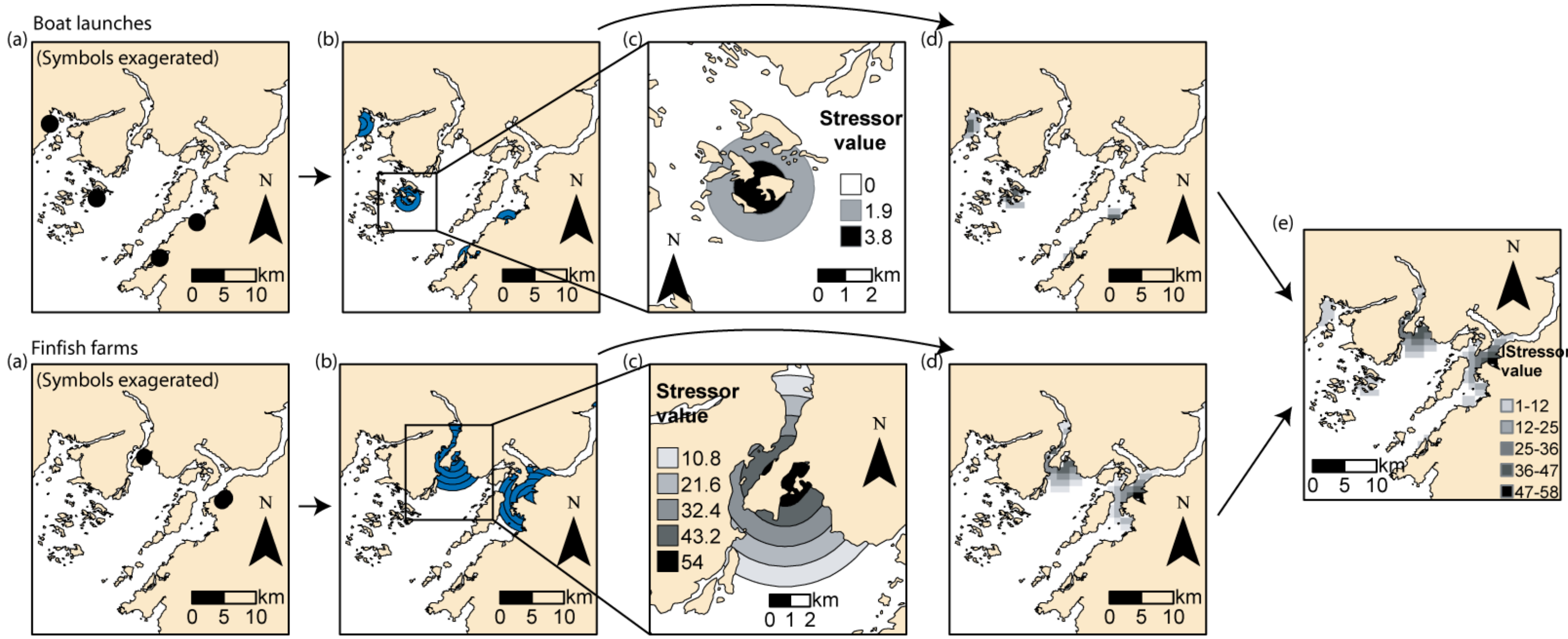

Figure 3.2. Schematic of the methodology used to generate the medium buffer (up to 5 km) data layers and analysis. (a) the activities are mapped. (b) Buffers are added in 1 km increments (up to 5 buffers) based on the assigned relative stressor beyond the location of occurrence (as outlined in Table 3.1). Activities with a high rating are given 5 one km buffers, activities with a medium-high rating are assigned 4 buffers, etc. (c) The stressor values are assigned based on the impact value associated with each activity (Table 3.1). To assign the stressor value, the innermost buffer multiplies the impact value by the number of rings (n). The next buffer is calculated as the impact value times the number of buffers minus one, the next buffer is the impact value times the number of buffers minus two, etc. This created decreasing buffer values, with the innermost buffer having the highest stressor value. (d) The maps are converted to a 1 km2 raster grid. (e) All the layers are added. The same methodology was used for the large buffers (up to 25 km), using 5 km buffers instead of the one kilometer buffers described above.

A visual report of the pattern of use can help identify the stressors of human activities in the ocean. Three composite maps overlaying all activities were created, one for each of the above assumptions: a small buffer with 1 km around line and point data, a medium buffer with maximum buffer distance of 5 km for activities with a high rating for the extent of stressors beyond the location of occurrence, and a large buffer with a maximum buffer distance of 25 km for activities with a high rating for the extent of stressors beyond the location of occurrence.

Two metrics were applied to gauge the extent of stressors of different types of activities. First, the stressor value of each occurrence of an activity in a raster cell was added, then averaged over all available raster cells. This gives an indication of the highest average stressors. Second, the total number of raster cells where an activity occurs was tabulated. This provides a measure of the extent of activities. Both of these metrics were calculated for each of the three buffer assumptions.

The marine areas currently covered by protected area designations were calculated to gauge existing protection. The following designations are included in the calculation: *Oceans Act* Marine Protected Areas, National Parks, Rockfish Conservation Areas, provincial Protected Areas, Parks, Marine Parks, and Ecological Reserves. The BC marine ecological classification system was used to divide the marine area into the offshore and continental shelf and slope regions (Zacharias et al. 1998; Zacharias & Howes 1998). The Inner Pacific Shelf, Outer Pacific Shelf, and Georgia Basin ecoregions comprise the continental shelf and slope. The Subarctic and Transition Pacific ecoregions comprise the offshore region.

Results

The continental shelf and slope of BC is being used extensively by humans (Table 3.3). Examining the map showing the number of overlapping activities with small buffers (1 km), 83 percent of the continental shelf and slope is affected by stressors from human activities (Figure 3.3). Under this buffer assumption, fishing activities appear most prominently. With buffers up to five kilometres, the resulting map does not appear very different (Figure 3.4), with 85 percent of the continental shelf and slope affected by stressors. Once buffers were extended to 25 km, however, the number of activities that overlapped increased substantially in inshore coastal areas, 15 percent of the area used having more than nine overlapping activities (Figure 3.5). Under this assumption, 98 percent of the continental shelf and slope lay in areas with stressors from human activities.

Table 3.3: Area affected by anthropogenic stressors

Buffer assumptions	Area of EEZ affected by anthropogenic stressors (ha)	Percent of EEZ affected by anthropogenic stressors	Area of continental shelf and slop affected by anthropogenic stressors (ha)	Percent of continental shelf and slope affected by anthropogenic stressors
Small (0-1 km) buffer	12,596,412	27.78%	11,157,972	83.30%
Medium (0-5 km) buffer	13,371,569	29.49%	11,431,612	85.34%
Large (0-25 km) buffer	14,598,777	32.20%	13,094,191	97.75%

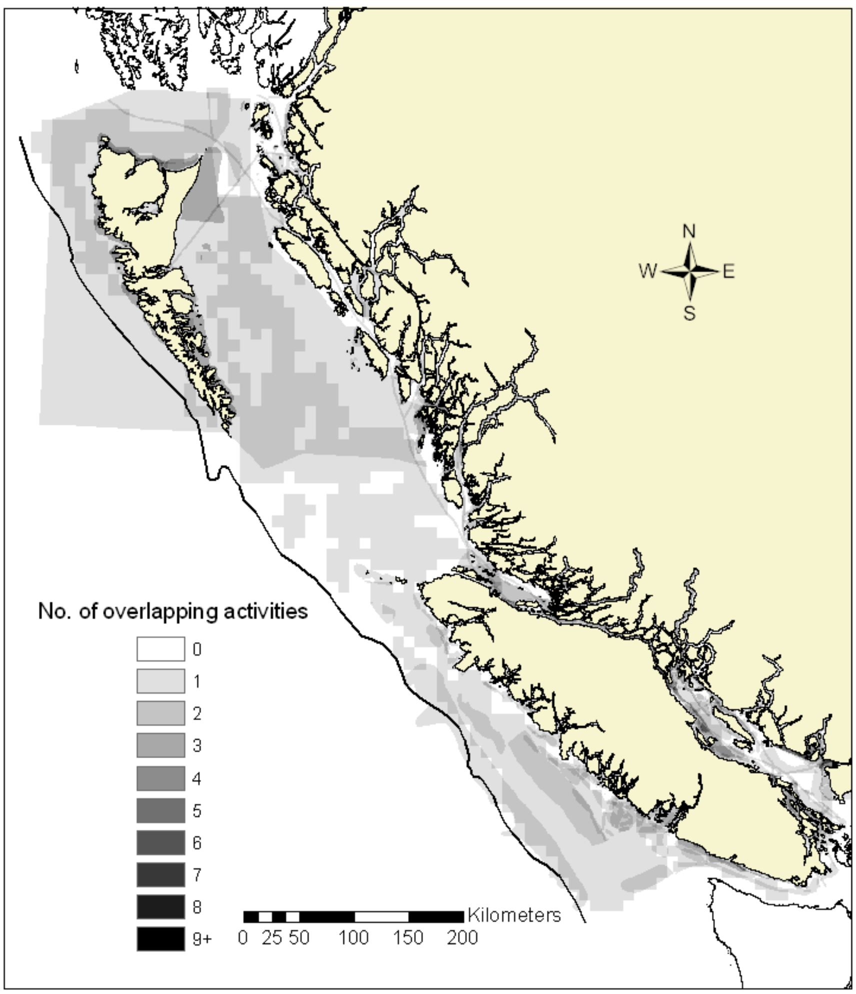

Figure 3.3. Number of overlapping activities, applying the small buffer (1 km buffer) assumption. The number of activities in each 1 km^2 grid cell is shown. The area inshore of the solid line is the continental slope and shelf.

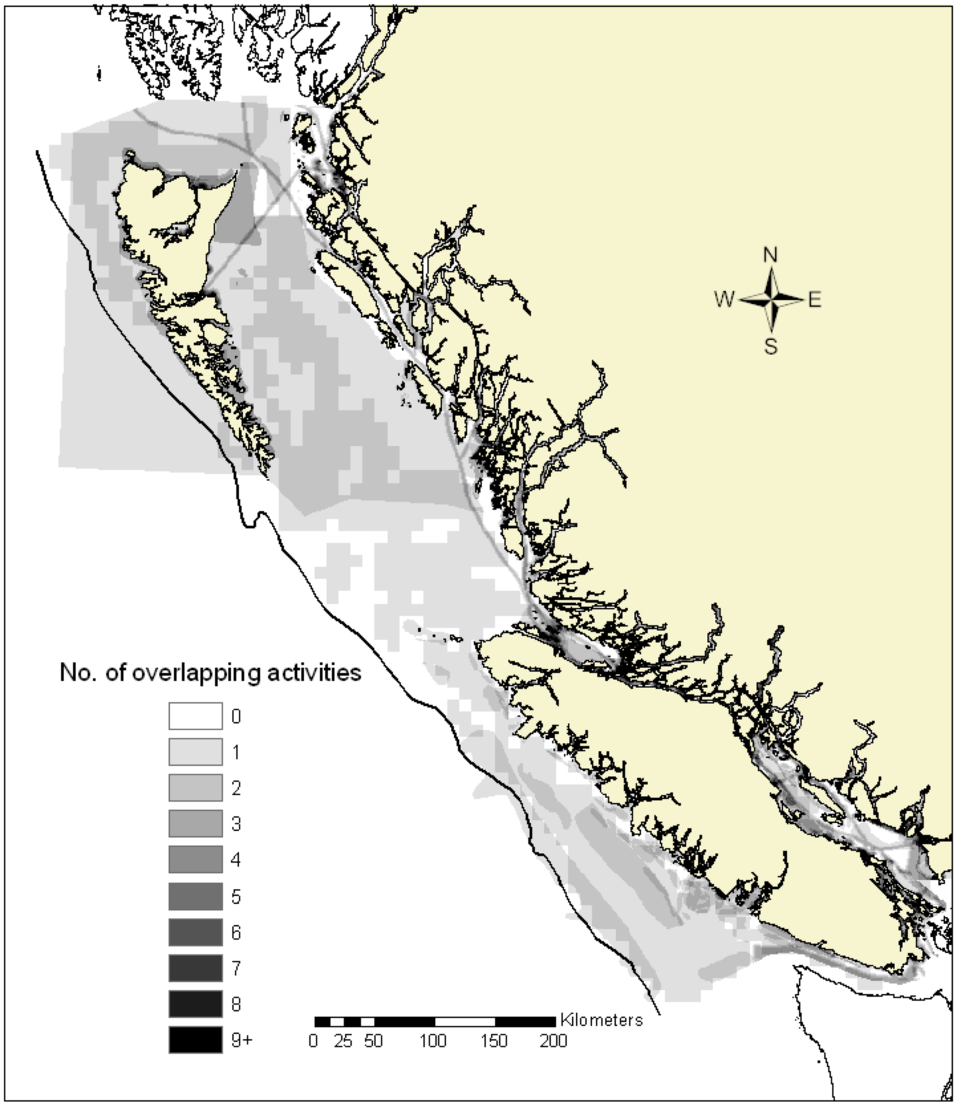

Figure 3.4. Number of overlapping activities, applying the medium buffer (up to 5 km buffer) assumption. The number of activities in each 1 km^2 grid cell is shown. The area inshore of the solid line is the continental slope and shelf.

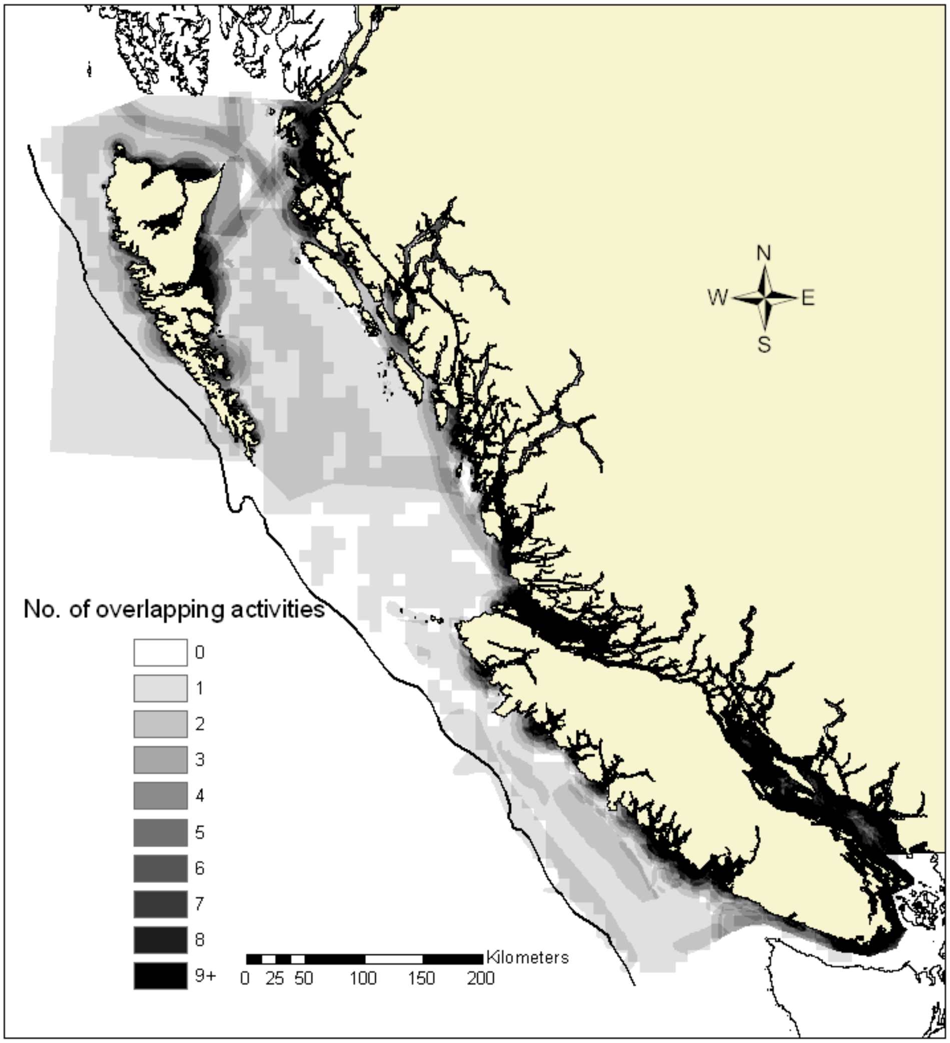

Figure 3.5. Number of overlapping activities, applying the large buffer (up to 25 km buffer) assumption. The number of activities in each 1 km^2 grid cell is shown. The area inshore of the solid line is the continental slope and shelf.

Accounting for the impact of activities by mapping predicted stressors, much of the continental shelf and slope region appears more affected than depicted by overlapping activities (Figure 3.6, 3.7, and 3.8 for the small, medium, and large buffer assumptions). As with the small and medium buffer maps showing the number of overlapping activities (Figure 3.3 and 3.4), the small and medium buffer maps depicting stressors are similar to each other (Figure 3.6 and 3.7). The large buffer map (Figure 3.8) highlights most coastal areas as having high relative values of stressors.

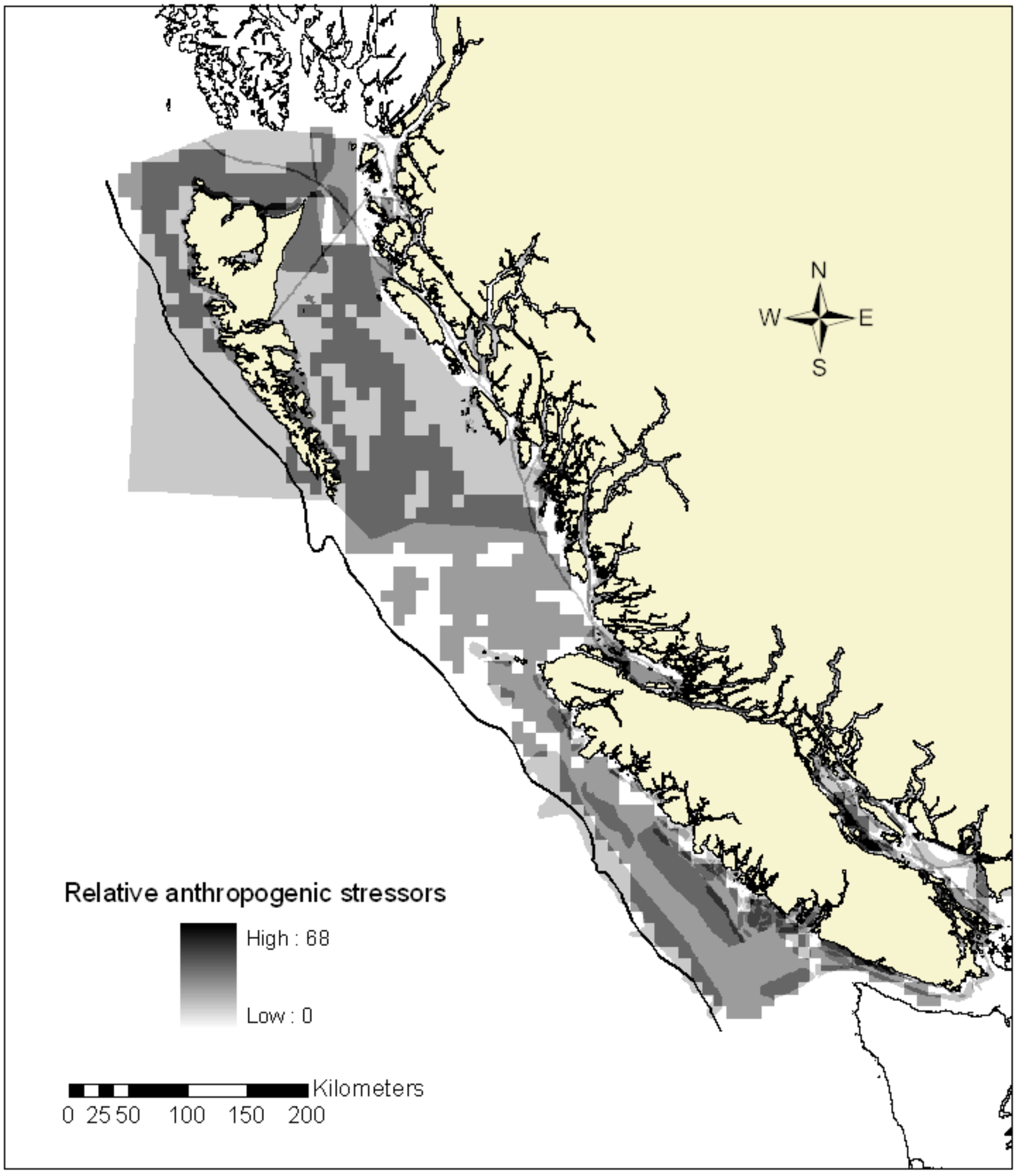

Figure 3.6. Stressors resulting from human activities, applying the small buffer assumption (1 km). The area inshore of the solid line is the continental slope and shelf.

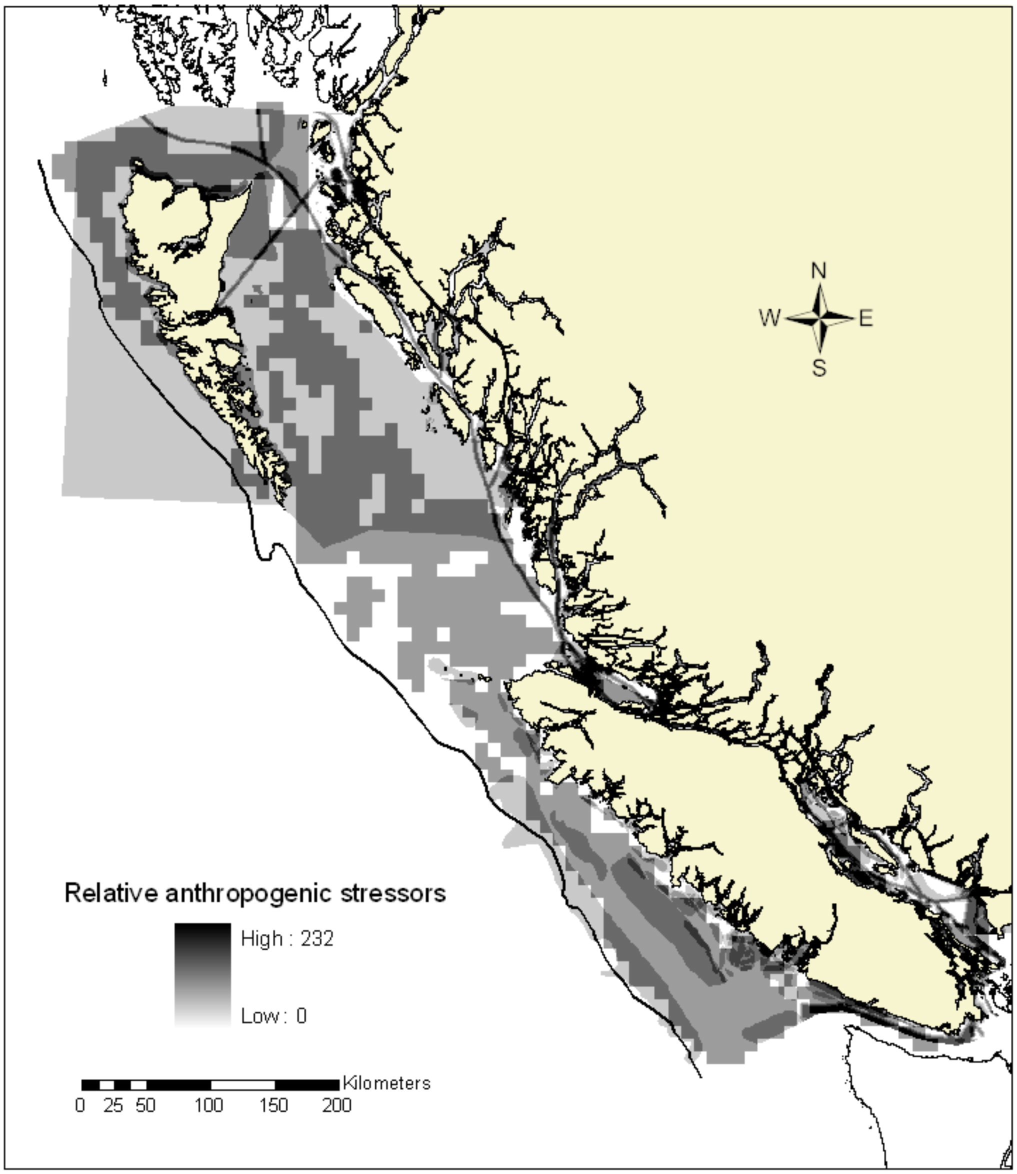

Figure 3.7. Stressors resulting from human activities, applying the medium buffer assumption (up to 5 km). The area inshore of the solid line is the continental slope and shelf.

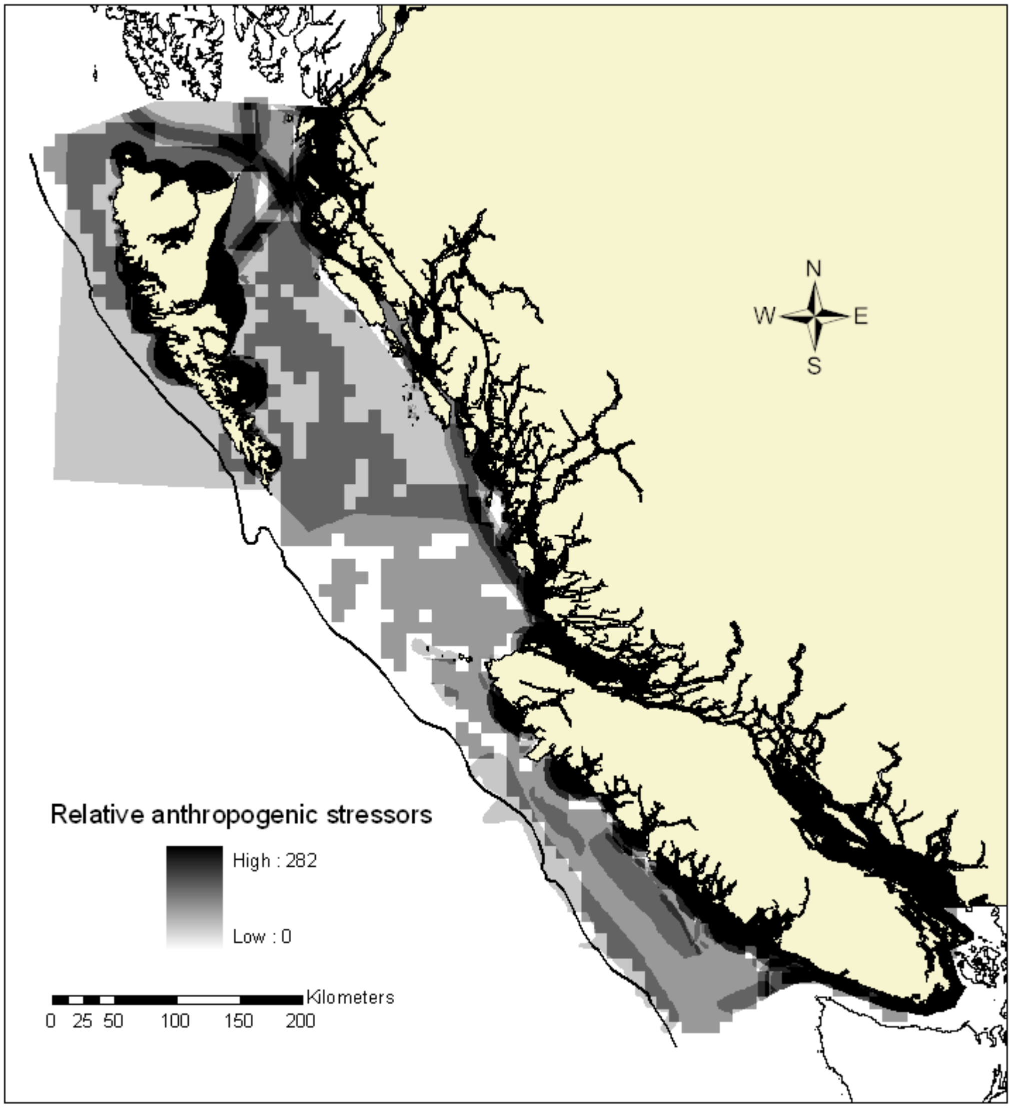

Figure 3.8. Stressors resulting from human activities, applying the large buffer assumption (up to 25 km). The area inshore of the solid line is the continental slope and shelf.

The marine activity with the highest stressor value when averaged over all raster cells is commercial bottom trawling for both the small buffer assumption (Figure 3.9), and the medium buffer assumption (Figure 3.10). Assuming stressors extend up to 25 km beyond the sites of occurrence, however, indicated that industry exceeded bottom trawling as the activity with the highest average raster cell value (Figure 3.11). The industry data were used as categorized by the Province of British Columbia (Table 3.1), and includes logging operations (*e.g.,* log booms, logging camps, log dumps), pulp and paper mills, industrial yards, oil tanks, conveyors, buildings, fish processing facilities, and ship yards.

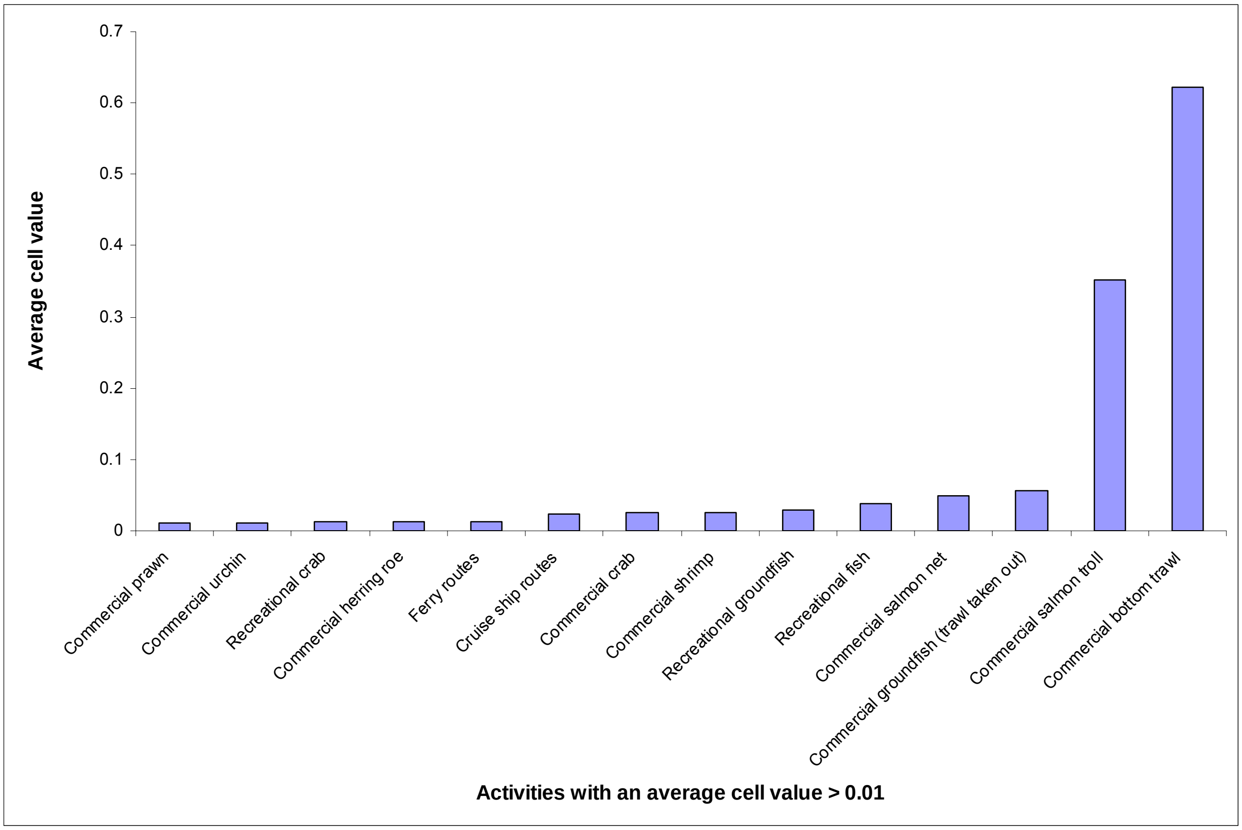

Figure 3.9. Stressors resulting from marine activities by average raster cell value, using the small buffer assumption (1 km).

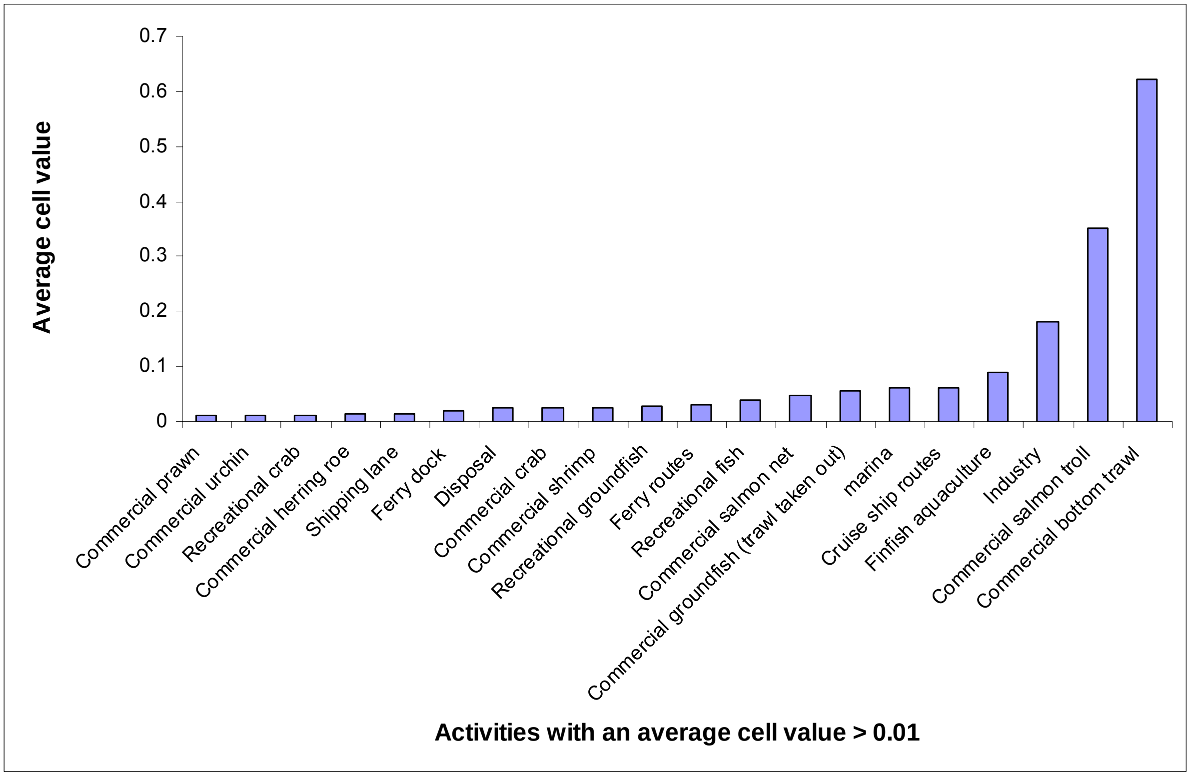

Figure 3.10. Stressors resulting from marine activities by average raster cell value, using the medium buffer assumption (up to 5 km).

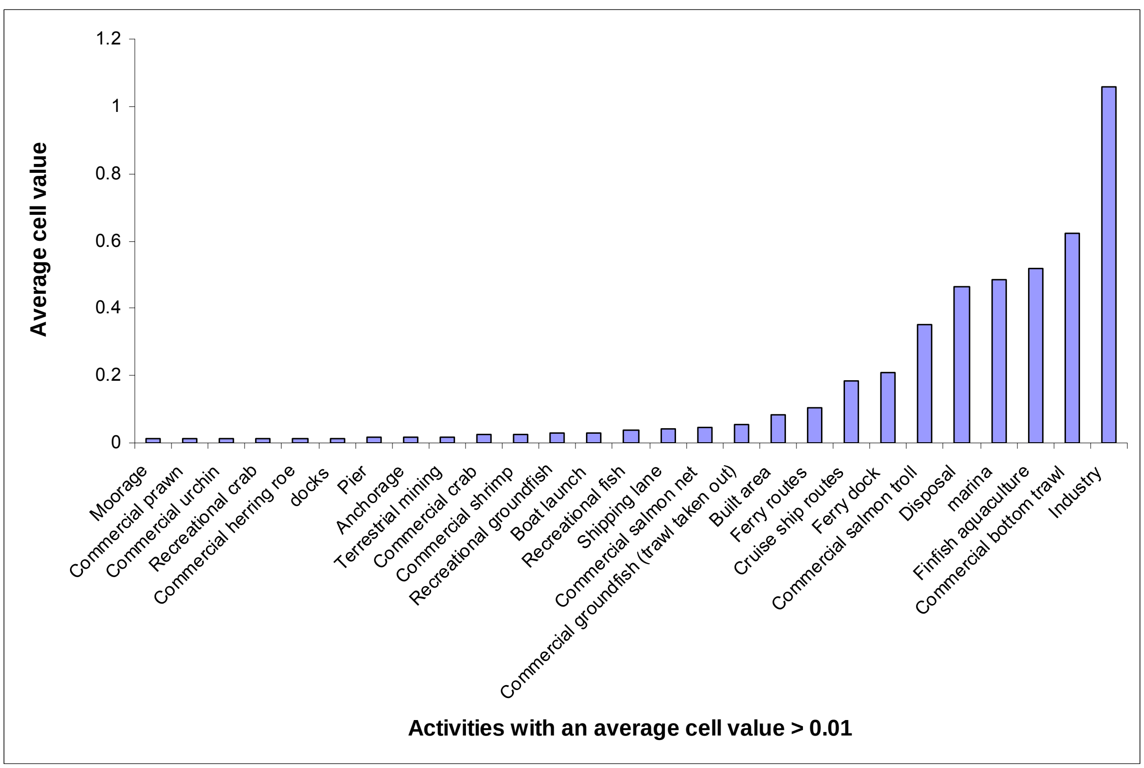

Figure 3.11. Stressors resulting from marine activities by average raster cell value, using the large buffer assumption (up to 25 km).

The category of marine activity in BC with the largest spatial extent was commercial fishing under both the small and medium buffer assumptions given stressors (Figure 3.12) and the area used (Figure 3.13). Accounting for stressors under the large buffer assumption, the transportation and infrastructure category and terrestrial use category exceed the stressors resulting from commercial fishing (Figure 3.12). Recreational fishing has the lowest stressor value under the medium and large buffer assumptions.

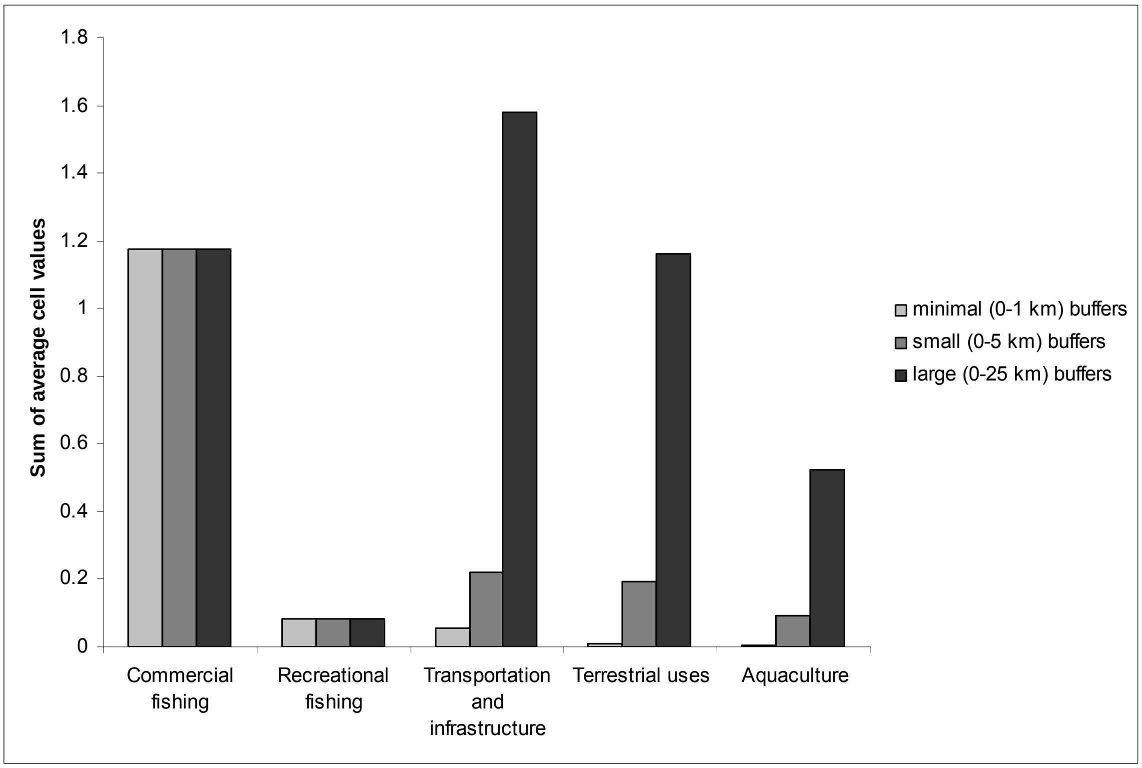

Figure 3.12. Stressors resulting from marine activities by category.

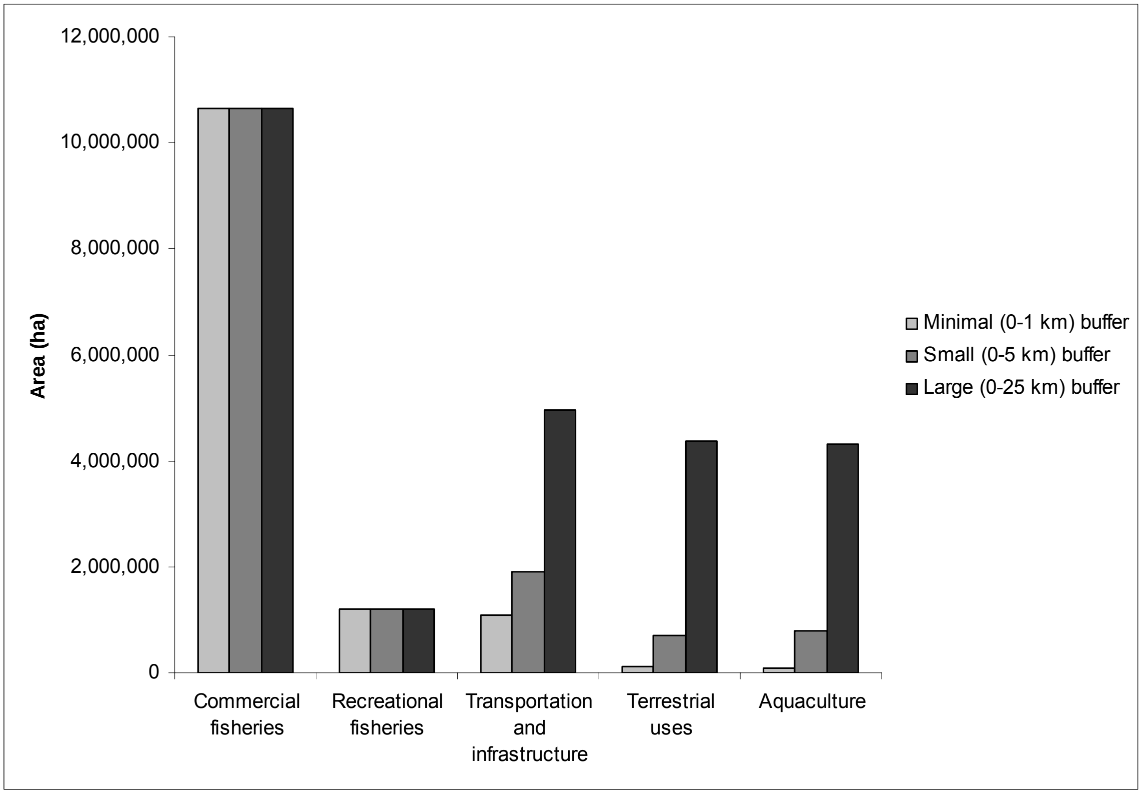

Figure 3.13. Area impacted by categories of marine activities.

Very little of BC's marine environment is currently protected (Table 3.4). BC's EEZ has 1.47 percent of waters protected, while the continental shelf and slope has a proportionally higher protection at 4.69 percent.

Table 3.4. Marine protection in British Columbia

	Protection provided	Hectares	% of EEZ	% of continental shelf
Marine Protected Areas	Variable	93,812	0.21%	0.00%
Rockfish Conservation Areas	Recreational fishing activities allowed are hand picking or diving for invertebrates, crab by trap, prawn by trap, smelt by gillnet. Commercial fishing activities allowed are invertebrates by hand picking or dive, crab by trap, prawn by trap, scallops by trawl, salmon by seine or gillnet, herring by gillnet, seine and spawn-on-kelp, sardine by gillnet, seine, and trap, smelt by gillnet, euphausiid (krill) by mid-water trawl, opal squid by seine, and groundfish by mid-water trawl (Fisheries and Oceans Canada, 2005)	393,152	0.87%	2.93%
Provincial Protected Areas, Provincial Parks, Marine Parks, National Parks, Ecological Reserves	None to variable	237,635	0.52%	1.77%
Total*		666,078	1.47%	4.69%

Discussion

Intensity of use

The continental shelf and slope of British Columbia is used intensively by humans. Even under the most conservative assumption about the extent of stressors resulting from human activities, a greater proportion of BC's continental shelf and slope is affected than the adjacent coastal terrestrial area, where 45.9 percent of the coast and mountains ecoprovince is considered intact (Government of Canada & Province of British Columbia 1998). The data used in this analysis indicate that deep ocean beyond the continental shelf and slope remains little used, yet globally an increase in deep-sea fishing has been observed (Roberts 2002). With such extensive use of the continental shelf and slope, very little if any of BC's marine environment is untouched.

Bottom trawling has been documented repeatedly as a destructive form of fishing (Ardron 2005; Collie et al. 2000; Fossa et al. 2002; Watling & Norse 1998). In this analysis, bottom trawling has the highest average stressor value. However, this may be influenced by how this particular data layer is summarized. Groundfish bottom trawling areas from 1996 to 2005 were obtained as documented in logbooks, summarized in 10 km by 10 km grid cells from the federal department Fisheries and Oceans Canada (DFO). Each such cell shows groundfish trawling activity when three or more distinct vessels trawled in that cell. Thus areas that do not show trawling may be trawled by one or two vessels, and it should not be inferred that all seabed within the trawled cells has been towed. More accurate bottom trawling data that do not exclude data would be very valuable.

Recreational fishing is likely much more extensive than available data show. Recreational fishing is a popular activity in BC. For example, in 2001 55% of the reported catch of chinook salmon was caught in the tidal recreational fishery (FAO 2001). A recent review of global recreational and commercial fisheries suggests that the two types of fisheries may have similar impacts (Cooke & Cowx 2006). Yet government creel surveys are not designed to collect spatial data on recreational fishing (Fisheries and Oceans Canada 2006). Incorporating questions about the spatial distribution of fishing into creel surveys would greatly improve the information on recreational fishing. It may therefore be dangerous to dismiss recreational fishing as an insignificant source of impact based on our results.

Protection of the marine environment

Very little of the marine environment in BC is protected even though marine protected areas have been a subject of Canadian government policy development since the 1960s (Government of Canada & Province of British Columbia 1998). Marine protected areas can eliminate or reduce the direct impact of human activities on the ocean. British Columbia has several protected area designations that have previously been considered as marine protected areas (Zacharias & Howes 1998). Fisheries and Oceans Canada can designate Marine Protected Areas under the *Oceans Act*, where the level of protection varies. National Parks, designated under the *National Parks Act* may include a marine component, although the level of protection also varies. Parks Canada can designate National Marine Conservation Areas under the *National Marine Conservation Areas Act*, which are meant to be zoned to allow various levels of use and must include a no-take component. Environment Canada can establish marine wildlife areas, aimed primarily at protecting seabird foraging areas. Fisheries and Oceans Canada has been designating Rockfish Conservation Areas aimed to reduce the decline of inshore rockfish species. Activities unlikely to harm rockfish are permitted. Provincial Protected Areas, Marine Parks, and Ecological Reserves in the marine environment similarly vary in the level of protection.

The area protected has changed little since a previous analysis in 1997 (Government of British Columbia 2006; Zacharias & Howes 1998). In total, all marine areas designated for conservation or protection purposes combined cover 1.5% of British Columbia's marine environment (Table 3.4). However, most of these areas were designated for terrestrial purposes, with boundaries extending into the marine environment without necessarily providing comprehensive marine protection (Jamieson & Levings 2001). Rockfish Conservation Areas currently cover 0.9% of British Columbia's EEZ. These areas were set up to protect inshore rockfish under the inshore rockfish conservation strategy (Fisheries and Oceans Canada 2002), with the eventual goal of protecting about 20 percent of inshore rockfish habitat. One Marine Protected Area (Endeavour Hydrothermal Vents) covers 0.2% of BC's marine area. No areas currently exist under the National Marine Conservation Area or Marine Wildlife Area designations in British Columbia. Given that a very large percentage of BC's marine environment is already exposed to human activity, the need to provide more protection is urgent. The results from this analysis could be integrated as a cost layer into reserve selection tools (for such an approach in BC see Ardron 2003) to identify suitable conservation areas that are predicted to be relatively less impacted.

Canada's Oceans Strategy, led by Fisheries and Oceans Canada, is meant to provide an integrated approach to ocean management, coordinate policies and programs across governments, and generate a shift towards an ecosystem approach (Government of Canada 2002a). While the strategy has been in place since 2002, progress in achieving it has been very slow (Jessen & Ban 2003; Auditor General of Canada 2005). In British Columbia, most aboriginal groups have not signed treaties, and all have a right to fish for food, social and ceremonial purposes. Successful implementation will require meaningful involvement of aboriginal people (Government of Canada 2002b).

Data issues

The marine uses considered for this analysis were limited by data availability (Table 3.5). The resulting maps should therefore be considered a preliminary and conservative estimate of human use of the ocean in BC. Many human activities and influences were not included in our analysis because data were unavailable (Table 3.5). Therefore, the emphasis in this analysis is on the general patterns of use, not the precise locations of activities and influences. As such, this static assessment of anthropogenic stressors is not sufficient to comprise a complete picture of anthropogenic stressors and their impacts.

Table 3.5: Incomplete and missing data

Type of data	Status of data	Comments
Additional recreational fisheries	Missing	Limited spatial information on recreational fishing areas is currently collected through creel surveys, and data on some targeted recreational fisheries, such as salmon, is missing from the spatial data files
Additional terrestrial: clear cuts, agriculture	Proprietary	This kind of information could also be incorporated into a non-point source pollution database
Commercial marine tourism: wildlife viewing, sports fishing, diving	Missing	Some recreational fishing areas are currently included, but sports fishing lodges are not. Other commercial tourism operations should also be considered
Non-commercial marine tourism areas: pleasure boating	Missing	
Invasive species	Missing	Problem areas for invasive species would help identify marine areas under stress
Non-point source pollution	Missing	
Aboriginal fisheries	Missing/proprietary	
Shipping routes	Missing	Aside from the designated shipping lane in Juan de Fuca Strait and Strait of Georgia, we were unable to find data on routes used by shipping/tanker traffic
Climate change	Missing	Climate change, such as rising temperatures in the ocean, has a ubiquitous impact. Yet there may be areas that are seeing more changes than average.
Acidification	Missing	Acidification of the ocean is also a ubiquitous impact. We do not know whether any information exists for acidification in British Columbia
Historical impacts	Missing	
Natural disturbance regimes	Missing	
Vulnerable and sensitive habitats	Missing	
Risk of impact from activities	Missing	
Future developments and their potential impacts	Missing	

The resolution and accuracy of the data that exist for BC vary. For extractive uses, areas delineated by DFO were used. These data were collected from 1992 to 2002 from a variety of sources and compilers, including interviews with fisheries officers and managers, commercial and recreational fishermen and other records. While the metadata indicate that the accuracy of the information is good, the metadata do not provide information about the completeness of the datasets. Using DFO's logbook data for all fisheries would be preferable, as it would provide up-to-date and complete coverage of fishing areas. Unfortunately, density information was not uniformly available for all data layers, and was therefore excluded from the analysis.

The buffers used for mapping stressors are based on a limited number of studies that report on the distance beyond which activities are having a measurable impact (Table 3.2). Yet many of these studies were not designed to measure the maximum detectable distance of impacts, and therefore the distances in Table 3.2 are minimum estimates. For example, a study measuring the effects of acid mine drainage from a copper mine on blue mussels measured mussel survival 2.1 kilometres north and 1.7 kilometres south of the mine (Grout & Levings 2001). Acid mine drainage was deleterious to blue mussels at least to this distance, but it is unknown how far beyond this distance the impact may be felt. Similarly, activities can have multiple types of impacts. For example, finfish aquaculture has effects on the substrate immediately below the sea pen, serve as a vector for sea lice infestations, contribute to the introduction of exotic species through escapes, and necessitate reduction fishing to produce their feed (Dalton 2004; Krkošek et al. 2005; Milewski 2000). The geographic extent of each of these stressors will likely vary.

Ecological considerations

Little is known about the response of multi-trophic marine communities to multiple anthropogenic stressors (Petchey et al. 2004). Because of the limited understanding of such interactions, in this analysis the simplifying assumption is that activities have an additive effect. In reality, stressors can be synergistic, or cumulative, when the combined effect is larger than the additive effect of each stressor would predict (Folt et al. 1999; Vinebrooke et al. 2004). Stressors can also be antagonistic, when the impact is less than expected (Folt et al. 1999; Vinebrooke et al. 2004). How individuals or species react to multiple stressors depends on the ability of individuals or species to tolerate each stressor, termed co-tolerance (Vinebrooke et al.

2004). When positive co-tolerance is observed, ecosystem functioning will be more likely to withstand an additional stressor. Negative co-tolerance would likely result in an increased decline or loss of species with additional stressors (Vinebrooke et al. 2004). Examples of both responses have been observed in aquatic environments (Folt et al. 1999; Lotze & Milewski 2004; Scheffer et al. 2005; Vinebrooke et al. 2004). In addition, a debate exists in the ecology literature about whether more diverse ecosystems are more stable (the diversity-stability debate) (Chapin III et al. 2000; Doak et al. 1998; Ghilarov 2000; Grime 1997; Huston 1997; Loreau & Hector 2001; Loreau et al. 2001; McCann 2000; McGrady-Steed et al. 1997; Naeem 2002; Naeem & Li 1997; Schläpfer & Schmid 1999; Tilman 1999, 2000; Tilman et al. 1998). One hypothesis suggests that the stability of ecological communities is affected by the interaction strengths between predators and their prey (Bascompte et al. 2005; de Ruiter & Neutel 1995), and therefore the structure of a community will affect its response to stressors.

The frequency and magnitude of natural disturbances will influence the response of individuals, species and functional groups to anthropogenic stressors (Hughes & Connell 1999; Nyström & Folke 2001). Similarly, the history of natural and anthropogenic disturbances in any particular area will affect the response of individual, species and functional groups to additional natural or anthropogenic stressors (Lotze & Milewski 2004). Thus environments also vary in their sensitivity to particular stressors given both the habitat structure and past impacts (Zacharias & Gregr 2005). For example, a muddy substrate subject to frequent natural disturbance events such as storms that perturb the sediment will be less sensitive to trawling than an area comprised of deep sea corals. Such less physically transient habitats are generally inhabited by more opportunistic species that are better able to recover from trawling until a threshold beyond which the system enters a permanently altered state (Collie et al. 2000). Because of the lack of spatial data on natural disturbance regimes and historical human use, a limitation of this analysis is that only recent anthropogenic stressors were mapped.

Future direction

Given the confounding effects of positive and negative co-tolerance, natural disturbances, and past anthropogenic impacts, it is unknown whether or how the maps of intensity of use and anthropogenic stressors translate into ecological impacts in the ocean. Additional mapping that incorporates models of the vulnerability and sensitivity of habitats to different types of stressors (sensu Zacharias & Gregr 2005) would contribute to understanding the impacts such stressors may have on the marine environment. Including the risk of impact from activities, and the potential contribution of planned and potential developments (*e.g.,* port expansions, oil and gas development, inshore tanker traffic) would further the assessment. Plans for undertaking such additional mapping work are under way. A future step in verifying the analysis would be to ground-truth the results and determine whether a correlation exists between impacts and the areas mapped as having a high level of stressors.

Conclusion

The purpose of this paper was to depict the intensity of use, and evaluate the sum of stressors resulting from human activities in British Columbia, Canada. Results show that the continental shelf and slope of British Columbia is extensively and intensively used by humans, yet very little protection is offered to the marine environment. The analysis provides a preliminary and conservative look at the patterns and intensity of use and resulting stressors given spatial data currently available. The resulting maps can be used as a baseline of human activities for comparison with future analyses. The results may also assist in the development of integrated management plans by providing a spatial representation of the location of activities.

As this study has shown, even when mapping only current stressors for which spatial data exist, most of the marine environment is affected by stressors resulting from humans activities. Given the extent of use of the ocean in BC and the paucity of protected areas, it is paramount that additional protection is offered to stop degradation and assist recovery while additional research is carried out on the impact and location of human activities. With the limited number areas that are either fully protected or not currently exploited, a related issue is the lack of reference areas to which impacted areas can be compared. Without a baseline to compare to degraded systems, compounded with the "shifting baselines syndrome" (Pauly 1995), it becomes very difficult to gauge the impact of human activities. The establishment of areas where direct impacts are eliminated can provide a basis for comparison to impacted areas. Yet anthropogenic activities can have an impact many kilometers beyond the location of occurrence, and thus even fully protected areas will likely continue to receive some stressors from outside of the boundaries.

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4. Moving towards spatial solutions in marine conservation with indigenous communities[4]

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Introduction

Marine Protected Areas can be a valuable conservation tool to halt the decline of overexploited fish and invertebrate populations (Dayton et al. 2000; Halpern & Warner 2002; Kelleher & Kenchington 1992; Walters 1998; Worm et al. 2006) and may also enhance fisheries (Abesamis & Russ 2005; Halpern & Warner 2002; McClanahan & Mangi 2000). However, less than 0.01% of the ocean is currently protected, and the pace of establishment of new MPAs is too slow to meet international conservation commitments (CBD 2006; Wood et al. 2007; World Summit on Sustainable Development 2003). This is a concern because people are increasingly dominating coastal ecosystems (Weinstein et al. 2007), and accelerating degradation (Ban & Alder 2008; Pauly et al. 1998).

MPA selection experiences repeatedly confirm the importance of engaging those affected by the designation (Drew 2005; Helvey 2004; Kessler 2003; Lundquist & Granek 2005; Morin Dalton 2001). Indeed, social factors are often highlighted as the primary determinants of MPA success (Drew 2005; Kessler 2004; Mascia 2003; Morin Dalton 2001). In particular, MPAs affect extractive users the most, which can be commercial, recreational, or subsistence / indigenous fisheries. To prevent fishing grounds from being fished more intensively due to the establishment of MPAs, a recommendation for improving MPA success is to proportionally reduce fishing pressure elsewhere (Halpern et al. 2004; Hilborn et al. 2006; Walters 2000).

Recently, there has also been increased international recognition of the rights of indigenous peoples' to resources on traditional territories, including marine resources (United Nations General Assembly 2007). Consequently, part of the challenge of establishing MPAs lies in adequately respecting such rights. In some countries, indigenous people constitute local communities as well as a level of government (*e.g.*, Canada: Harris 2002; Australia: Sanders 2002; U.S.A.: Zaferatos 2004), and they may also either claim or have established rights and title to marine resources (Bess 2001; Davis & Jentoft 2001; Mulrennan & Scott 2000; Ross & Pickering 2002).

Our research focused on indigenous peoples in Canada, where they are known as First Nations. They are considered a level of government and have rights established by common-law and protected by the Canadian Constitution to fish for food, social, and ceremonial purposes

(Constitution Act 1982). We refer to food, social and ceremonial fishing as indigenous or aboriginal fishing, and exclude commercial fisheries from this definition. In some cases, First Nations have confirmed a right to fish for commercial purposes. Many First Nations interpret the "social" component of fishing rights to include commercial fisheries, but the federal government disagrees. Whether or not commercial fishing is a First Nations' right is very contentious, and so far it is being addressed on a case-by-case basis as First Nations take the issue to court. Case-law has also affirmed the federal and provincial governments' responsibility to meaningfully consult First Nations and accommodate their interests when making resource management decisions (Harris 2002; Houde 2007). After the R. vs. Sparrow decision, the court rules that aboriginal rights to fish for food, social and ceremonial purposes have priority over all other uses of the fishery (Fisheries and Oceans Canada 2008). Such fisheries are termed aboriginal fisheries, and are a separate category from commercial and recreational fisheries, authorized by communal licenses (Fisheries and Oceans Canada 2008). First Nations commonly view strict marine conservation measures, *e.g.,* no-take MPAs, as an infringement of these rights, and are afraid that no-take MPAs would preclude them from exercising that right. Because potential infringement of rights and title are of enormous concern and part of a much larger issue than marine conservation, many First Nations have not been willing to let MPAs set a precedent of infringement. Hence, even though many First Nations have a strong conservation ethic, the issue has provided one more impediment to MPA establishment in Canada (Ayers 2005). First Nations' concerns regarding MPAs also stem from inadequate consultations (LeRoy et al. 2003), and a fear of compromising their negotiating position in the treaty process where treaties do not exist (Ayers 2005).

However, many First Nations are interested in ensuring sustainable use of the oceans because seafood continues to comprise an integral part of First Nations' culture and economy (Garibaldi & Turner 2004), and because they still rely on traditional foods for sustenance (Weinstein & Morrell 1994). First Nations communities are, of course, also faced with the universal challenges of balancing economic development with conservation.

There is some precedent in Canada to incorporate the sustainable use of marine resources by indigenous people into MPAs. In Canada's Arctic, the Beaufort Sea Beluga Management Plan includes a protection zone that permits traditional harvest of belugas by Inuit. Other activities are either excluded, or allowed only if they do not have a deleterious effect on belugas (Fast et al. 2001). These zones are currently being considered as MPAs under the Oceans Act, in order to support the beluga management plan (Fast et al. 2001). The situation in the Arctic differs from British Columbia, though, in that there are comprehensive land claim agreements in the Canadian North (Berkes et al. 2007). Because of this, governance regimes and responsibilities are well defined. This is not the case in British Columbia, where treaties have not been settled.

The purpose of this research was to develop and test a framework to integrate the preferences and concerns of First Nations into the site selection of potential MPAs, summarize their views, and document constraints and challenges. In particular, we address the following questions: (1) What kind of protection would they like to see and where, and how might this affect commercial fisheries? (2) How can their views potentially help to advance marine management, and are there any gaps in current marine conservation approaches? (3) What are the limitations of their suggestions? Our research was undertaken in partnership with the Gitga'at and Huu-ay-aht First Nations. We use the traditional territories of these First Nations as our case studies. Given the sociopolitical context for our work, we need to emphasize that our study is academic; the resulting information was shared with the First Nations partners and will only be used for planning purposes if the First Nations partners decide to do so.

Both study areas have had some involvement in conservation issues, although primarily terrestrial. Recently, the Gitga'at First Nation has been involved in the creation of the Great Bear Rainforest agreement. The Gitga'at and other First Nations on BC's North and Central coasts have recently established numerous conservancies through integrated planning of BC's coastal lands working with the BC government. Several of the new land conservancies in Gitga'at territory also include coastal and foreshore areas. However, marine fish harvesting activities are not explicitly included in these conservancies since it is not under the provincial government's jurisdiction. Gitga'at are currently developing joint management plans for these conservancies with the provincial government. The Huu-ay-aht's territory encompasses part of Pacific Rim National Park Reserve, and hence they have had some involvement in the national park. Generally, though, the Huu-ay-aht First Nation has been less involved in conservation issues recently compared to the Gitga'at First Nation. Much of the Huu-ay-aht's efforts have been on negotiating a treaty, which is currently in the final stages of approval (Indian and Northern Affairs Canada 2007).

While fisheries spatial management measures exist within both traditional territories, neither have MPAs. Some parks exist that have boundaries that extend into the marine environment (*e.g.,* Pacific Rim National Park Reserve), but none are presently managed for marine conservation. Rockfish conservation areas have been established within both case study areas, but these were designated to protect a guild of species, rather than marine resources more generally. From the perspective of marine conservation generally, spatial management of marine resources for conservation is limited at present in both case studies.

Methods

To carry out our research, we focused on two indigenous groups in British Columbia, Canada. The Gitga'at First Nation is a Tsimshian First Nation on the north coast of British Columbia. The Huu-ay-aht First Nation is one of the Nuu-chah-Nulth First Nations, located on the west coast of Vancouver Island. These First Nations were selected as case studies because of their interest in partnering in this research and because they differ in their participation in the treaty process: Gitga'at First Nation has temporarily suspended its involvement in the treaty process (http://gitgaat.net/contact/treatyoffice.htm), whereas the Huu-ay-aht people have ratified their treaty (http://www.maanulth.ca/about_fn_huu-ay-aht.asp). The two First Nations also provide contrasts in the accessibility of the main communities: the Gitga'at First Nation's community of Hartley Bay is more remote, accessible by a four-hour boat ride or by float plane (weather permitting); the Huu-ay-aht First Nation's town of Anacla is accessible by logging road, and is located in proximity to the small town of Bamfield.

While the First Nations are located hundreds of kilometers apart, they exhibit similar histories and social structures. All First Nations in British Columbia experienced rapid change since the arrival of Europeans (Harris 2002). While European customs and foods have been incorporated into daily life, many indigenous customs continue to be practiced, and the clans, or house groups, remain as an important component of village organization (Menzies et al. 2001). Fisheries and the collection of other seafoods was essential for survival prior to the arrival of Europeans, and as industrial fisheries expanded, many First Nations people made a living from various aspects of commercial fisheries (Harris 2001). As fisheries declined and ownership of licenses became more centralized, many First Nations lost access to commercial fishing opportunities (Harris 2001). However, as with other coastal First Nations communities, both the Gitga'at and Huu-ay-aht communities continue to use seafood in their diet and as a fundamental part of their culture and economy.

We developed a framework to integrate the preferences and concerns of First Nations into marine conservation. The framework consisted of three phases: (1) establishment of research collaborations, (2) semi-structured individual interviews with First Nations community members, and (3) feedback from the communities about marine conservation preferences obtained through the interviews (Figure 1). Phase 2 and 3 focused on the goals for the marine territories and in particular examined issues affecting the area, and preferred management solutions. Phase 2 built on components of issue-action analysis through interviews by identifying issues and the associated actions that could be taken to address them (Salm & Clark 2000). Phase 3 approximated consensual planning through community meetings (Innes 1996; Kay & Alder 2005). The framework we used differs from others (*e.g.,* as reviewed in Kessler (2004) and Brody et al. (2003)) because we interviewed individuals in addition to holding community meetings. Also, our research required participants to use their knowledge to make management recommendations, rather than collecting traditional ecological knowledge *per se* (Berkes et al. 2000; Drew 2005).

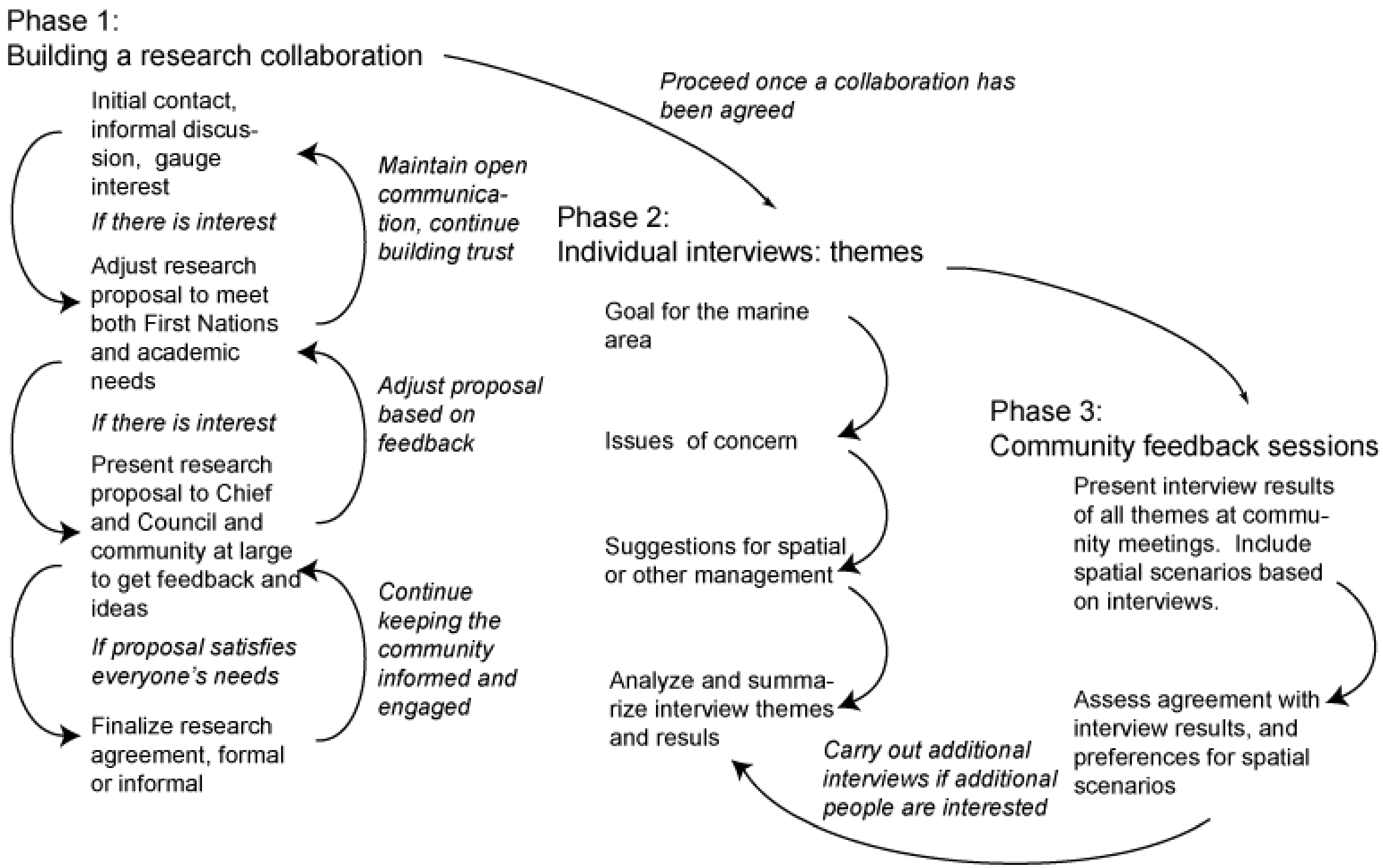

Figure 4.1. Framework for eliciting community preferences for spatial management options.

Phase 1: Establishment of research collaborations

To establish research collaborations, we started informal conversations with community contacts responsible for marine and/or fisheries management within the territories. After initial interest was established, we worked together to formulate the research approach, and took the proposal to the chief and council (the First Nations' governance body) for their approval.

Phase 2: Semi-structured individual interviews with First Nations community members

Interviews with members of the First Nations comprised the core of the research project. We used a snowball approach (Goodman 1961) to focus on people who are, or have been, active users of marine resources, and/or who have a particular depth of knowledge about the marine environment. We chose our focus because these people would be most affected if management changes were implemented. We stratified our snowball approach to ensure that we interviewed participants from all clans. We interviewed 20 self-identified Gitga'at marine resource users and 19 in Huu-ay-aht territory. We carried out semi-structured interviews in November 2005, February-March 2006, and November 2006 with Gitga'at members, and February to June 2007 with Huu-ay-aht members. In the Gitga'at territory, interviews were carried out in Hartley Bay (n=11) and in Prince Rupert (n=9), where about 2/3 of Gitga'at members currently live. In the Huu-ay-aht territory, interviews were focused in Anacla (n=17), with a couple of interviews in Port Alberni. We interviewed people from a range of age groups, with some younger (ages 18 to 35) participants (Gitga'at n=2, Huu-ay-aht n=4), most ranging from ages 36 to 60 (Gitga'at n=13, Huu-ay-aht n=11), and some over the age of 60 (Gitga'at n=5, Huu-ay-aht n=4). Because there is a traditional bias towards males in people who are out on the water, the majority of participants were male (Gitga'at n=15, Huu-ay-aht n=17).

All interviews included the same themes: the participant's goal(s) for the marine component of the territory, issues of concern, suggestions for spatial and other management, general opinions about no-take MPAs, and the participant's history of marine involvement (*i.e.,* indigenous, recreational, and/or commercial fishing; other extraction; processing, etc.). Responses to these questions were unprompted (hereafter referred to as unprompted responses). This means that we did not provide a list of possible answers to our questions. The style of the interviews was conversational, and we did ask probing questions without providing potential answers.

Interviews lasted between half an hour and four hours, and most were one-on-one, with four interviews in which two participants were interviewed at the same time. We kept detailed notes during the interviews.

The core of the interview process focused on participants' suggestions for spatial protection measures. In a map-based process, participants drew freeform polygons on tracing paper overlaying nautical charts of the study region to indicate their suggested areas for protection (Ardron 2005). Participants were comfortable reading nautical charts and using them to indicate their preferences. The nautical charts encompassed the entire marine portion of the Gitga'at and Huu-ay-aht territories. We did not specify the extent of the traditional territories during the interviews, but rather let participants decide where they felt comfortable suggesting protection measures. There are many areas of overlap and/or contention with neighboring First Nations.

Participants were also asked what kind of protection they envisioned for each of the polygons they drew. All information was stored using a Microsoft Access 2003 database and analyzed using ESRI's ArcGIS version 9.1. Spatial analysis focused on the management options drawn by participants. We combined polygons from each management suggestion to depict the overlap of suggested areas amongst participants. We created several management scenarios of the most commonly suggested type of protection to represent a range of options based on the overlap of polygons. We used the Getis-Ord General G statistic (Getis & Ord 1992) to test the congruence of areas selected by participants. We also asked participants about their opinion of no-take MPAs.

Phase 3: Feedback from the communities

Following individual interviews, we held open community meetings (November 2006 in Gitga'at territory, November 2007 in Huu-ay-aht territory) to present findings and receive feedback. All community members were invited to attend, and participants self-selected their attendance. Interview participants and others attended these meetings, including a range of age groups and positions within the community. Meetings allowed people to see the aggregated results of interviews, and to voice their opinion on the areas marked by interview participants. Prior to discussing the results, we asked participants to record (individually and anonymously) their agreement with the management scenarios on a feedback form, using a five-point Likert scale (Matell & Jacoby 1971). The form also asked them to check a box if they had been interviewed for our study. Once the feedback forms had been collected, we discussed the scenarios derived from interviews, with participants providing comments on potential protected areas that had been missed, were superfluous, or with which they disagreed. We incorporated these changes into the GIS file to most closely approximate the goal that emerged from the meetings. We then calculated the area covered by these maps of preferred protected areas. For the Gitga'at territory we analyzed the proportion of traditional fishing areas encompassed by the map of preferred protected areas. We did not have traditional fishing locations for the Huu-ay-aht territory.

Possible impacts of community zoning on a key stakeholder - commercial fisheries

We then analyzed how community-chosen protected areas might impact commercial fisheries. We obtained spatial catch information from 1993-2005 from Fisheries and Oceans Canada in a summarized form of either 4 km by 4 km or 10 km by 10 km grid cells. We received these data for eleven commercial fisheries found in the study areas. Spatial data for other fisheries was not available. We calculated the mean catch in kilograms per year for each fishery. We scaled catch values to the suggested protected areas, which might be less than cell size, thereby assuming that catches were distributed homogeneously within each grid cell. Spatial data on sports fishing catches are not collected by the government so the analysis was limited to commercial fisheries data.

Results

Participants in our study were associated with a range of resource extraction and management activities. The majority of participants in the Gitga'at case study have been involved in commercial fisheries (75%), and some were seasonally employed by sports fishing lodges (10%) or involved in other tourism activities (5%). Similarly, in the Huu-ay-aht case study the majority (79%) have been involved in commercial fisheries, while some were involved in recreational fisheries (16%), shellfish aquaculture (11%) or stream restoration (11%).

Goal of the area

Participants' goals of what they would like to see in their territory (unprompted responses) were remarkably congruent within the communities and between case study areas. The most commonly stated priority in Gitga'at territory (85%) was to ensure that food fisheries were protected for present and future generations, with recovering depleted species as the next goal (80%), with particular reference to abalone (*Haliotis kamtschatkana*) (35%). In Huu-ay-aht territory the order was reversed, with recovering depleted species as the most common goal (95%) and food fisheries as the next priority (64%). Other priorities expressed were a desire for co-management of activities that take place in the territory, and additional economic opportunities including commercial fisheries. In both communities, participants commented on declines in the abundance of species. In particular, many participants noted decreases in the abundance of salmon, groundfish (especially rockfish), herring, eulachon, and halibut.

Issues of concern

In both communities, commercial and recreational fisheries were a concern because of the observed environmental changes due to biomass extraction (Table 1). Other issues raised included illegal fishing, the environmental impacts associated with tourism, and frustration at the waste of bycatch with quota regulations. Issues unique to Gitga'at territory included potential future impacts of a proposed pipeline and associated oil tanker traffic. In Huu-ay-aht territory, the anticipated decline of shellfish due to increasing abundance of sea otters was a concern.

Table 4.1. Proportions of the participants who raised particular marine issues during semi-structured interviews in two First Nations territories (unprompted responses).

	Gitga'at territory (n=20)		Huu-ay-aht territory (n=19)	
Type of issue	Proportion	Ongoing or potential/ anticipated issues*	Proportion	Ongoing or potential /anticipated issues*
Population declines due to commercial fishing	80%	Ongoing	68%	Ongoing
Population declines due to recreational fishing	75%	Ongoing	68%	Ongoing
Population declines due to illegal fishing	45%	Ongoing	5%	Ongoing
Potential environmental damage of the proposed pipeline and associated oil tanker traffic (*e.g.,* oil pills)	45%	Anticipated	NA	
Environmental degradation associated with tourism	30%	Ongoing	37%	Ongoing
Habitat damage caused by logging and its effects in the ocean	25%	Ongoing	11%	Ongoing
Population declines due to bycatch, and wastefulness of bycatch regulations	20%	Ongoing	0%	
Environmental degradation due to finfish farming	20%	Anticipated	16%	Ongoing
Unpredictable changes due to climate change	20%	Ongoing	5%	Ongoing
Mismanagement of fisheries	20%	Ongoing	32%	Ongoing
Uncertainty in seismic testing on marine life	20%	Anticipated	NA	
Environmental degradation due to oil and gas exploration	15%	Anticipated	0%	
Habitat damage due to anchoring	5%	Ongoing	0%	
Commercialization of seaweed	5%	Anticipated	0%	
Population declines due to First Nations fishing	5%	Ongoing	16%	Ongoing
Environmental impacts of pollution	5%	Ongoing	16%	Ongoing
Invertebrate declines due to increasing abundance of sea otters	0%		26%	Anticipated
Fish population declines due to increases in seals and sea lion	0%		21%	Ongoing
Habitat damage due to trawling	0%		16%	Ongoing
Changes in food web interactions due to invasive species	0%		5%	Ongoing
Genetic contamination due to hatcheries	0%		5%	Ongoing

* Ongoing refers to issues that are currently taking place and are expected to continue into the future. Potential/ anticipated refers to issues that are not yet occurring but will potentially appear in the future.

Results from individual interviews: suggestions for spatial protection measures

We found strong agreement amongst participants in the suggested types of protection. The vast majority of participants suggested areas that should exclude commercial fishing, recreational fishing, and /or both (Table 2), while allowing indigenous exploitation. Most participants did not provide details on which commercial fisheries should be allowed or excluded. Rather, most participants suggested the exclusion of commercial fisheries in general. Similarly, participants did not distinguish between recreational fishing from sports fishing lodges and individual sports fishermen, instead suggesting the exclusion of recreational fishing in general. A few people suggested no fishing zones or other types of areas. On average, each person in Gitga'at territory selected 21 areas for protection (range 7 to 38), comprising 6.2% (range 2% to 21%) of the claimed marine territory. In Huu-ay-aht territory each participant suggested an average of 4 areas for protection (range 1 to 14), comprising 3.6% (range <1% to 27%) of the claimed territory.

Table 4.2. Spatial marine protection measures suggested by participants

	Proportion of participants who suggested this management zone	
Type of management zone	*Gitga'at territory (n=20)*	*Huu-ay-aht territory (n=19)*
No commercial fishing	100%	91%
No recreational fishing	100%	64%
Neither recreational nor commercial fishing	100%	64%
No fishing	14%	
No tourism		18%
Tourism area	14%	
Other	14%	18%

There was a significant degree of overlap amongst community members for suggested areas to exclude from commercial fishing, recreational fishing and both (Figures 2 and 3). The Getis-Ord General G spatial statistic indicated less than 1% likelihood that the pattern for each of the three management types (within management measures amongst respondents) could be the result of chance for the Gitga'at study area. The same applies to the Huu-ay-aht study area for the no commercial fishing suggestions, and less than 10% likelihood that the pattern is a result of chance for the other two management suggestions. Some participants recognized that their suggestions for protection measures fell into areas that overlap with neighboring First Nations.

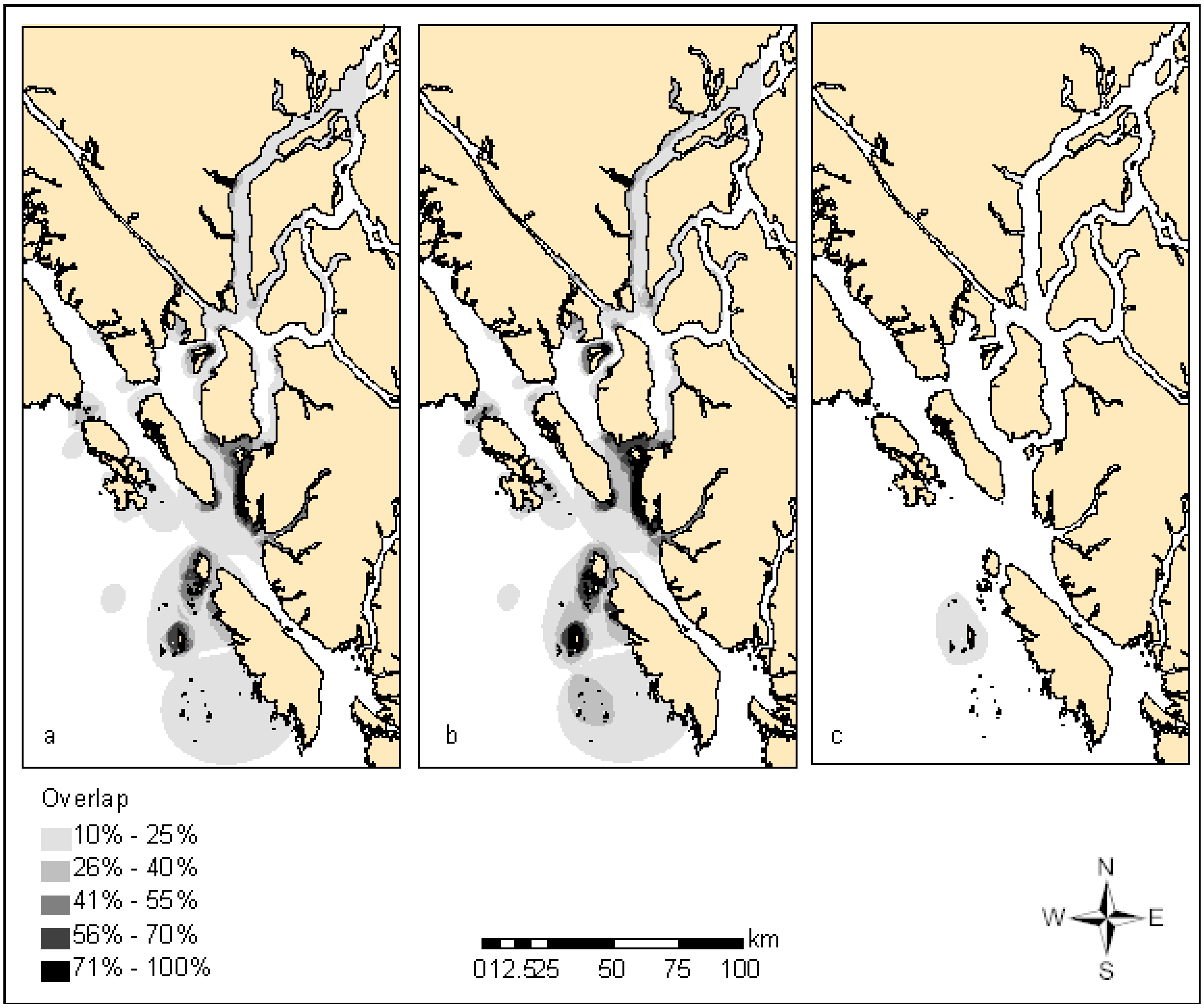

Figure 4.2. Gitga'at case study, British Columbia: Overlap of spatial protection zones suggested by interview participants (n=20). Types of zones suggested are 'a' no recreational fishing, 'b' no commercial fishing, and 'c' no fishing. Map 'a' and 'b' are similar because participants commonly drew polygons that they felt should be closed to both commercial and recreational fishing. Participants could make as many management suggestions as they wished. The areas noted on the map reflect participants' interpretations of the traditional territory.

Figure 4.3. Huu-ay-aht case study, British Columbia: Overlap of spatial protection zones suggested by interview participants (n=19). Types of zones suggested are 'a' no recreational fishing, 'b' no commercial fishing, and 'c' neither commercial nor recreational fishing, but aboriginal fishing allowed. Map 'c' shows the overlap of maps 'a' and 'b'. Participants could suggest as many management suggestions as they wished. The areas noted on the map reflect participants' interpretations of the traditional territory.

Only 2 people in the Gitga'at case study, (and none in the Huu-ay-aht case study) suggested areas where no fishing, including aboriginal fishing, should take place. However, when asked, most participants (60% Gitga'at, 43% Huu-ay-aht) offered no strong opinion for or against such no-take areas. The rest indicated support for the concept. Two people in the Huu-ay-aht study area opposed the idea.

Results from community feedback sessions

Feedback forms and discussion at the community feedback session in the Gitga'at study area revealed that participants at the meetings (n=21) preferred a scenario that covered a large portion of the territory (between scenarios in Figure 4a and 4b). Using the feedback form, 83% of meeting participants gave scenario ‘a’ the highest ranking, compared to 50% for scenario ‘b’, 17% for scenario ‘c’ and ‘d’, and 33% for scenario ‘e’. Some respondents gave multiple scenarios their highest ranking. The discussion that followed revealed that people at the meetings thought that scenario ‘a’ was too large an area, whereas a few areas represented in ‘a’ but not ‘b’ were seen as important for protection. The discussion mirrored the responses received on the feedback forms. The areas that were missing from ‘b’ were subsequently added to create a map that currently represented the closest approximation to the community goal for areas for protection. This map has 26 areas, each with a mean size of 2,250 hectares, comprising in total 7% of the marine territory of the Gitga’at First Nation, and 82% of Gitga’at traditional fishing [point] locations as identified by elders (Chris Picard, unpublished data). When excluding the large offshore component of Gitga’at territory (5km offshore from outer islands and beyond) – which few participants knew, used, or nominated – a total of 15% of Gitga’at inshore waters were selected for protection.

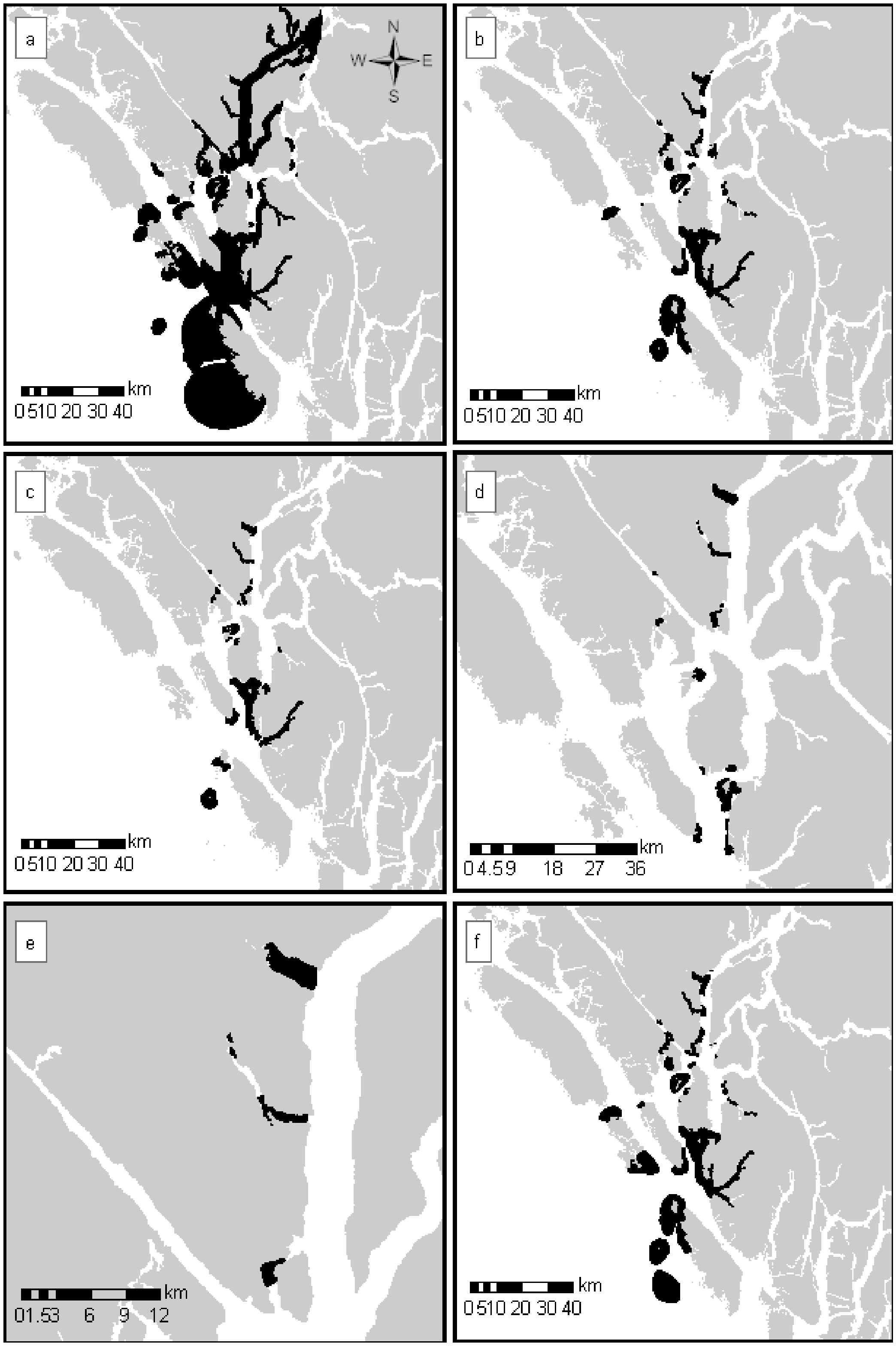

Figure 4.4. Gitga'at case study: Five scenarios were created from individual interviews for the most common protection type where only aboriginal rights fisheries would be allowed: 'a' all areas selected by any participant; 'b' areas selected by 30% of participants; 'c' areas selected by 50% of participants; 'd' areas selected by 70% of participants; and 'e' areas selected by all participants. 'f' depicts the revised preferred protected areas based on feedback meetings. The areas noted on the map reflect participants' interpretations of the traditional territory.

The feedback results were similar for the Huu-ay-aht session (n=8). Using the feedback form, 80% of participants gave scenario ‘b’ the highest ranking, followed by scenario ‘a’ and ‘c’ as the second most popular choices. As with the Gitga’at case study, it was the larger, but not largest, of the options that was preferred. This most preferred option consists of 4 areas, each with a mean size of 1,216 hectares, comprising 0.4% of the marine territory. When excluding the offshore component, this rises to 3% of the territory. Due to inclement weather, the feedback session was not very well attended. Follow-up conversations with people unable to attend confirmed the preferences stated by participants at the meeting.

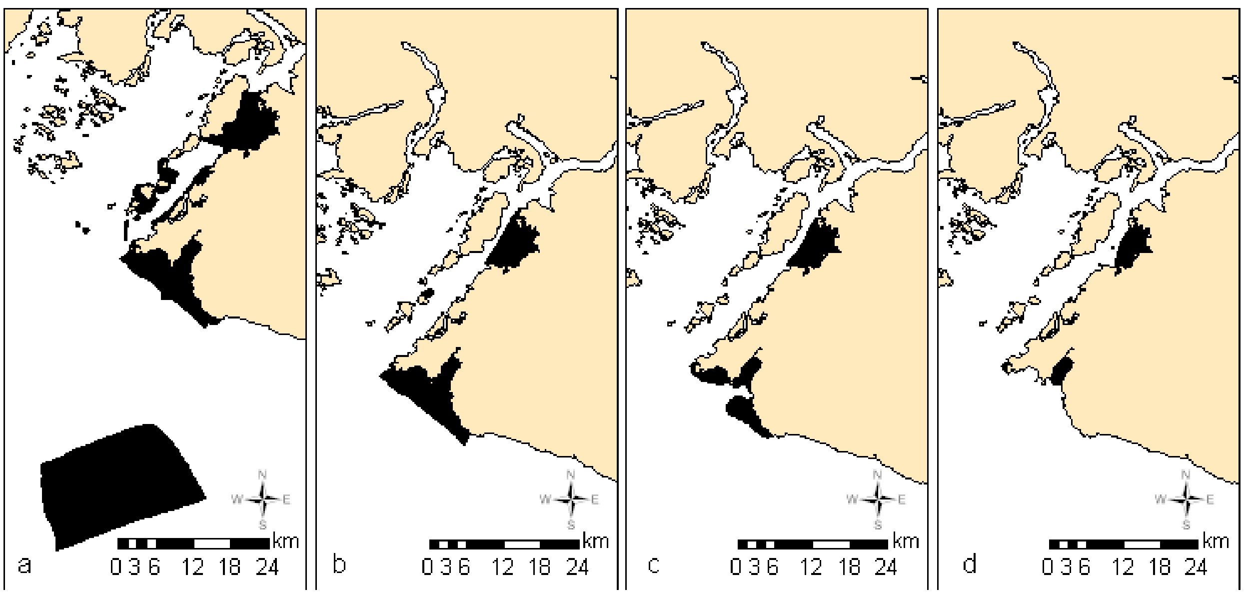

Figure 4.5. Huu-ay-aht case study: Four scenarios were created from individual interviews for the most common protection type where only aboriginal rights fisheries would be allowed: ‘a’ all areas selected by any participant; ‘b’ areas selected by 20% of participants; ‘c’ areas selected by 30-50% of participants; and ‘d’ areas selected by >50% of participants. No areas were areas selected by all participants. ‘b’ was most preferred by participants at the feedback meeting. The areas noted on the map reflect participants’ interpretations of the traditional territory.

Potential impacts on commercial fisheries

Our analysis of the potential impact of the suggested protected areas on commercial fisheries revealed that the commercial catches are roughly proportional to the area of the preferred protection areas for the whole territory (Table 3). However, some fisheries would be impacted much more than others, and we did not have spatial catch data for all fisheries. Notably, spatial information for commercial salmon and halibut was not available. These are highly mobile species, though, and therefore the corresponding fisheries may have flexibility in adjusting their fishing areas. If the First Nations decide to pursue implementation of these areas as MPAs that permit aboriginal food fishing only and reduce commercial quotas so that fishing effort is not increased outside of the preferred protection areas, then those reductions would be about proportional to the percent of the marine territory protected. The two active Gitga'at commercial fishermen whom we interviewed both indicated a willingness to avoid the preferred protection areas for their commercial fishing, should they be implemented. Both of these participants suggested MPAs in areas where they currently fish commercially, noting that they could change the areas where they fish commercially.

Table 4.3. Overlap of preferred protection areas and commercial fisheries for which spatial catch data are available.

Fisheries and Oceans Canada fishing data layer			Gitga'at area commercial catches within preferred protection areas		Huu-ay-aht area commercial catches within preferred protection areas	
Fishery	Grid size	Years	kg/yr	% of catches within the territory	kg/yr	% of catches within the territory
Crabs	4km	2000-2004	791.5	7.8%	589.1	11.9%
Geoduck	10km	2002-2004	13,161.3	10.2%	388.7	17.0%
Groundfish trawl	4km	1996-2004	0	0.0%	0	0.0%
Prawn	4km	2001-2004	1,626.8	7.5%	3,736.2	22.4%
Red urchins	10km	1997-2003	220,638.9	14.9%	1,078.9	14.0%
Sablefish trap	4km	1996-2004	151.5	15.4%	0	0.0%
Sablefish longline	4km	1996-2004	0	0.0%	0	0.0%
Schedule 2 (hook and line)	4km	1996-2004	108.8	1.5%	0	0.0%
Sea cucumber	10km	1997-2004	22,381.7	10.0%	na	na
Shrimp trawl	4km	1996-2004	0	0.0%	2,482.3	0.1%
ZN (hook and line)	4km	1993-2004	2,212.2	7.1%	84.1	5.2%
Total			261,072.7	7.4%	8,359.3	7.1%

Discussion

In this research, we sought to synthesize the perspectives of indigenous people on marine conservation issues. Amongst participants of this study we found a willingness to embrace spatial protection measures. The main goals for the marine territories amongst participants were the recovery of depleted species and sustainability of indigenous fishing, and the preferred management approach was to protect areas from commercial and recreational fishing. Overall, our research revealed a gap in our current conservation efforts: areas of importance to indigenous people warrant protection that allows their continued use of those areas. However, the scientific literature places much greater emphasis on no-take MPAs, and hence additional research is needed to assess the efficacy of partial-take MPAs.

The areas most commonly identified for protection were of high importance for indigenous fishing, an activity that occurs within the context of First Nations' legally-protected right to fish. Even though indigenous people have a right to fish for food, social and ceremonial purposes in Canada, transforming that right into practice is an ongoing struggle with a proliferation of lawsuits (Harris 2002; Houde 2007). Elsewhere in the world, indigenous people similarly struggle to be involved in resource extraction and management decisions within marine areas claimed as their territory (Bess 2001; Davis & Jentoft 2001; Mulrennan & Scott 2000, 2001). This makes it particularly important to incorporate indigenous perspectives into marine protection and zoning, and to create areas that are specifically established to protect indigenous fishing grounds.

The First Nations we interviewed shared the goal (within and across communities) of recovering depleted species and ensuring sustainability of traditional foods. This conservation focus was by no means a foregone conclusion. The emphasis could have been placed on, for example, economic development or commercial fishing opportunities, especially given the high level of unemployment in both communities. The common goals likely reflect the ongoing reliance of British Columbia coastal First Nations on traditional foods, especially seafood, to supplement their diet (Weinstein & Morrell 1994). These foods are also an integral part of indigenous culture (Garibaldi & Turner 2004). The goals expressed by participants are very similar to conservation and ecosystem-based management objectives of ensuring the persistence and representation of species (Margules & Pressey 2000), and allowing for sustainable fisheries (Pikitch et al. 2004).

The apparent emphasis on conservation by participants also incorporates other issues. In particular, by suggesting the exclusion of commercial and recreational fishermen from their historical fishing areas, participants are hoping to assume some control over the use and management of these areas. Their suggestions entail implementation of aboriginal rights in a conservation context. This can be interpreted as an attempt by participants to manage common pool resources by excluding other users (Ostrom et al. 1999). Also, participants commented on the decline of many species within their territories, notably abalone, salmon, groundfish, herring and eulachon. These declines have been happening with the federal and provincial governments as the managing authorities (*e.g.,* Slaney et al. 1996). The resulting lack of confidence in the current management of marine resources is likely another factor contributing to a desire by participants to be involved in the management of marine resources in their territories. Given their local knowledge of the trends in marine species, participants noted frustration in observing declines without being able to assist in managing for recovery.

Not many participants mentioned access to commercial fishing opportunities as a priority when asked about their vision for the future. This omission is interesting given that both communities used to be engaged in commercial fisheries, yet few people remain involved. Also, fishing for food is expensive in the case study areas, requiring at minimum boat and gear maintenance, and fuel; commercial fisheries could be seen as a means to subsidize indigenous fishing. The limited mention of commercial fisheries by participants could be explained in several ways. First, it could also be that the local resource depletions noted by participants suggest to them that commercial fisheries may not be a viable option. Second, recent development emphasis in both communities has been on shellfish aquaculture, perhaps decreasing the perceived importance of commercial fisheries. Third, it is also possible that participants perceived the commercial fishery issue as being pursued through other means, and hence did not think to mention it during the interviews. For instance, the Huu-ay-aht First Nation is part of the Maa-nulth treaty that is currently under negotiation. The harvest agreement of the Maa-nulth treaty includes long-term, guaranteed and renewable commercial fishing privileges for Maa-nulth First Nations to a defined share of the commercial catch (Indian and Northern Affairs Canada 2007). The Gitga'at First Nation has several successful participants in the commercial salmon and groundfish fisheries. They are currently using this core of fishing capacity to incrementally increase community participation in these and other fisheries. They are also seeking additional acquisition of commercial fishing assets (*i.e.,* licenses, quotas and equipment) through partnerships with other First Nations, governments and private interests. Finally, the conservation focus of most of our questions may have had the unintended consequence of discouraging responses that might imply increased fishing effort.

In considering the proposals made by First Nations' respondents, it is important to note that most were involved in either commercial or sports fishing, the very activities they suggest should be excluded from the protected areas. Their insistence on allowing only aboriginal fisheries in some areas was therefore not just self-serving. Many of the aboriginal commercial fishers interviewed stated that they would be happy to avoid the preferred protection areas while fishing commercially. It does seem, however, that First Nations people would benefit the most from the suggested protection measures. Commercial fisheries, for example, would be displaced approximately proportionally to the area protected.

The two Nations agreed on the most important protective measures, but the Gitga'at selected more areas. Such a difference might be associated with the longer, more complex coastline in the Gitga'at territory, or their ongoing use of seasonal camps widely-scattered across the territory. When considering the inshore areas, the percentage of the Gitga'at territory selected as preferred protected areas (15%) exceeds the 10% goal of the Convention on Biological Diversity (CBD 2006), but falls short of the 20% to 50% commonly suggested in the scientific literature (Dahlgren & Sobel 2000; Parnell et al. 2006; Plan Development Team 1990; Roberts et al. 2003b; Stewart et al. 2007). The Huu-ay-aht case study falls short in reaching any of these targets. The differences in the number of areas selected emphasizes that each local context will provide its own solutions.

The preferred scenarios discussed at the feedback meetings were not those that had the most overlap, but rather those that covered a larger – but not the largest – area. While our focus was not on the dynamics that occurred at the feedback sessions, we observed that no one person dominated the meetings. Participants included hereditary chiefs, however, and tradition dictates that they are allowed to speak before others. It did not seem to us that this tradition prevented others from speaking up, but perhaps further study of the dynamics in such meetings in indigenous communities is warranted.

The specifics of how the MPAs suggested by participants might be managed, and the rules of engagement, were not the focus or our work. Should the First Nations seek to pursue the establishment of such MPAs, setting out some rules of how these areas might be managed, will be very important. Many participants mentioned the importance of the First Nations having a lead or key role in the management of these areas. Co-management was mentioned many times as an important concept. Also, the rules of engagement would need to be very clear. For example, what activities would be excluded, and which ones would be allowed? In our interviews we focused on getting participants' suggestions for marine management, rather than setting out such a set of rules.

If the community-preferred areas are to serve as the backbone of a MPA network, then those areas should be analyzed for representation of habitats and species. Systematic conservation planning emphasizes that all ecosystem components should be represented in a protected area network (Margules & Pressey 2000). If the local goal is to recover depleted species, then those species and their habitats have to be included in the network. Decision support tools such as Marxan (Ball & Possingham 2000; Possingham et al. 2000) can be used to assist in planning to ensure representation. The community-preferred areas can be used as the core for building such a network.

The goals of First Nations, and those outlined in MPA theory (*e.g.,* Roberts et al. 2003a) follow similar themes, but are not the same. The main difference is that MPA theory emphasizes representation (Margules & Pressey 2000; Roberts et al. 2003a), which was not mentioned by participants. Through the principle of representation, MPA theory values all species and habitats, and recommends their inclusion in MPAs. First Nations, on the other hand, may value some species more than others (*e.g.,* Garibaldi & Turner 2004), and therefore may focus conservation efforts on those species. However, the indigenous world-view of "everything is one" is prevalent amongst many First Nations in British Columbia (*e.g.,* Atleo 2004). This world-view recognizes the importance and interconnectedness of all ecosystem components.

One of the limitations of following recommendations from participants is that effective conservation tools may be missed. In our case studies, few people suggested no-take areas, for example, which have much more scientific evidence of success than partially protected areas (Hutchings 2000; Murawski et al. 2000). We did not specify the type of protection when interviewing participants. As a next step, it would be possible to use the community-preferred areas as the basis for a network of MPAs, and then add other types of zones, such as no-take areas, to strengthen the network. This could be done either through additional interviews with community members and through community meetings, and/or using a decision support tool such as Marxan. Our discussions with participants suggest that there may be support for no-take areas once a basic set of aboriginal fishing areas have been protected. Some participants commented that no-take areas should not be located in important traditional fishing areas, and that First Nations communities should have a say in where no-take zones should be placed. Indeed, Canadian governments have a legal obligation to consult and accommodate prior to making such decision (Harris 2002).

The scientific literature places emphasis on no-take MPAs (Ballantine 1995; Botsford et al. 2003; Dayton et al. 2000; Gell & Roberts 2003; Halpern & Warner 2002; Roberts et al. 2003a; Roberts et al. 2003b), yet the preferred spatial management option by indigenous people interviewed for this study was to allow indigenous extraction for food, social and ceremonial purposes to continue. Since the Biodiversity Convention and many national laws declare that aboriginal rights must be respected, where lies the future for no-take MPAs? One route may lie in engaging First Nations in extended talks on the merit of no-take MPAs, which are already of potential interest to some of our respondents. However, given the past history of First Nations' opposition to no-take zones (Ayers 2005; Guenette & Alder 2007), it would also be logical to develop a category of protection that permits indigenous extraction for food. Such MPAs are being pursued in the Canadian Arctic (Berkes et al. 2007). The IUCN categories for parks include a sustainable use category (Phillips 2003), which could be adopted for the marine environment.

However, the effectiveness of such partially closed marine areas has not been studied extensively, and their efficacy compared to no-take areas is debated (Agardy et al. 2003; Ballantine 1995, 1999). Some researchers suggest that certain types of partially protected areas may provide similar benefits to fully closed areas (Ley et al. 2002; McClanahan et al. 2006), but others disagree (Hutchings 2000; Murawski et al. 2000). If other extractive activities are prohibited in such areas, then the reduced total extraction should result in conservation benefits as well. Such aboriginal fishing zones certainly have the support of participants in our research, and hence may provide a solution to the slow pace of MPA establishment. User groups would need to be supportive as well, and just compensation would be necessary. Further research is urgently needed to identify the conservation effectiveness of zones where limited extraction is allowed to continue.

Conclusion

Marine protection is a controversial topic in British Columbia and elsewhere in the world (Ayers 2005; LeRoy 2002), and the engagement by indigenous people in this research and the success of the framework was by no means certain. Our three-phased framework was very well received by participants, and may be a promising approach for other communities. All three phases were important in the success of our project:

(1) Developing research partnerships, including data sharing agreements, clearly laid out the objectives of our work, how we would partner with the First Nations, and served to develop a level of trust in our partnership. The data sharing arrangement was perhaps the most important aspect of our work, as it allowed the First Nations partners to retain control of the information and how to use it in the future. Setting up partnerships can be very time consuming, as are individual interviews. If indigenous groups carry out such research and marine planning themselves, however, then the long process of building partnerships would be superfluous.

(2) The advantage of the individual interviews was that they allowed us to build individual relationships with participants, and they encouraged participants to express their own goal, issues, and suggest management actions. The disadvantage of the individual interviews was that they are time-intensive. However, given the importance of participation (Dalton 2005; Kessler 2003, 2004), the additional time investment may be worthwhile for achieving greater engagement.

(3) The community meetings provided an opportunity for interview participants and others to see the results of the study and provide feedback. Community meetings alone, however, have been criticized for incorporating only the opinions of the loudest and most outspoken participants (Petts et al. 2000).

The suitability of participants' suggestions in addressing local issues, and the apparent willingness of participants to forgo some personal benefits to achieve their goal, emphasizes the importance of eliciting perceptions of issues and encouraging locally appropriate solutions. We found strong support for spatial protection measures amongst participants. Our study highlights a gap in our current conservation approaches: The conservation of areas that are important to indigenous people, where they can continue to practice and adapt their culture.

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5. Minimum data requirements for designing a set of marine protected areas, using commonly available abiotic and biotic datasets[5]

[5] A version of this chapter has been submitted for publication: Ban, N. Minimum data requirements for designing a set of marine protected areas, using commonly available abiotic and biotic datasets

Introduction

Managers, planners, and conservationists are commonly forced to make decisions with limited data when designing marine protected area (MPA) networks. Political and social pressure to establish MPA networks quickly has grown rapidly in recent years due to the decline in many marine species and degradation of habitats. I used two relatively data-rich marine regions in British Columbia, Canada, to address the question of how many datasets are necessary to proceed with a systematic site selection approach for biodiversity representation. I also tested the effectiveness of abiotic data and biotic data to serve as surrogates for each other. This research is directly relevant to practitioners faced with decision-making given limited marine biodiversity data, and contributes to systematic conservation theory by highlighting minimum requirements for robust results using a decision-support tool.

Marine protected areas, especially no-take MPAs or marine reserves, have been shown to increase biomass, size, density, and diversity of fishes within their boundaries (Halpern & Warner 2002; Roberts et al. 2001; Tetreault & Ambrose 2007), leading to international interest in establishing such reserves to protect biodiversity (Convention on Biological Diversity 2004; World Summit on Sustainable Development 2003). Yet few countries have protected more than a small percentage of their marine waters (Wood et al. 2007) and there is urgency to use available data to identify sites for additional protected areas. How can we make the best use of the marine data that are typically available to identify areas for protection? How many datasets are enough for robust analyses?

A range of approaches to site selection is emerging in the marine protected area (MPA) literature, with most emphasis on either of two extremes: 1) community or stakeholder-based site selection, often referred to as ad-hoc or opportunistic selection (Ballantine 1997, 1999; Pressey 1994; Salmona & Verardi 2001; Stewart et al. 2003; Walmsley & White 2003), and 2) science-based, systematic selection of MPAs (Leslie et al. 2003; Roberts et al. 2003a; Roberts et al. 2003b). In many cases, a combination of these site selection approaches is used, incorporating both stakeholder and science approaches, with varying levels of emphasis on different parts of the spectrum (Ban 2008). For this study, I focus the analysis on a scientific approach because of its increasing popularity (Sarkar et al. 2006; The Ecology Centre 2004), and because of the importance of ensuring representation of marine biodiversity within MPAs.

I explore a systematic approach to selecting MPAs to ensure representation of known and mapped components of biodiversity (Margules & Pressey 2000; Roberts 2000; Roberts et al. 2003a; Sala et al. 2002). Systematic conservation refers to the selection of a set of MPAs that together achieves biodiversity protection goals. Such planning requires a decision about which surrogates to use to represent biodiversity, the formation of objectives, and the articulation of simple and explicit methods for selecting new reserves (Margules & Pressey 2000).

Mathematical formulae, or algorithms, are commonly applied to make the planning process as efficient as possible while meeting conservation objectives. Such algorithms are the most practical tool for achieving systematic reserve selection because they are freely available and facilitate the use of spatial data on species distributions or habitats (Pressey & Cowling 2001; Pressey et al. 1997; Pressey et al. 1996). Selection algorithms have also been applied in recent marine planning ventures (Airamé et al. 2003; Fernandes et al. 2005; Sala et al. 2002). I therefore used such a selection algorithm, Marxan, to answer my research questions (Ball & Possingham 2000; Possingham et al. 2000).

While many countries have committed to establishing networks of MPAs for biodiversity protection, there is a dearth of marine biodiversity data to make informed decisions using a systematic planning framework. Datasets that do exist are commonly biased in that they over-represent charismatic or easily detectable species, are more heavily surveyed near field stations, or are commercially important (Grand et al. 2007; Mace et al. 2000; Possingham et al. 2000). Given the reality that data will never be complete, the most common types of biodiversity features used in conservation planning are a combination of easily accessible abiotic (physical or coarse-scale) data and biotic (primarily species-specific) layers (Brooks et al. 2004). The challenge is to make the best use of such datasets and understand the biases and constraints of using data collected in disparate ways.

MPA design theory asks for more detailed data than are generally available. For example, to have a successful network of MPAs, theoretical precepts say that the protected patches need to ensure persistence of biodiversity in addition to representing all components of biodiversity, termed representation (Mace et al. 2000; Margules & Pressey 2000; Roberts et al. 2003a). This could be achieved by including connectivity (Palumbi 2004), population viability analyses (Guichard et al. 2004) and extinction risk (Nicholson et al. 2006) into reserve design. While the necessary data for these analyses are occasionally available or can be collected in few locations or small areas for particular species (Parnell et al. 2006; Salomon et al. 2006), they are generally too difficult and costly to obtain at the scales used for planning MPA networks, and therefore not practical for planners to consider when protecting more than a few species. I therefore only consider representation in my study, to approximate conditions under which MPA decision support tools are currently used. My inclusion of coarse abiotic features can be seen as a very rough measure of ecological processes in the design (Pressey et al. 2007).

The purpose of this research was to (1) examine how many datasets are necessary for robust reserve design, and (2) assess the effectiveness of abiotic and biotic data as surrogates for each other. I used data as commonly available; systematically surveyed high quality data did not exist for the study areas (but see Grand et al. 2007 for an analysis of surrogates based on systematically surveyed data; Ward et al. 1999). Specifically, I compared abiotic and biotic data, as well as the following proxies for biotic data: very coarse data, local and traditional knowledge, and fisheries catch data. I wished to determine whether some datasets were more critical to the outcome and conversely whether there was redundancy in other datasets. I also wished to assess whether features with restricted distributions had a disproportionately large influence on the identification of priority areas.

I examined the research questions in the traditional marine areas of the Gitga'at and Huu-ay-aht indigenous peoples in British Columbia, Canada, using readily available abiotic and biotic spatial data. Compared to most marine regions in the world, British Columbia is relatively data rich. While there remain many data quality and bias concerns, this region can nevertheless be used to assess the relative importance of different datasets to the design process. Because of the relative data richness, I used the case study areas to approximate regions with poorer data by removing datasets from the analyses. My focus was on the influence of data once decisions about some inputs into the decision-support tool (planning unit shape, size, and compactness of the desired reserves) had already been made (see Leslie et al. 2003; Stewart et al. 2003; Warman et al. 2004 for sensitivity analyses of these inputs).

Methods

The study areas

I located my study in the claimed traditional areas of two indigenous groups in British Columbia, Canada, called First Nations, as the case studies (Figure 5.1). The study area of the Gitga'at First Nation was 918,309 hectares, and that of the Huu-ay-aht First Nation was 282,996 hectares. Both lie in temperate marine waters of the Pacific Ocean. I focused on these areas and at a regional scale because indigenous peoples are increasingly active in marine planning in British Columbia, and want to develop marine use plans for their areas. I communicated my findings to my indigenous partners, and provided them with copies of all analyses and maps for use in their marine planning as desired.

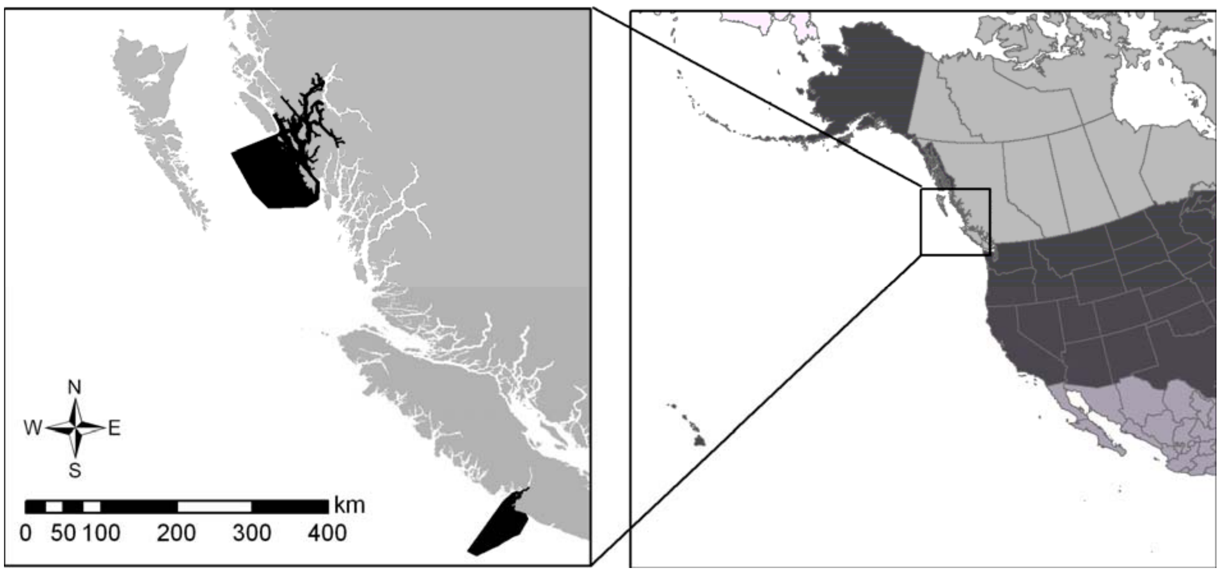

Figure 5.1. Maps showing the location and extent of the study areas in British Columbia, Canada: a) Gitga'at study area, b) Huu-ay-aht study area.

The study areas contained very similar features. For the Gitga'at study area, with all types of data included, there were 117 features, consisting of 25 abiotic features, 27 biotic features, 27 coarse-scale biotic features used as proxies (NOAA features), 28 local and traditional ecological knowledge (LEK) features as biotic proxies, and 10 fisheries catch features as biotic proxies (hereafter referred to as fisheries data or features). For the Huu-ay-aht study area, with all types of data included, there were 88 features, consisting of 24 abiotic features, 23 biotic features, 31 course-scale biological features (NOAA data), and 10 fisheries features.

The decision support tool and how it works

I used the decision support tool Marxan (Ball 2000; Ball & Possingham 2000; Possingham et al. 2000), which is designed to identify representative, spatially compact and efficient reserves. It does this by calculating reserve options that achieve targets set by the user at a minimum cost. In preparation for using Marxan, I divided the study areas into planning units, or grid cells, which I then populated with the biodiversity components. These biodiversity components (*e.g.,* species distributions, habitat types, bathymetry, etc.) are called conservation features. I then set a target, which is the percentage (or other measure) of each feature that the algorithm should include in the reserve system. I applied the simulated annealing (Kirkpatrick et al. 1983) option within Marxan. This uses iterative improvement while accepting some bad moves in initial iterations in order to avoid eventual sub-optimal results (Ball & Possingham 2000).

The objective function Marxan uses is as follows:

Total score = $\sum$ planning unit cost + (boundary length modifier * $\sum$ boundary cost) + feature penalty

The score is a measure of the overall cost of the reserve system. As the formula indicates, there are three components to calculating the score. The first component is the sum of costs of the selected planning units. This measure can be monetary cost of land acquisition, anticipated management costs, lost opportunities or any other cost (monetary or otherwise). In the marine environment, area is most commonly used as the cost and this is the measure I applied.

The second component of the score is a measure of the perimeter of the selected planning units. It is calculated by multiplying a boundary modifier by the boundary cost. The boundary modifier weights the importance of minimizing the boundary cost. The boundary cost is usually measured as the length of the boundaries (*e.g.,* in meters or kilometers). This component is used to control the compactness of the reserve network. I set the boundary length modifier at a relatively high level (3) to create somewhat compact reserves. I determined this value through trial and error until the outputs consisted of a variety of reserve sizes that were relatively compact and resulted in individual reserves rather than long connecting corridors. Determining this input into Marxan is by nature arbitrary. I used fewer and bigger reserves, rather than less compact reserves, because they are more practical from a management perspective as they are more easily patrolled and monitored. This does not preclude the possibility of smaller reserves also being important. For my analysis, setting the same boundary modifier for all analyses was important; the specific boundary modifier value used was inconsequential.

The third component of the score, feature penalty, is the cost imposed for failing to meet conservation targets. For example, if one species is not represented at the target within the Marxan result, then this penalty is added to the score. To determine the penalty factor, I used trial and error to increase the factor until Marxan met all targets. I set the penalty factor to 20, the lowest number at which targets were achieved. I applied the penalty factor uniformly to all features so that they are weighted equally.

Together, these three components comprised the score. The reserve system with the lowest score is considered the best solution, also called the minimum set. I set the algorithm to restart 300 times with 1.5 million iterations per restart; these settings ensured that the decision-space was sampled adequately. To ensure that the sampling was adequate, I compared the selection frequency results – the number of times each planning unit was selected – from two identical runs until the results were very similar. Some low levels of variation were expected because of the random element of the simulated annealing approach.

Data and data quality

I used the best available data for both of my study areas (see Appendix 1). I obtained the abiotic data from the Province of British Columbia; these data consisted of depth, exposure, relief, salinity, slope, stratification, substrate and temperature classes. These datasets provide information on the composition of each of these abiotic features. In addition, I used a physical complexity data layer made available by the Living Oceans Society (Ardron 2002).

Most of the biotic data were available only as presence-absence species distributions, although some datasets indicated relative importance. The biotic data were less complete than the abiotic data, with a bias towards seabirds, marine mammals, and marine flora and a dearth of invertebrates or fish. Most metadata were labelled as “good” but an absence of information on when, how, or why the data were collected precludes direct comment on their quality. For the

Gitga'at study area, I also obtained local whale survey data. When I had two datasets for the same feature, I used the more complete set, so that features were only represented once.

I used three types of additional data or proxies to compensate for the inadequate information on invertebrates and fish: data at a very coarse continental scale, local and traditional ecological knowledge, and fisheries catch data. The National Oceanic and Atmospheric Administration's (NOAA) west coast atlas of living marine resources includes British Columbia's waters but these data are very coarse, intended to be viewed at a scale of 1:2,000,000 (NOAA 2007) whereas I used a coastline of 1:250,000 for the other datasets. The difference in scales meant that the coastlines did not match up, and the NOAA data tended to skew species distributions away from the coast. In the absence of anything better, however, I used these data as proxies in some scenarios.

The local knowledge originated from interviews undertaken by Fisheries and Oceans Canada of local experts such as fisheries officers and field biologists. The traditional knowledge data was collected by the Gitga'at First Nation to identify species-specific traditional fishing areas. My assumption in using these data was that the fishing areas represent good habitat for the respective species, and hence served as a proxy for biotic data. Traditional knowledge data were not available for the Huu-ay-aht case study. I refer to the combination of local and traditional ecological knowledge data as local ecological knowledge (LEK).

I used the fisheries catch data as proxies for species distributions because these fish distribution data were not available through fishery-independent datasets. I obtained spatial fisheries catch data from Fisheries and Oceans Canada, summarized in 4 km by 4 km or 10 km by 10 km grid cells, depending on the dataset. I assumed the catches were uniformly distributed within each grid cell.

Target scenarios

I set two levels of target for all my analyses for each feature: 1) 10% protected as adopted by the Convention on Biological Diversity (CBD 2006), and 2) 30% protected as an arbitrary mid-level estimate of the frequently recommended 20 to 50% in the scientific literature (Dahlgren & Sobel

2000; Parnell et al. 2006; Plan Development Team 1990; Roberts et al. 2003b; Stewart et al. 2007). I used uniform percentage targets to give equal importance to all of the features.

Data scenarios

To assess the influence of data inputs into the reserve selection algorithm, I used scenarios whereby I included: (1) all available data; (2) only biotic data; (3) only abiotic data. I further subdivided the scenarios that contain biotic data into those that included and excluded the three proxy datasets (very coarse species distribution data from NOAA, local and traditional knowledge, and fisheries catch data). In total I therefore had five scenarios: (1) all data including proxy datasets, (2) all data excluding proxy datasets, (3) biotic data including proxy datasets, (4) biotic data excluding proxy datasets, and (5) abiotic data only.

Within each scenario, I sequentially removed the features with the smallest geographic distributions to test the effect of these features on the results. Features that only occur in a small geographic area tend to carry considerable weight in Marxan solutions because there is little spatial flexibility in representing such features in the reserve system. I tested this effect by deleting these datasets from the input files, one by one for the abiotic and biotic data, and, for the sake of expediency, in groups of three for the scenario with all data (see Table 5.1 for the removal sequence). For the proxy datasets, I was interested in their aggregate effect. I therefore either included or excluded each of the three proxy datasets in their entirety, rather than disassociating them.

Table 5.1. Removal sequence of features after coarse data (NOAA), LEK data, and fisheries data have been removed. The feature with the smallest geographic distribution was removed first, followed by the second smallest, etc. b=biotic feature; a=abiotic

Gitga'at case study	Huu-ay-aht case study
1. Eulachon (b)	1. Sea lions (b)
2. Sea lion haul-out (b)	2. Eelgrass (b)
3. Marsh grasses (b)	3. Herring (b)
4. Transient orcas (b)	4. Kelp (b)
5. Surf grasses (b)	5. Salinity mesohaline (a)
6. Resident orcas (b)	6. Current: high (a)
7. Red bleached algae (b)	7. Sea otter (b)
8. Soft brown kelps (b)	8. Exposure: low (a)
9. Dark brown kelps (b)	9. Depth: shallow (a)
10. Humpback survey (b)	10. Exposure: moderate (a)
11. Sponge reefs (b)	11. Harbour porpoise (b)
12. Geese (b)	12. Steller sea lion (b)
13. Eelgrass (b)	13. Blue heron (b)
14. Herring (b)	14. Stratification: stratified (a)
15. Gray whale (b)	15. Dall’s porpoise (b)
16. Diving ducks (b)	16. Northern fur seal (b)
17. Shorebirds (b)	17. Geese (b)
18. Fulmar (b)	18. Dabbling ducks (b)
19. Other pelagic birds (b)	19. Depth deep (a)
20. Gulls (b)	20. Temperature warm (a)
21. Current: high (a)	21. Slope sloping (a)
22. Alcids (b)	22. Eagles (b)
23. Depth: shallow (a)	23. California sea lions (b)
24. Harbor porpoise (b)	24. Black oystercatcher (b)
25. Marbled murrelets (b)	25. Depth: photic (a)
26. Eagles (b)	26. Complexity (a)
27. Kelp (b)	27. Alcids (b)
28. Temperature: warm (a)	28. Stratification weakly mixed (a)
29. Other waterfowl (b)	29. Gray whale (b)
30. Pacific white-sided dolphin (b)	30. Diving ducks (b)
31. Complexity (a)	31. Shorebirds (b)
32. Salinity: polyhaline (a)	
33. Relief: high (a)	

I used hierarchical clustering (JMP 2007 Version 7) to analyze the differences in the results of the data scenarios. I used the selection frequency output from Marxan as the basis of comparison. This output measures how many times out of the 300 restarts each planning unit was selected, and is often interpreted as a measure of importance or irreplaceability (Ball & Possingham 2000; Grand et al. 2007). I chose hierarchical clustering as the analysis method because it allows for the comparison of all scenarios to each other, and it has previously been applied in studies that have used Marxan (Airamé et al. 2003).

To gauge how well the different types of data served as surrogates for each other, I compared the minimum set (the result for each scenario that met targets for all features at the lowest cost) for each of the five data scenarios to a random solution and to each other. I generated the random solution by generating one feature that covers the entire study area equally, then running Marxan with the same settings as for the other scenarios. This was repeated for the 10% and 30% targets. I then assessed how many of the features used in the other scenarios are captured in the random solution.

Results

Scenarios

The maps depicting the five data scenarios revealed that there are clear patterns of more and less important areas (Figure 5.2). From visual inspection, it appeared that the 10% and 30% target scenarios showed similar but not identical patterns. Likewise the maps of the scenarios with all data were similar in the pattern of the selection frequency to those with all biotic data (including proxies) and those without proxies. The scenarios with only abiotic data highlighted fewer important areas. The contrast of areas of higher and lower importance was more apparent in the Gitga'at than in the Huu-ay-aht study area.

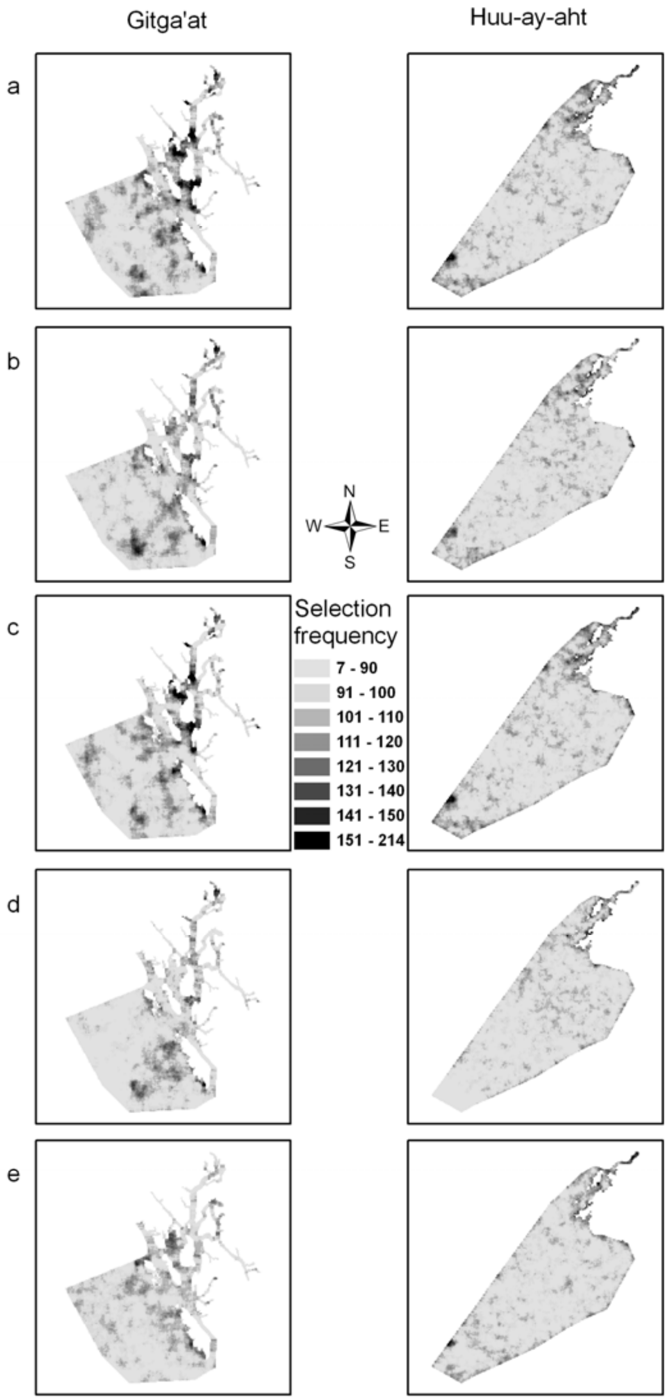

Figure 5.2. Selection frequency maps of five data scenarios for both study areas: a) all data, b) all data with proxy datasets excluded, c) all biotic data, including proxy datasets d) all biotic data excluding proxy datasets, e) abiotic data. The legend is out of a possible 300 restarts. The 10% and 30% target scenarios yielded similar results, so only the 30% target scenario is portrayed here.

No planning units were selected all of the time (*i.e.*,300 times out of 300 restarts). The most a planning unit was selected over 300 runs was 62% in the 10% target scenario (Huu-ay-aht study area), and 71% in the 30% target scenario (Gitga'at study area). The mean selection frequency of planning units remained quite consistent among the scenarios, as did the standard deviations.

All data type scenarios resulted in a higher percentage of conservation features represented than the randomly selected reserve system, with similar patterns for both the 10% and 30% target scenarios (Table 5.2). Using biotic data, including proxies, as the only features driving the selection algorithm resulted in meeting the targets for more than 90% of all features. When proxies were excluded, however, the biotic data had a more mixed performance, though usually slightly higher (65-80%) than with only abiotic data (58 to 87% of other features).

Table 5.2. Effectiveness of different data types at achieving targets. The percentages show the proportion of all targets met with the best solution when only the data types in the left column were included in the runs.

	Targets met (out of 117 for Gitga'at, 88 for Huu-ay-aht study area) for two target scenarios.			
	Gitga'at	Huu-ay-aht	Gitga'at	Huu-ay-aht
Scenarios: data types included in the run	10%	10%	30%	30%
Randomly selected	26.5%	38.6%	44.4%	46.6%
All abiotic and biotic data	100%	100%	100%	100%
All, excluding proxies	80.3%	81.8%	89.7%	84.1%
Biotic data, including proxies	92.3%	93.2%	94.9%	96.6%
Biotic data, excluding proxies	77.8%	64.8%	76.9%	79.5%
Abiotic data	58.1%	88.6%	76.1%	85.2%

The results of the hierarchical cluster analysis were similar for the two case studies. In the Gitga'at study area, the 10% and 30% target scenarios clustered in a similar, but not identical, fashion. Six clusters emerged (Figure 5.3a), described here in order of the most to the least data included. First, scenarios where all data were included were similar to those where only biotic data were included. This pattern continued when NOAA data were removed, and for the 30% scenario these runs were also similar to those where fisheries data were removed. For the 10% target, the biotic scenario with NOAA and fisheries data excluded clusters with this group. Second, scenarios with all data but NOAA, fisheries, and LEK data, and up to 15 of the most spatially constrained features removed were similar to each other. Third, runs with biotic data only, but not NOAA, fisheries and LEK data removed clustered with runs where up to 15 (10% target scenario) or 18 (30% target scenario) spatially constrained biotic features were removed. Fourth, biotic data scenarios with many features removed (between 16 to 23 for the 10% target scenario, and 20 to 26 for the 30% target scenario) clustered with each other. Fifth, runs with abiotic data only were most similar to those with all data where 18 to 30 of the features had been removed (depending on the target), most of them biotic. Finally, in both target scenarios there were two runs with most features excluded that clustered with each other but not other runs.

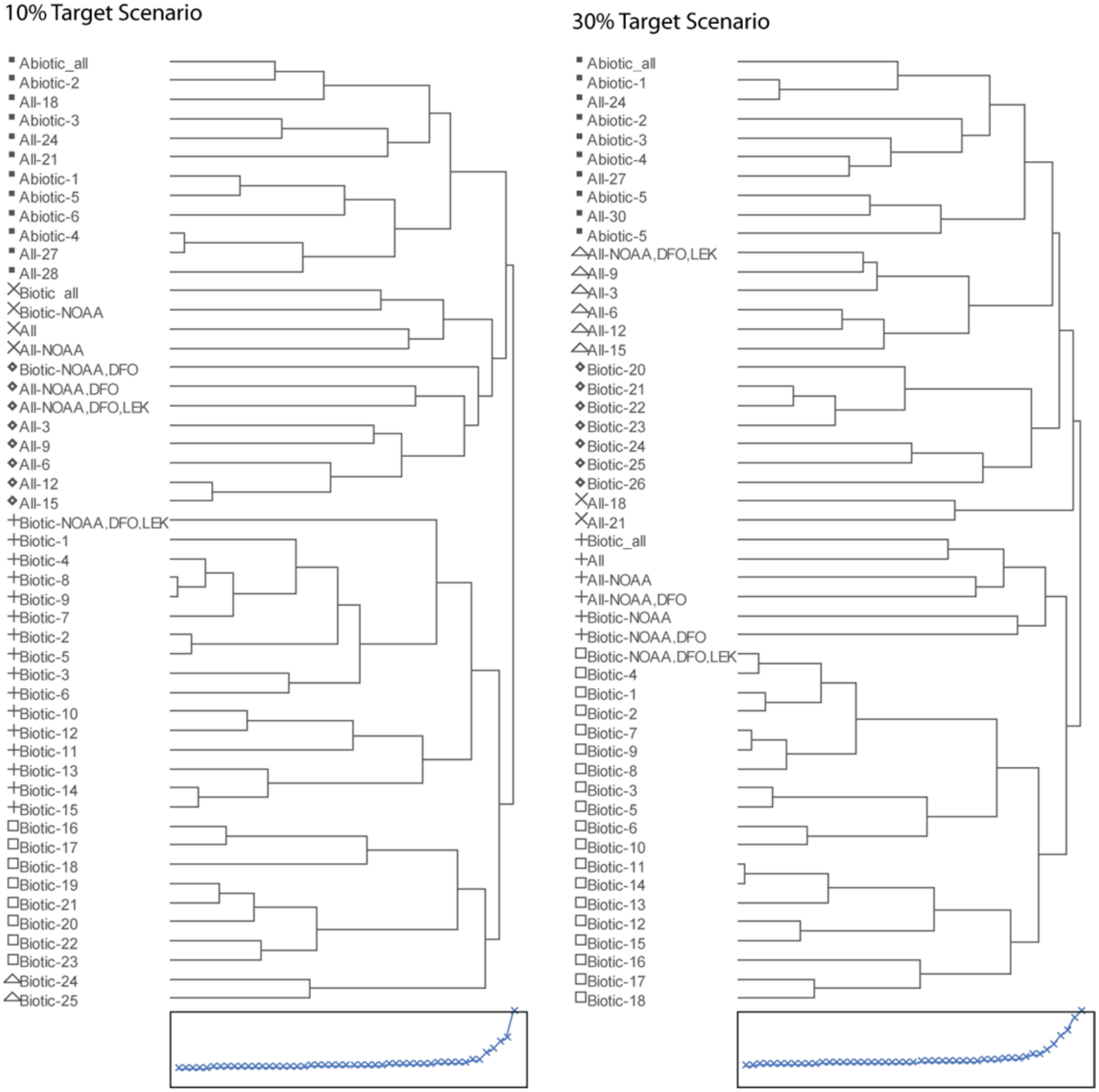

Figure 5.3a: Comparison of the selection frequency of different scenarios using hierarchical cluster analysis of (a) the Gitga'at study area. The abbreviations used in the descriptors are as follows: all = all datasets; abiotic = abiotic data, biotic = biotic data. What follows after the minus sign indicates the type or number of datasets removed from that scenario, with the proxy datasets also removed. The symbols on the left indicate the clusters. A distance graph is shown at the bottom of each of the dendrograms, indicating that most of the variation is explained by the cluster groupings.

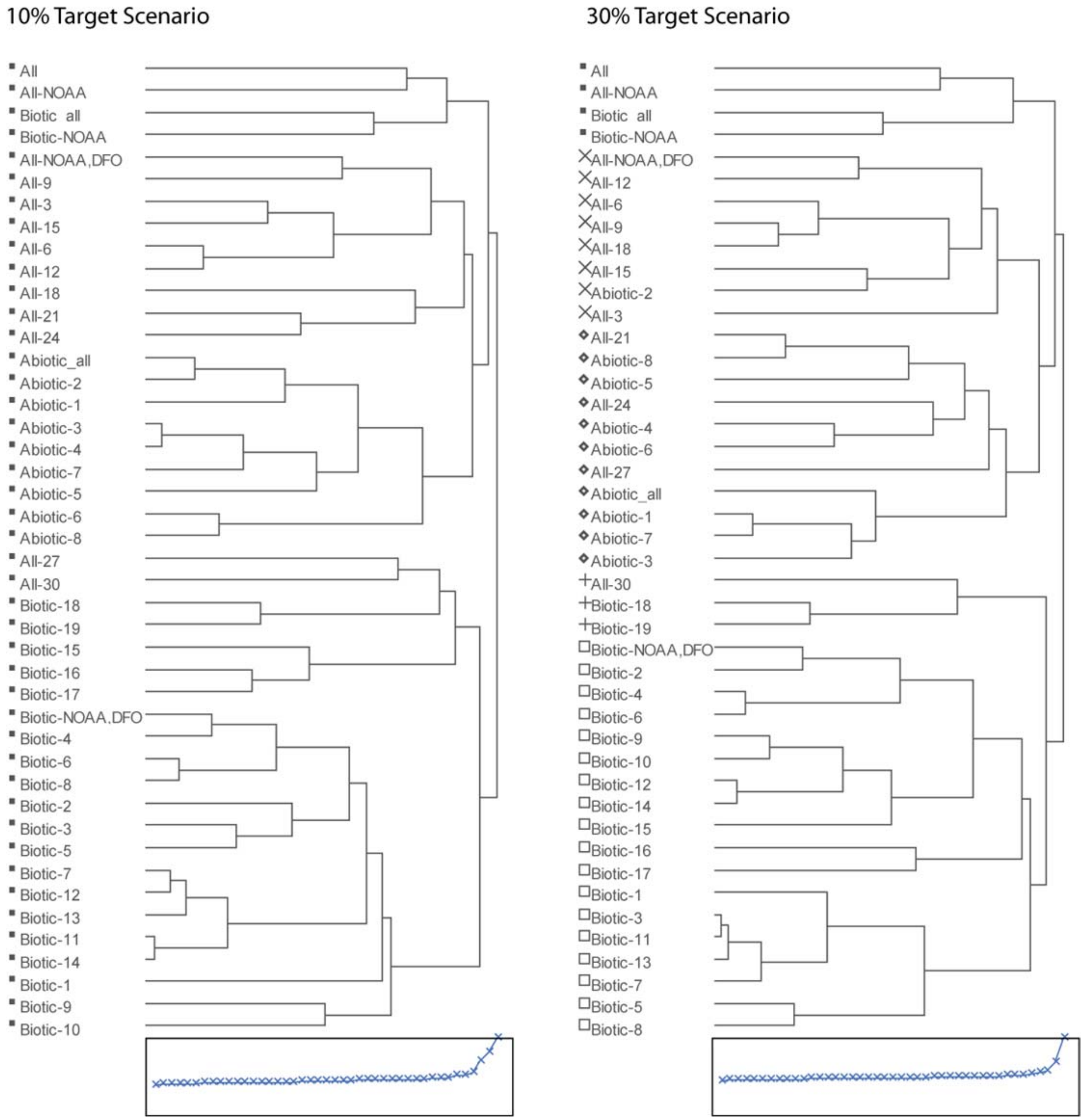

Figure 5.3b: Comparison of the selection frequency of different scenarios using hierarchical cluster analysis of (b) the Huu-ay-aht study area. The abbreviations used in the descriptors are as follows: all = all datasets; abiotic = abiotic data, biotic = biotic data. What follows after the minus sign indicates the type or number of datasets removed from that scenario, with the proxy datasets also removed. The symbols on the left indicate the clusters. A distance graph is shown at the bottom of each of the dendrograms, indicating that most of the variation is explained by the cluster groupings.

The clustering patterns were very similar for the Huu-ay-aht study area, where five groupings emerged (Figure 3b). First, scenarios where all data were included were similar to those where only biotic data were included. This held true when NOAA data were removed. Second, scenarios with all data but NOAA and fisheries data, and up to 24 of the most spatially constrained features removed were similar to each other. Third, runs with biotic data only, but not NOAA or fisheries data, clustered with each other. Fourth, runs with abiotic data clustered together in the 10% target scenario, and included runs with all data where most (21 to 27) of the biotic features had been removed. Finally, the scenarios with many features removed clustered together.

Discussion

I found that removing some datasets from the analyses did not significantly influence the spatial pattern of results. This indicates that waiting to have complete data before using a tool such as Marxan to assist with selecting reserves will not produce results that are very different from using slightly fewer datasets. My analyses also showed that it is best to use a combination of biotic and abiotic data, if representation of biodiversity is the goal.

Biotic data (including proxies) served as a better surrogate for abiotic data in achieving conservation objectives than vice versa, and both were better than using random data to selected reserves. When proxy datasets were excluded from the biotic scenarios, the performance of biotic data in representing abiotic features was mixed. When only abiotic features were included, the result was better than using random data, but captured between 58 and 89 percent of features. Small differences in the results of the two case studies are most likely explained by the variations in the features (not all features are found in both case studies) and their geographic distributions.

Many possible reserve combinations could achieve targets in the case studies, as illustrated by the fact that few planning units occurred in a great proportion of the solutions. From a planning perspective, this flexibility in the scientific results means that there is room for social factors (political, economic, cultural, and others) to play a role in determining the reserve network while still achieving ecological objectives.

I found that there are diminishing returns for including additional data. The selection method benefited from a diversity of types of information but was robust to some missing data, as evident in the cluster analysis results. Inclusion of different data types (*i.e.,* abiotic, biotic, proxies) altered the spatial pattern of the selection frequency of planning units, but excluding particular features from the analysis had little effect. If I assume that the best result is the one that includes the data in which I have the most confidence – *i.e.*, abiotic and biotic data, but not the coarse NOAA data, LEK, or fisheries data – then many features (up to 15 for Gitga'at study area, up to 24 for the Huu-ay-aht study area) could be removed before solutions clustered differently. In other words, including most of the abiotic data, which tended to have broader geographic distributions, with 11 (Huu-ay-aht) or 12 (Gitga'at) biotic features produced about the same results as using all abiotic and biotic data.

Excluding the most geographically restricted species or habitats did not have as much of an affect as I had anticipated. Because there is less flexibility in representing such species or habitats in a reserve system, the areas in which they occur can have an influence on the selection frequency pattern. I found that removing such features did not result in a significantly different selection frequency pattern until several such features had been removed. The implication of this finding is that decision makers need not wait for additional datasets to become available in order to proceed with reserve selection.

Scenarios that include local and traditional ecological knowledge (LEK) as a proxy for biotic data were different from others. LEK was only available for the Gitga'at study area, where both scenarios that included LEK data – all features, and biotic features – were similar to each other. The LEK data were more geographically limited than some other types of data inputs, and hence drove the solutions the algorithm selected. The Gitga'at traditional knowledge data, part of the umbrella of the LEK data in this scenario, were fisheries-dependent; they represented fishing areas for those species. Because these fishing areas are geographically concentrated, they appear with a high selection frequency in the Marxan results.

Abiotic data were important tools in generating the solutions but cannot, by themselves, represent biological features. It is tempting to use features such as basic bathymetry and exposure because they use are often the most readily available type of information (Airamé et al. 2003; Ward et al. 1999). Satellite imagery and remote sensing may also provide additional abiotic data that are relatively easily accessible. However, my results suggested that using abiotic data with fewer than 10 biotic features produced different solutions than when more biotic data were included. This is a concern because past studies have relied on such physical data alone to capture biological features (*e.g.*, Airamé et al. 2003). I acknowledge, however, that the biotic data I included were biased towards certain taxa, and that it would be useful to assess abiotic data against more balanced biotic data

The two case studies I examined in this study exhibited remarkably similar results. Located hundreds of kilometers apart, the study areas are affected by some different processes, and might therefore also have different biodiversity patterns. The Gitga'at study area contains many inland sections with long and deep fjords and channels. The Huu-ay-aht study areas experiences much more exposed weather. The study areas nevertheless exhibited very similar clustering patterns when datasets were removed, and showed very similar patterns in the effectiveness of surrogates. This provides some indication that such patterns may also be seen in other areas.

My findings are similar to other studies of surrogates. For example, Ward et al. (1999) considered biological data as better surrogates than habitat categories at low target levels. Ward et al., who used much more complete biological data, also found that habitat categories are better surrogates at high representation levels. I did not test high representation levels (40 to 80%) in my study, because I placed emphasis on more realistic percentage targets. Beger et al. (2007) examined the effectiveness of taxa in coral reef systems as surrogates for other taxa. They found that no taxonomic group was a reliable surrogate for the other groups (Beger et al. 2007). While I did not test the effectiveness of specific taxa as surrogates, the general conclusion is the same: That it is best to include the features of interest for protection if data for them exist.

My study could give some sense of the levels of data necessary for robust results from decision support tools, although the specific findings represent the species and habitat distributions and associations of the temperate waters of British Columbia. The fact that the study areas can be considered data-rich should be seen as a reflection of the general dearth of marine biodiversity data. The data I used have many problems (*e.g.,* incomplete metadata, biases towards marine mammals and seabirds), yet the datasets are more numerous than in most marine regions of the world, especially in developing countries.

Given our limited knowledge of most marine systems, the best that we can do at the moment is to compare amongst imperfect solutions to the reserve selection problem. At present, we do not have the capability to produce a single correct answer of where marine reserves should be placed. It is important to note that the only measure of robustness that I used in my study was whether the spatial patterns of the selection frequency changed. In ecosystems where more data exist, and much is known about the processes that drive biodiversity, other measures of robustness could be used, such as connectivity of the reserve systems, and potential persistence of species within reserve systems. My analyses are therefore comparisons of imperfect answers.

Conclusion

My findings provide guidance and reassurance for managers needing to make MPA design decisions using typically available data. Because scenarios that included all data resulted in different patterns of clustering from abiotic or biotic data alone, it is best to include both of these types of data if available. Biological data served as a better proxy for physical data than vice versa. Therefore if some basic physical data are already available, as would be the case for any area where nautical charts exist, efforts should be placed on generating biological data rather than improving physical data. If only physical data are available, targets should be set higher to improve the chances that biological features are represented. Where resources are not available for biological surveys, I speculate that fisheries independent LEK data may serve as a suitable proxy. I did not have such fisheries independent data, and was unable to test this assumption. The removal of some data layers did not, however, result in distinct clustering patterns, indicating that the lack of some data is unlikely to change the results significantly.

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6. Comparing and integrating community-based and science-based approaches in prioritizing marine areas for protection[6]

[6] A version of this chapter has been submitted for publication: Ban, N.C., C. Picard, A.C.J. Vincent. Comparing and integrating community-based and science-based approaches in prioritizing marine areas for protection.

Introduction

Marine protected areas (MPAs) can help address the problem of declining biodiversity in the oceans (Sala & Knowlton 2006), changing food webs (Pauly et al. 1998), collapsing fisheries (Jackson et al. 2001), and pervasive human impacts (Halpern et al. 2008). Indeed, there have been repeated calls for establishing MPAs to address these threats (CBD 2006). Yet MPAs currently cover less than one percent of the ocean and progress towards establishing new MPAs is slow (Wood et al. 2007).

It is generally acknowledged that both social and ecological variables affect MPA success (Christie 2004; Klein et al. In Press; Roberts et al. 2003). From a social perspective, support from and compliance by people affected by MPA designation is crucial to enabling its success (Christie 2004; Walmsley & White 2003). Indeed, creation of most existing protected areas was dominated by social, economic and/ or political issues (Davis 2005; Knight & Cowling 2007). Conservation biology, however, tends to refer to the establishment of MPAs by communities as "ad hoc" or "opportunistic" (Pressey 1994; Stewart et al. 2003) while highlighting systematic planning as the most efficient method of protected area selection (Margules & Pressey 2000; Sarkar et al. 2006; Stewart et al. 2007). In theory, the latter will better prioritize areas for protection that are representative of biodiversity and ensure the persistence of species and habitats (Margules & Pressey 2000; Sarkar et al. 2006). However, some conservation biology literature is now embracing community initiatives as a means of establishing protected areas (Knight & Cowling 2007; Knight et al. 2006).

Both approaches to MPA establishment – community-based and science-based – have their limitations (Ban 2008). The former leads to MPAs that may or may not capture essential aspects of biodiversity, in an unpredictable fashion. Communities have a variety of motivations for wanting a site protected, ranging from ecological (*e.g.*, protection of a spawning ground) to social (*e.g.*, prevention of incompatible uses). Indeed, the MPAs established by communities may still be the subject of controversy and conflict. Science-based approaches, however, can depend to a crippling extent on obtaining high quantity and quality of biodiversity data, or surrogates thereof (Sarkar et al. 2006; Ward et al. 1999). Because very little is known about the spatial distribution of marine species in most marine regions, limited data availability can lead to conservation paralysis, with a focus instead on collecting more data or undertaking more analyses instead of implementing the MPA (Knight et al. 2006). Moreover, implementation and compliance are uncertain without involvement of user groups who would be affected by MPA designation (Byers & Noonburg 2007).

It should be possible to reconcile community-based and science-based approaches by taking a systematic approach to MPA planning, designed to ensure the persistence of all levels of biodiversity (Margules & Pressey 2000; Pressey et al. 2007). An emerging trend is to establish sets of MPAs, rather than the single sites that have hitherto dominated. Decision support tools are increasingly being used to facilitate the selection of areas that are representative of surrogates of biodiversity (Airamé et al. 2003; Fernandes et al. 2005; Klein et al. in press). Many such ventures are considering and integrating preferences of user groups and other stakeholders. While encouraging in some respects, such integration of ecological and socioeconomic-political criteria *ab initio* makes it difficult to understand the relative roles of these approaches (but see Gonzales et al. 2003).

We set out to compare and contrast social and biophysical approaches in MPA prioritization, drawing on community opinions and feedback and on abiotic and biotic data. Our approach differs from others in that we incorporated community preferences for MPAs, rather than treating their uses as a cost. We also tested the effect of using socioeconomic data in a science-based approach, by varying the cost of protection to incorporate commercial fisheries and human impacts.

We carried out our study in British Columbia, Canada, focusing on marine areas of the Gitga'at First Nation and the Huu-ay-aht First Nation as our case studies. First Nations – as indigenous groups are called in British Columbia – are especially important when considering marine conservation issues, because they have a constitutional right to fish for food, social and ceremonial purposes. The people whom we interviewed sought recovery of depleted marine species and sustainable food fisheries for current and future generations. In line with their expressed preferences (Ban et al. 2008), we here focus on designating areas that would exclude commercial and recreational fishing, while allowing for indigenous fishing for food, social and ceremonial purposes.

First Nations are also becoming increasingly proactive in planning their marine areas, not least because marine resources are an integral part of indigenous cultures in coastal Canada (Garibaldi & Turner 2004; Turner et al. 2000). Moreover, recent legal decisions affirmed that First Nations need to be consulted and accommodated in issues affecting their rights and title (Fisheries and Oceans Canada 2008). Other user groups (*e.g.,* commercial fishermen, recreational fishermen, tourism interests, transportation sector, aquaculture) are also very important in marine planning but are outside the scope of our study.

Given the sociopolitical context for our work, we need to emphasize that our study was academic in nature; the research was developed in partnership with the First Nations involved, conducted with their support and approval, and then fed back to the First Nations for their use. It will only be used for planning purposes if the partners decide to do so.

Methods

Study sites

The case studies are located on the west coast of Canada, in the temperate marine waters of British Columbia. The traditional territory of the Gitga'at First Nation (one of the Tsimshian First Nations) is located on the north coast of British Columbia, in a remote region accessible only by boat or float plane. The community of Hartley Bay is the only permanent settlement in the area. The Huu-ay-aht First Nation is one of the Nuu-chah-nulth Nations on the west coast of Vancouver Island. The main Huu-ay-aht community is Anacla, which is bordered by the non-First Nations town of Bamfield, and their territory encompasses part of Pacific Rim National Park Reserve. The towns of Anacla and Bamfield are accessible by an unpaved logging road. Both indigenous groups are comprised of approximately 600 members, with 200 of those living in towns within their respective areas.

Community-based approach

We carried out semi-structured interviews with marine resource users in the two study areas, and held community meetings to receive feedback (Ban et al. 2008). The aim of the interviews was to identify individual preferences for long-term goals in the marine environment, possible areas for protection, and envisaged levels of protection. In the Gitga'at study area we conducted

20 individual interviews, and in the Huu-ay-aht study area we conducted 19, all with the support of the communities. Our interviews focused on former or present marine resource users and spanned a range of ages (including elders) and representation from the communities' traditional clans. Participants drew their preferences on nautical charts of the study areas, with which they were familiar and comfortable. We then digitized, summarized and presented these maps at community meetings to ascertain the level of agreement amongst those present. The invitation to community meetings was extended to all community members. Attendance ranged from 35 (Gitga'at study area) to 10 (Huu-ay-aht study area) individuals - while the Huu-ay-aht attendance appears small, it still comprised some 10% of the adult population of Anacla. We used feedback forms and discussions during these community meetings to obtain a map of community-preferred MPAs. We held three community meetings in the Gitga'at territory: Two to review the results of the individual interviews (one in Hartley Bay (n=17) and one in the small city of Prince Rupert (n=10)), and one to receive feedback on the science-based and integrated maps (n=35). We held one community meeting in the Huu-ay-aht territory, reviewing the interview results, and science-based and integrated maps in the same meeting. See Ban et al. (2008) for a detailed analysis of the interviews and community meetings.

Science-based approach

We used the decision support tool Marxan to seek systematic prioritization of potential protected areas in the study locations (Ball & Possingham 2000; Possingham et al. 2000). Marxan is a software tool that helps to select protected areas that meet biodiversity targets. Biodiversity features can be species, sub-species, populations, habitats, physical features, or anything else the user wishes to set as a biodiversity target or surrogate. The study area is divided into planning units, which are then populated with the biodiversity features. Marxan uses a simulated annealing algorithm to identify potential MPAs. It uses the information contained within each planning unit to pick a collection of planning units that achieve biodiversity targets while minimizing their cost (Possingham et al. 2000). A component of Marxan called the boundary length modifier can be used to set the compactness of the protected areas selected. Marxan can be run many times, with each run producing a potential MPA system. Marxan also keeps track of the importance of each planning unit, by counting how many times each planning unit is chosen in solutions. This result, called the selection frequency, is a measure of the conservation value (Carwardine et al. 2007). The "cost" is minimized by

Marxan. The cost can be any number of spatially explicit measures, monetary or not. Most commonly, area is used as a cost, but it could also be foregone revenues, the cost of management and enforcement, human impacts, fishing effort, etc.

We used Marxan to select areas that would exclude commercial and recreational fishing, but would allow indigenous fishing. We set two target scenarios to represent biodiversity: (1) protecting 10% of all features, the recommended target in the Convention on Biological Diversity (CBD 2006), and (2) protecting 30% of all features, a mid-level estimate of the frequently recommended 20 to 50% in the scientific literature (Stewart et al. 2007). We used the best available spatial data for our conservation features (species and habitats targeted for conservation). These data included species distributions, physical measures, and local and traditional knowledge (Appendix 1).

We ran Marxan with 300 repeats at 1.5 million iterations for each repeat. Each of the 300 repeats provided a protected area scenario that achieved the target. We adjusted the boundary length modifier to create results that resembled the range of sizes of potential MPAs chosen by the communities. We used Marxan the way it is commonly used, with percentage targets for all features for which data were available. Because of data and knowledge gaps, we did not incorporate direct measures of connectivity or persistence. The difficulty of incorporating connectivity and persistence is acknowledged in the literature, and some advances are being made to improve selection methods (Nicholson et al. 2006; Pressey et al. 2007).

Preparing the community-based and science-based approaches to prioritization for comparison

We assessed the compatibility of the two prioritization approaches in three ways. First, we examined the spatial overlap using spatial overlap measures (Fielding & Bell 1997). To do this, we divided the maps of both approaches into three categories. For the community-based approach, we divided the prioritization map into the following categories: (1) areas not selected, (2) all areas selected at least once during interviews except for those that fell into the third category, and (3) areas on the map that resulted from the community meetings where we reviewed the results of the individual interviews. For the science-based approach, we used the selection frequency of each planning unit as our metric of conservation importance. We categorized each of the selection frequency maps using standard deviations: (1) those selected at a rate similar to or less than chance (one standard deviation to the right of the mean), (2) selected slightly more than chance (next standard deviation), and (3) selected a lot more than chance (the rest of the tail). Thus both approaches had three categories to be compared to each other.

Second, we assessed the percentage of species and habitats captured by the community-preferred areas. We used geographic information systems software (ArcMap 9.1) to carry out the gap analysis. We also calculated the area contained within each of the prioritizations.

Third, we held community meetings to show two science-based prioritization maps (see below) to community members. We asked for their opinion of the ecological importance and social acceptability of those areas. At these meetings we asked two questions: (1) Does the science-based prioritization map highlight most of the important conservation areas?; (2) Would you agree with having those areas protected? We received their opinion through feedback forms that were completed prior to discussions, and through subsequent exchanges.

Integration scenarios

We combined the community-based and science-based prioritization approaches in Marxan. To include the community-preferred areas in the Marxan scenario, we locked in those areas while augmenting them with biotic and abiotic data to achieve the conservation objectives using the same settings in Marxan as the other scenarios. We showed two aspects of the results of the integrated approach at the community meetings, and received feedback as described above. The maps we showed were the science-based Marxan result for the 10 percent target, and the Marxan results of the integration scenario for the 10 percent target. Both of these maps showed the selection frequency, as well as an example of a set of protected areas picked by Marxan. We further used the cost option in Marxan to integrate other socioeconomic variables, running scenarios with both human impacts and commercial fisheries. We then compared the selection frequency output of these two cost scenarios to the Marxan base scenario which used only area as the cost, and to the community-preferred areas.

For the cost scenario related to human impacts, we applied the version of the model where stressors extended up to 25 km from the source (see Ban & Alder 2008). Planning units that were more impacted by human activities were given a higher cost in Marxan, thus reducing their chances of being picked. This directed protection towards less impacted areas, which are also less used by humans, and should therefore minimize displacement of human activities in potential MPAs.

For a second cost scenario, we combined spatial commercial catch data for twelve fisheries in the study areas; spatial data did not exist for other commercial fisheries, or for recreational fisheries. We created a relative importance index for each fishery, ranging from one (least important) to five (most important) using natural breaks in ArcGIS 9.1. We then added the relative importance fields to create one cost map of commercial fisheries. Because Marxan tries to avoid areas of higher cost, its selections were directed away from areas of higher importance to commercial fisheries.

Methods for comparing community-based, science-based, and integrated approaches

To compare the scenarios, we twice carried out overlap assessments, initially using the community-based map as the basis of comparison to compare to the science-based map, and then the science-based map to compare to the community-based map. First, we assessed the overall convergence of the map used as the basis of comparison to the other map as outlined in a previous section. The overlap assessment gives the number of convergently classified planning units, divided by the total number of planning units.

Second, we calculated the Cohen's kappa statistic. The kappa statistic is a chance-corrected model of "accuracy" (here referred to as convergence or overlap), based on the agreement between predicted and observed values (here we represent the "observed" by the map used as the basis of comparison), and the chance agreement between each classification (Fielding & Bell 1997). The P-value from the Cohen's statistic reflects the probability that the model performs better than random chance at predicting category classes.

Finally, we examined the overlap of the third category by itself (the community-preferred areas for the community-based scenario, and the most frequently selected planning units for the science-based scenarios), which we refer to as the conservation category. This overlap measure gives the proportion of convergently classified areas for this category, divided by the total number of planning units containing the conservation category. We calculated these statistics using an extension to ArcView 3.2 (Jenness & Wynne 2007).

Results

Comparing community-based and science-based prioritization maps

The maps summarizing the community-based and science-based approaches showed many commonalities (Figure 6.1). In the Gitga'at study area, similar locations were highlighted in inshore areas in the community-based and science-based methods. The offshore areas were not selected during interviews or in community meetings, although some of these areas appeared to be as important as inshore areas in science-based scenarios. Fewer patterns were discernable in the Huu-ay-aht study area. In the science-based scenario, inshore areas were generally more frequently selected than offshore areas. The same applied to community-based and integration scenarios. One of the community-preferred areas appeared as an area of high conservation importance using the science-based approach, but another area did not. Only one offshore area was selected during interviews.

The overlap assessments comparing the community-preferred and science-based maps revealed that the two were statistically significant predictors of each other in both study areas at about 70% (Table 6.1). The kappa statistic was higher for the Gitga'at case study area than for the Huu-ay-aht case study, indicating that the community-preferred and science-based scenarios were more similar to each other in the former. This was also apparent when visually comparing the maps (Figure 6.1). The similarity of the conservation category – category 3 in our comparisons – was also better in the Gitga'at study area than in the Huu-ay-aht area.

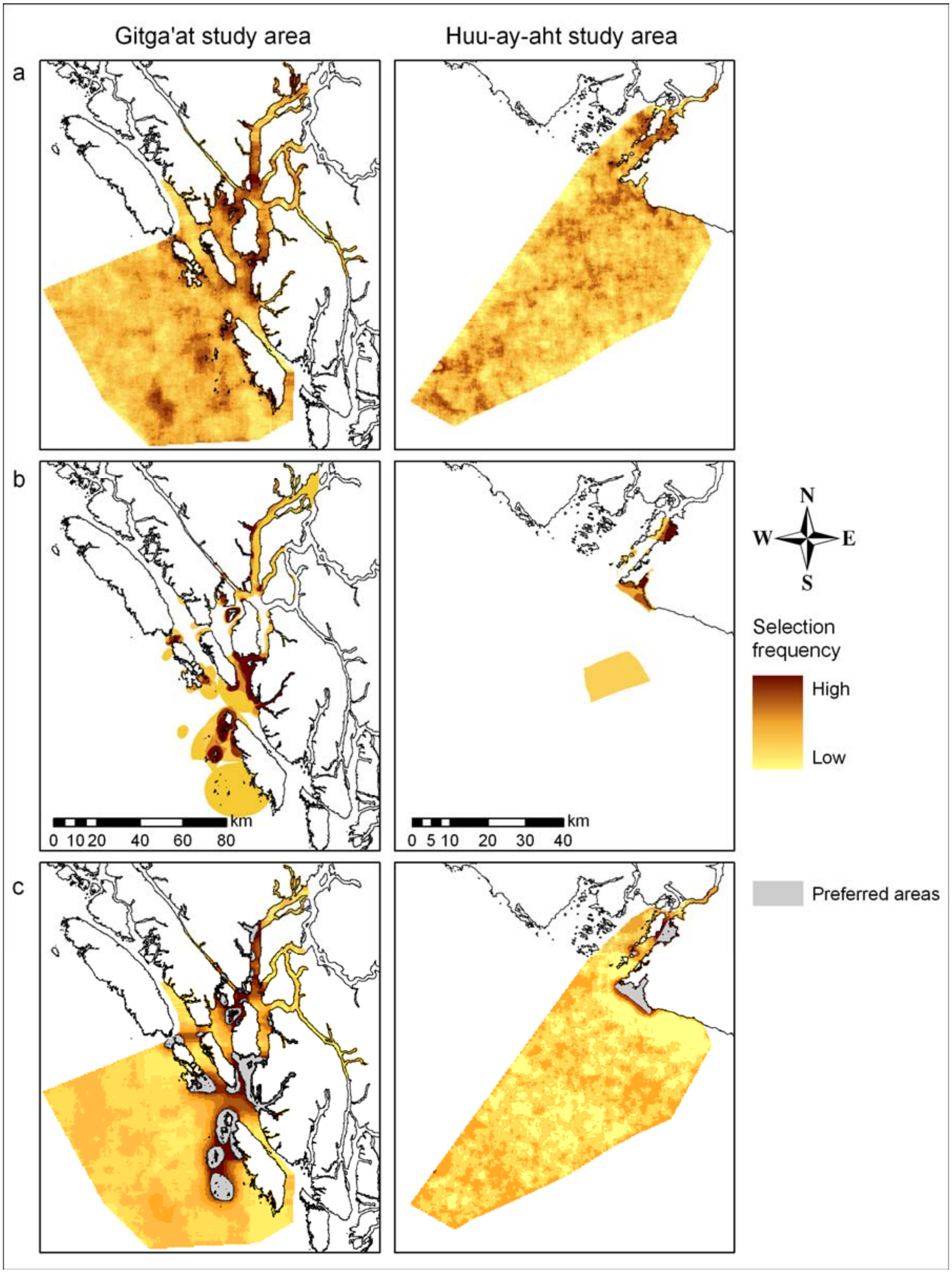

Figure 6.1. Maps comparing community-based and science-based prioritization. Areas of lighter color are less frequently selected, dark colors more frequently selected. Left panels show the Gitga'at study area, the right panels the Huu-ay-aht study area. "a" depicts the result of the science-based prioritization using Marxan, showing the 10% target scenarios. "b" shows the overlays of community preferences from individual interviews. "c" represents the combination of the two previous options, using Marxan with the community preferences (grey areas) locked into the scenarios.

Table 6.1. Spatial overlap assessments of conservation prioritization scenarios in two case studies, British Columbia, Canada.

	Gitga'at study area							
	10% target scenario				30% target scenario			
Comparison of the community-preferred scenario to these scenarios:	Overall accuracy	Kappa statistic ($\hat{K}$)	Signif. of $\hat{K}$	Accuracy of conservation category	Overall accuracy	Kappa statistic ($\hat{K}$)	Signif. of $\hat{K}$	Accuracy of conservation category
Science-based scenario	0.70	0.21	< 0.001	0.26	0.71	0.23	< 0.001	0.30
Human stressors as the cost	0.57	-0.01	0.82	0.06	0.69	0.20	< 0.001	0.23
Commercial fisheries as the cost	0.49	-0.21	1.00	0.00	0.49	-0.16	1.00	0.04
Comparison of the science-based scenario to these scenarios:								
Community-preferred scenario	0.70	0.21	< 0.001	0.27	0.71	0.23	< 0.001	0.29
Human stressors as the cost	0.63	0.14	< 0.001	0.11	0.77	0.39	< 0.001	0.47
Commercial fisheries as the cost	0.55	-0.07	1.00	0.03	0.56	-0.01	0.97	0.19

	Huu-ay-aht study area							
	10% target scenario				30% target scenario			
Comparison of the community-preferred scenario to these scenarios:	Overall accuracy	Kappa statistic ($\hat{K}$)	Signif. of $\hat{K}$	Accuracy of conservation category	Overall accuracy	Kappa statistic ($\hat{K}$)	Signif. of $\hat{K}$	Accuracy of conservation category
Science-based scenario	0.72	0.11	< 0.001	0.34	0.73	0.07	< 0.001	0.18
Human stressors as the cost	0.72	0.00	0.73	0.00	0.73	-0.02	< 0.001	0.11
Commercial fisheries as the cost	0.74	-0.02	1.00	0.04	0.74	0.05	1.00	0.21
Comparison of the science-based scenario to these scenarios:								
Community-preferred scenario	0.72	0.11	< 0.001	0.11	0.73	0.07	< 0.001	0.06
Human stressors as the cost	0.64	0.09	< 0.001	0.11	0.70	0.18	< 0.001	0.30
Commercial fisheries as the cost	0.63	0.05	< 0.001	0.06	0.68	0.15	< 0.001	0.31

Community preference embraced important ecological features. The community-preferred areas represented more biodiversity features than would be expected given their size. In the Gitga'at study area, the community-preferred areas covered 10.5 percent of the planning units, and contained 75 out of 79 features. The mean portion of each feature represented within the community-preferred areas was 34 percent. Eighty-seven percent of the features had more than five percent of their component features captured within these areas (Figure 6.2). The features that were not represented at all were primarily those that occur in offshore areas. The Huu-ay-aht-preferred areas, on the other hand, did not represent as many features. The average representation of each feature was 6.6 percent, with the preferred areas covering 2.2 percent of the planning units. Twelve features were represented at less than 5 percent, while 16 out of 47 were represented at more than 10 percent (Figure 6.2).

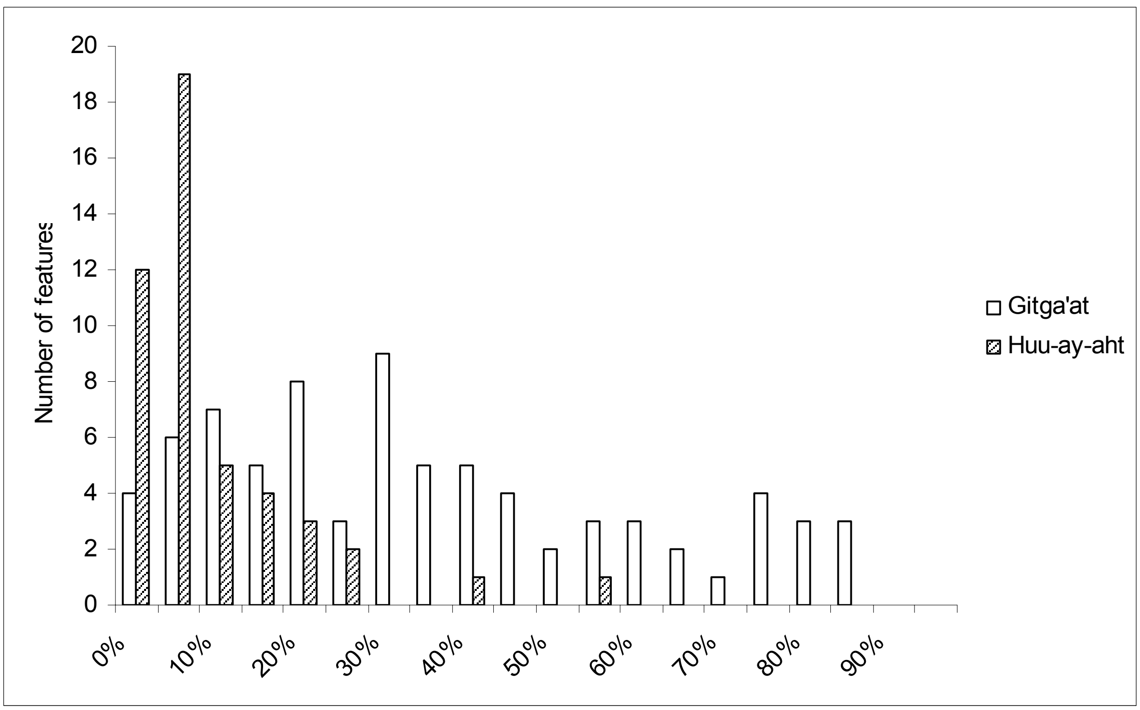

Figure 6.2: Proportion of features represented in the community-preferred marine areas in the Gitga'at and Huu-ay-aht study areas, British Columbia, Canada. For example, the first bars should be interpreted in the following ways: for the Gitga'at study area, there are four features whose coverage is represented between zero and five percent within the community-preferred areas; for the Huu-ay-aht study area, there are 12 features whose coverage is represented between zero and five percent within the community-preferred areas.

Participants at community meetings thought that the science-based maps represented important areas for conservation relatively well (Table 6.2), but did not capture them all. Such gaps, particularly for small inlets and bays in both case studies, are reflected in participants' comments about the science-based map. In the Gitga'at study area there was a broader range of opinions about the science-based maps than in the Huu-ay-aht study area (Table 6.2). In all cases, there was not a large difference between participants' scores of the maps, but subsequent discussions clarified participants' preferences for the integration approach.

Table 6.2. Summary of community feedback, with participant's opinion of areas of conservation importance on different maps. 5=represents their opinion extremely well, 4=pretty well, 3=neutral, 2=not too well, and 1=not at all

	Gitga'at (n=35)		Huu-ay-aht (n=10)	
	Mean	Standard deviation	Mean	Standard deviation
Community-preferred areas	3.43	1.34	3.60	0.55
Science-based map	3.50	1.40	3.80	0.45
Integration map	3.36	1.15	4.20	0.45
Minimum set of integration map	3.93	1.21	4.00	0.71

Integrating community-preferred and science-based areas

Given that the integrated scenarios combine the two approaches, we did not carry out overlap assessment because the comparisons to the constituent maps would not be independent. Using the same Marxan settings as for the science-based scenarios while locking in the community-preferred areas, the integration scenarios for the Gitga'at study area covered the following percentages of the study area: with a target of 10 percent of features, 23.7 percent of the study area was required to represent at least 10 percent of each feature; the scenario with a target of 30 percent required 40.5 percent of the study area to represent at least 30 percent of each feature. The solutions from the integration scenarios covered a larger area than the science-based Marxan solutions. Note that the area covered in the solution by Marxan would be further lowered by reducing the boundary length modifier. For the Huu-ay-aht study area, the solution with a target of 10 percent of features covered 12 percent of the area; the scenario with a target of 30 percent covered 32 percent. This is only slightly more than the science-based Marxan results. As can be seen from the integration maps, the results emphasized the regions surrounding the locked-in community areas as important for building onto the community-preferred protection scenario (Figure 6.1).

Participants considered the integration scenario as equivalent to, or better than, the science-based and community-preferred scenarios. During community feedback sessions, participants scored the integration map similar to the science-based map in both case study areas (Table 6.2).

In the Gitga'at study area, the scoring was very similar, but slightly lower than in the Huu-ay-aht study area. The example map of a protected area network (the minimum set) resulting from the integration scenario scored higher than the community-based scenario. While the feedback scales do not show a large difference among the scenarios, subsequent discussions highlighted participants' preferences for the integrated approach over the community-based or science-based approaches.

Human impacts and commercial fisheries as costs

Using alternate costs of human impacts and commercial fisheries yielded results that were quite different from science-based, community-preferred, and integrated maps (Figures 6.3 and 6.4, Table 6.1). The kappa statistic was not significant when comparing the scenarios with commercial fisheries as the cost of protection in the science-based scenarios or in the community-preferred scenarios. Using human impacts as the cost instead resulted in a lower overall overlap than in the other scenarios.

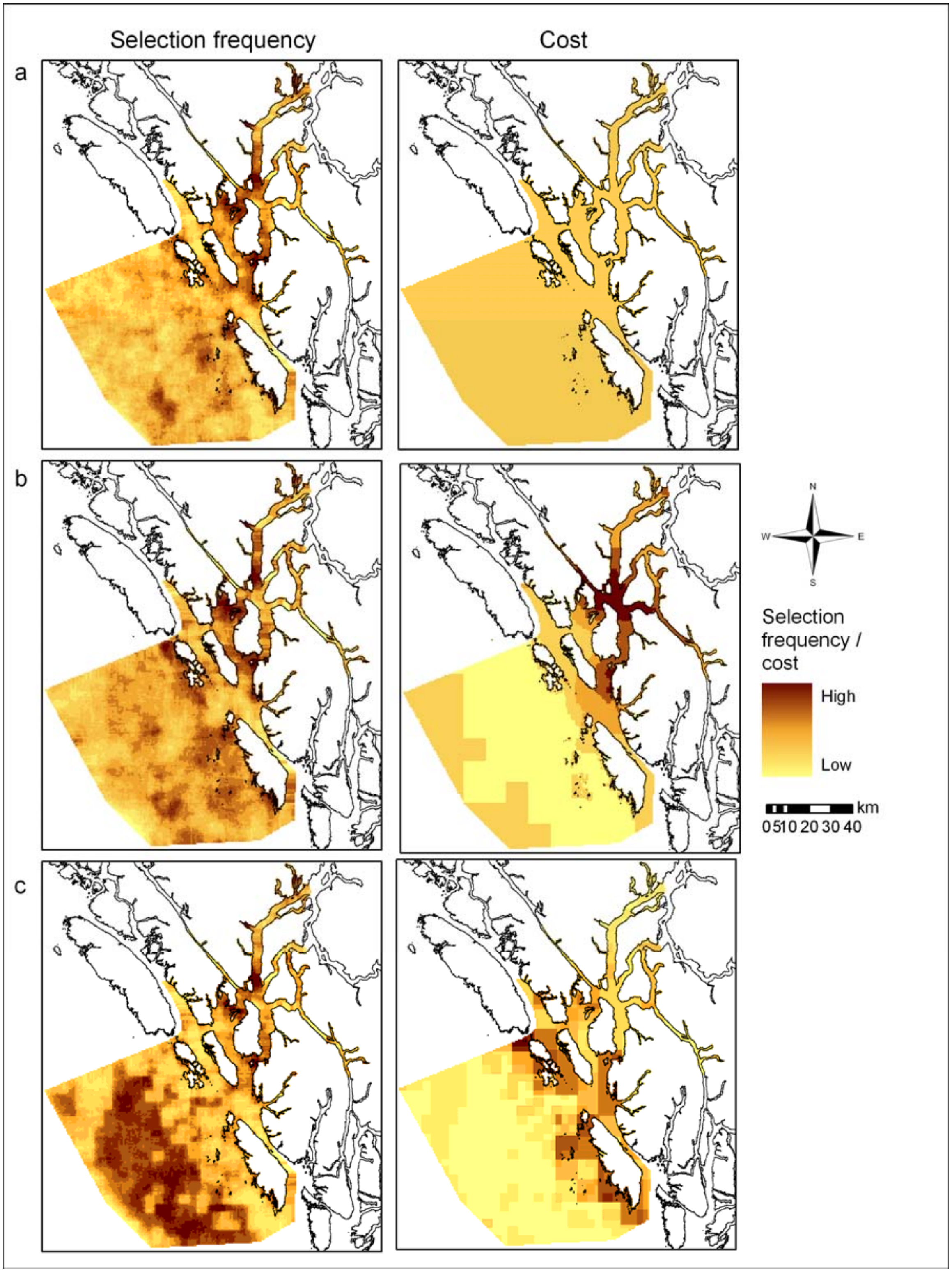

Figure 6.3. Cost scenarios, Gitga'at case study. The left column shows the selection frequency under a variety of cost scenarios; the right column depicts the corresponding costs. The cost scenarios are as follows: "a" uses area as a cost, "b" human impacts, and "c" commercial fisheries. The maps depict the 10% target scenario. The patterns of the 30% scenario are very similar, and are not shown here.

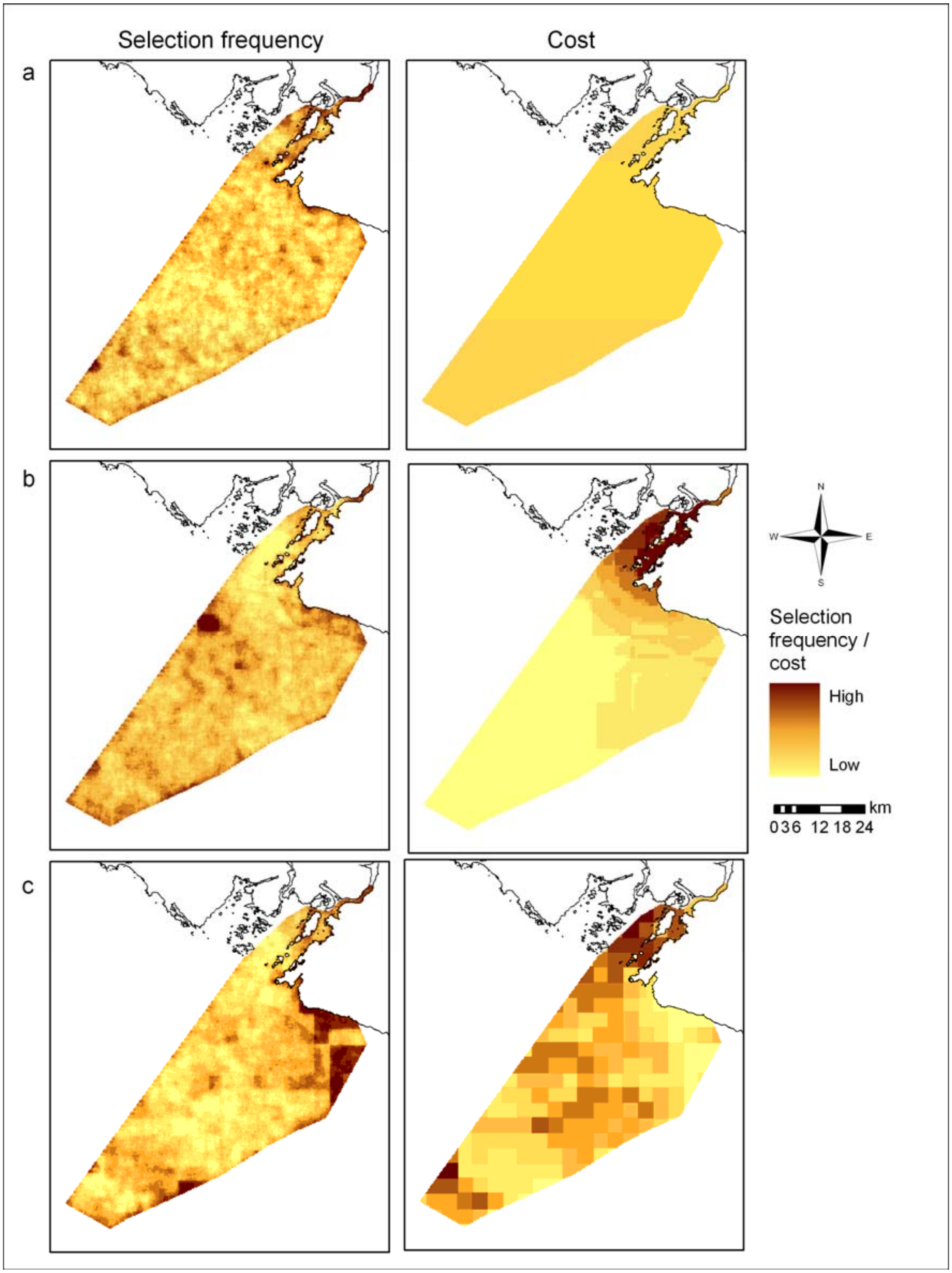

Figure 6.4. Cost scenarios, Huu-ay-aht case study. The left column shows the selection frequency under a variety of cost scenarios; the right column depicts the corresponding costs. The cost scenarios are as follows: "a" uses area as a cost, "b" human impacts, and "c" commercial fisheries. The maps depict the 10% target scenario. The patterns of the 30% scenario are very similar, and are not shown here.

Discussion

In the context of debate in the conservation biology literature about how to select areas for protection (Cowling et al. 2003; Knight & Cowling 2007), our findings lend credibility to both community-based and systematic science-based prioritization of MPAs. The overlap of the science-based approach with community preference, as well as the approval of community members, means that the results of Marxan captured most of the important areas identified by participants for protection. Similarly, the large overlap between sites preferred by participants and sites chosen by Marxan confirms the scientific validity of community-based choices.

In comparison to other studies, our results indicated that efficiency – selecting MPAs that minimize the cost – is not all that matters. Efficiency is commonly used as the main metric to argue for a systematic approach to site selection (Klein et al. in press; Stewart et al. 2003). However, in our case studies, participants scored the integration scenario higher than either of the constituent maps, and this was the least efficient result in that it covered the largest area. Influences other than efficiency apparently determine the acceptability of MPAs to people. Indeed, indigenous participants commented that they preferred to have more rather than less area protected, which the integration map provided. They also expressed approval of the representation approach inherent in the science-based approach (Margules & Pressey 2000; Possingham et al. 2000). The integration approach further alleviated the main criticism expressed by participants of the science-based approach, that it missed some areas they considered crucial for protection. Similarly, the integration approach ensured that offshore areas not apparent in the community-based approach were considered.

While our results showed that participants favored the integration approach, participants were assessing conservation areas that would be largely favorable to them as recommended during interviews: areas that exclude commercial and recreational fishing, but allow indigenous fishing (Ban et al. 2008). Thus perhaps it is not surprising that participants preferred the scenario that conserved the largest area. However, many participants also have commercial fishing licenses or are employed seasonally in the sport fishing industry. Thus, by favoring larger conservation areas, they are recommending the exclusion of some of their own activities from those areas.

Some participants commented on this specifically, indicating that they would be willing not to fish in those areas commercially.

Given the problems in obtaining good quality marine biodiversity data (Sala & Knowlton 2006), our results suggest that the community approach can be used as a reasonable proxy for the science-based approach. Spending time, effort and resources on collecting better species and habitat data may delay conservation actions. Instead, based on the results of our studies, we can recommend proceeding with a community-based approach rather than collecting additional data. There are other reasons for including people's opinions in the prioritization methods as well, including a better chance of implementation (Knight et al. 2006), and improved buy-in and compliance (Johannes 2000; Walmsley & White 2003).

The fact that science-based results are similar to community-based results indicates the robustness of a tool such as Marxan in the face of limited information. Our study used easily accessible abiotic and biotic data. Difficulties associated with such data are common (*e.g.*, limited metadata, presence-only data, emphasis on seabirds and mammals, very limited data for fishes, etc) (Ward et al. 1999). The community-driven approach pointed to some of the gaps in important areas that were missed when using Marxan. In particular, it missed some of the small inlets and bays which are important for invertebrates for which there are few or no data. We expect that the kind of data we used is representative of data availability elsewhere in developed countries. If similar analyses were undertaken elsewhere, those science-based results would also be built upon limited and imperfect data, particularly in many developing countries.

There were some differences between our case study areas. The Gitga'at study area showed more discernable patterns in the science-based analysis, and more overlap in the community-based and science-based approaches than the Huu-ay-aht study area. The reasons for such differences are not clear. Based on conversations with community members, we speculate that more Gitga'at than Huu-ay-aht members spend time on the water, and hence the Gitga'at participants might have better first-hand knowledge of marine ecosystems. Also, many Gitga'at community members travel to seasonal camps, giving them a broader exposure to the marine area. A larger proportion of the Huu-ay-aht case study is offshore, where travel is difficult and few people go. It is also possible that the science data is not as accurate for the Huu-ay-aht area, resulting in less overlap between the approaches.

The cost used in a decision support tool such as Marxan has enormous influence on selection frequency of planning units. The inclusion of socioeconomic data as a cost is sometimes seen as sufficient to represent the view of stakeholders (Klein et al. in press; Stewart et al. 2003). Different costs, however, can determine the pattern of conservation importance, and therefore the cost used has to be thought through and justified. Also, in the marine environment, the concept of cost is more nebulous than on land, where monetary value is often used as an accurate acquisition cost, for example. This can be done in the ocean as well, but spatially explicit cost data are much harder to come by. Because of the influence of cost, it would be good practice to analyze its effect by comparing scenarios using area as the cost. From a conservation perspective, the cost used has to be carefully considered to ensure that the results reflect practical cost(s) as closely as possible and that it treats multiple users fairly.

Our study corroborates a recent finding that integrating science-based approach and community-based approaches appears to be the best solution for MPA designation (Klein et al. in press). The other study used socioeconomic data as a cost in Marxan (Klein et al. in press). We instead used community and scientific approaches to judge each other – by asking participants to rate the science-based results, and by doing a gap analysis of the species and habitats contained within the community-preferred areas. Our two case studies, located hundreds of kilometers apart, both showed that the integration scenario was favored. Our integration approach, which built upon community preferences, was, however, not more efficient than the science-based approach.

Much of the literature portrays MPA selection based on social criteria as ad hoc or opportunistic (Pressey 1994; Stewart et al. 2003), but our study showed that selection by people (particularly those directly using marine resources) can be effective in conservation terms. During our interviews it became apparent that participants considered the whole study area known to them, and identified potential protected areas in several locations, thereby approaching a systematic assessment. Gap analyses of the features captured in the community-selected areas revealed a high level of representation compared to the area covered. This highlights the usefulness of local ecological knowledge by participants when selecting protection locations (Drew 2005). While we suspect that our interviews provided a representative sample of community opinions, we were unable to test this. Such use of local knowledge is crucial, especially when spatial data on conservation features are sparse or lacking completely.

The success of the integrated approach depends on good information from both the community and scientific information sources. For the community preferences to be meaningful, community members needs to have a solid understanding of their marine environment. Similarly, for the science-based approach to be ecologically relevant, the data need to be good enough to reflect ecological patterns. Success will further depend on the implementation of appropriate conservation measures that are socially acceptable and ecologically appropriate. In our case studies, we do not have the data to know whether allowing indigenous fishing while excluding commercial and recreational fishing would allow for the recovery of depleted species. Monitoring programs would have to be implemented to test the success of such conservation measures.

Our finding that an approach that integrates community-based and science-based data may be most successful in terms of community acceptance and biodiversity objectives means that there is hope for the successful establishment of MPAs. We speculate that by incorporating community preferences into MPA selection, subsequent implementation will be more expedient. Given the slow rate of MPA establishment, coupled with the continued decline of marine resources, establishing conservation measures is paramount.

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7. Beyond marine reserves: exploring the approach of selecting areas where fishing is permitted, rather than prohibited[7]

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Introduction

The oceans have suffered declines in faunal biomass and biodiversity (Myers & Worm 2003; Sibert et al. 2006; Worm et al. 2006), with fisheries constituting the single biggest human-induced pressure on marine life (Jackson et al. 2001). Marine reserves (no-fishing zones) have been widely hailed as providing one powerful tool for halting the decline of overexploited fish and invertebrate populations (Conover & Munch 2002; Halpern & Warner 2002; Hilborn et al. 2004; Roberts et al. 2001; Sala et al. 2002). The evidence that they increase biomass, abundance, and average size of exploited organisms within their boundaries (Halpern & Warner 2002; Roberts et al. 2001) has prompted international commitments to marine protected areas (including reserves) under the Convention on Biological Diversity and at the World Summit on Sustainable Development (Mora *et al.* 2006; Wood *et al.* 2007). Nevertheless, and despite this accord on the value of marine reserves, they are being implemented far too slowly to meet agreed targets for marine protection (Wood et al. 2007).

Given the slow rate at which marine reserves are being implemented, we here turn the problem on its head. What if, instead of initially assuming that the entire ocean were open to fishing, we began from the position that it was all protected from fishing? Management would then focus on designating areas where fishing was permitted, rather than prohibited (Dayton 1998; Walters 1998, 2000). At present, fisheries exploitation is specifically excluded in less than 1% of the world's oceans (Wood et al. 2007). Given biodiversity concerns and the challenging task of managing fisheries with limited data, it is increasingly vital to explore ways to restrict fisheries spatially while respecting their socioeconomic and nutritional contributions. Such restrictions should, ideally also meet systematic conservation planning criteria of representation and persistence (Margules & Pressey 2000). Conceptually this approach is very similar to using fisheries as a cost in marine reserve selection (Stewart & Possingham 2005; Stewart et al. 2003), except that here the emphasis is on reaching a fisheries target.

Quite apart from serving conservation goals, spatial restrictions on the total area fished could help secure sustainability in fisheries (Walters & Martell 2004). They would, for example, reduce overall fishing mortality, help rebuild depleted stocks (Roberts et al. 2001), and provide an insurance factor for the uncertainties inherent in fisheries management (Walters 1998).

Historically, sustainable fisheries have been those where a large part of the population occurred outside areas fished (Pauly et al. 2002).

The goal of our research was to assess whether selecting fishing areas may be feasible, and in particular what the conservation implications may be. We used data from the Pacific coast of British Columbia, Canada (approximately 49ºN to 54ºN latitude) as our trial study area.

Selection of Permitted Fishing Areas

The decision support program used

We applied Marxan (Ball & Possingham 2000; Possingham et al. 2000) - a decision support tool that has commonly been used to plan reserves - to spatial catch statistics for 13 commercial marine fisheries in British Columbia, Canada. Marxan tries to find the least expensive solution to the following objective function:

Total score = ∑ planning unit cost + (boundary length modifier * ∑ boundary cost) + feature penalty

We created a 2 km by 2 km grid, or planning units, to cover the study area, populated it with the spatial catch data, and then ran scenarios to select fishing areas. As spatial catch data were not available for recreational fisheries, our trial analysis is limited to commercial fisheries. We set the boundary length modifier – which controls the compactness of the output of Marxan – high enough so that the results were spatially compact. We set the penalty factor high enough to ensure that pre-specified commercial catch targets were met. Marxan provides a good approximation to an optimal solution by incorporating a random component to adding and removing planning units. Rather than settling on a single outcome, Marxan produces many solutions for any target that is proposed. The frequency with which particular planning units are chosen across different solutions is a measure of how important those planning units are for meeting the commercial catch targets efficiently.

The data

We obtained spatial catch data from Fisheries and Ocean Canada for 13 commercial fisheries in British Columbia, Canada. For confidentiality reasons, 8 sets of data had been summarized in 4 km by 4 km grids: ZN fishery (hook and line inshore rockfish), shrimp trawl, schedule 2 (hook and line, other species), sablefish trap, sablefish longline, prawn trap, groundfish trawl, and crab. Five sets of data had been grouped into 10 km by 10 km grids: sea cucumber, red urchin, krill, green urchin, and geoduck.

We normalized all data to the average annual catch (kg) for each planning unit. Data were summarized over the temporal duration of spatial data collection, which extended between 3 and 12 years for any given fishery (1993-2004). The 2 km by 2 km planning units we used assumed even spatial distribution of catches within each original 4 km by 4 km or 10 km by 10 km grid.

The scenarios and analyses

We set each scenario to maintain a particular target level of recent mean annual commercial catches, from 98% to 10%; the yield reductions thus ranged from 2% to 90% for eleven scenarios. We repeated each scenario ten times, with 100 runs of one million iterations each. The results for each scenario integrated all 13 fisheries, with each fishery maintaining at least that target catch. Therefore, our approach treated the commercial fisheries equitably.

We carried out a detailed assessment of the run with a 5% reduction in catches by examining the proportion of different habitat types or surrogates that fell within the areas where fishing was allowed to continue (Permitted Fishing Areas). These habitat types were described by depth, exposure, relief, slope, current, temperature, substrate, salinity and stratification. In addition, limited spatial information was available for the distribution of kelp, eelgrass, herring spawn areas, and clam beds.

We further assessed the performance of the run with a 5% reduction in catches on annual spatial catch data for four of the 13 commercial fisheries. We did this for the four fisheries for which we had annual data: geoduck, green urchin, red urchin and sea cucumber fisheries. Furthermore, we assessed the predicted reduction in catches for all 13 fisheries achieved by the 2%, 5% and 10% reduction scenario resulting in the least area fished.

Nature of Permitted Fishing Areas

Our analyses show that very small reductions in fisheries yields – if allocated in a strategic manner across space – can offer promising conservation benefits in both space and composition. For example, catch reductions of only 2%-5% could result in no-fishing areas covering 20% or 30% of previously fished areas (Figure 7.1). Every subsequent reduction in target catches yielded yet larger no-fishing areas (Figure 7.1 and Figure 7.2). Moreover, for each scenario, the multiple solutions that released the greatest area from fishing (Figure 7.2) described no-fishing areas that included representation from all twelve ecosections in British Columbia (Table 7.1); these ecosections delineate marine regions based on physical criteria. Maintaining catches at 95% of recent levels (or more, depending on the fishery) resulted in no-fishing areas that protected at least 17%, and an average of 55%, of each physical and habitat feature (Table 7.2). In this scenario, the total area protected would be 30% in exchange for a mean 4.6% reduction in catches (Table 7.3) and, perhaps, in profits if these map onto catches evenly.

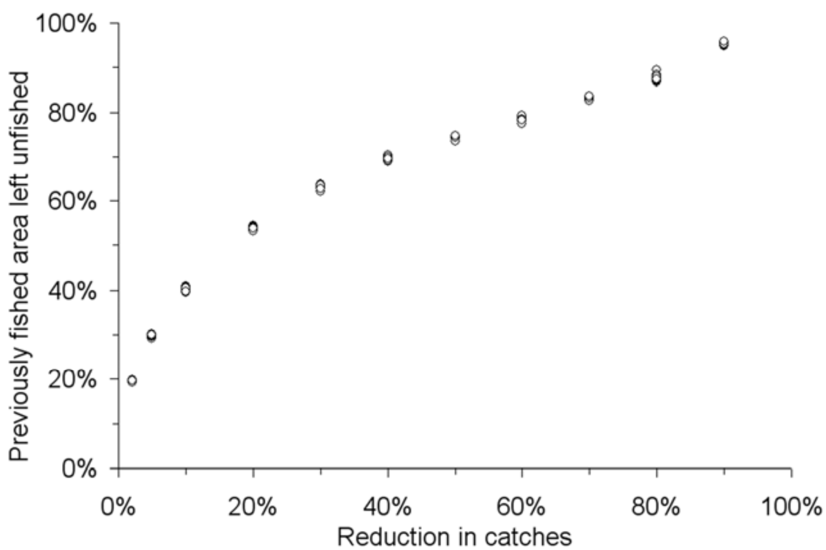

Figure 7.1. Decreases in areas fished resulting from reductions of catches for 13 commercial marine fisheries (British Columbia, Canada). Each of 11 scenarios was repeated 10 times, with 100 runs of one million iterations each (11,000 runs). The result requiring the least area of each of the 10 repetitions per scenarios is graphed.

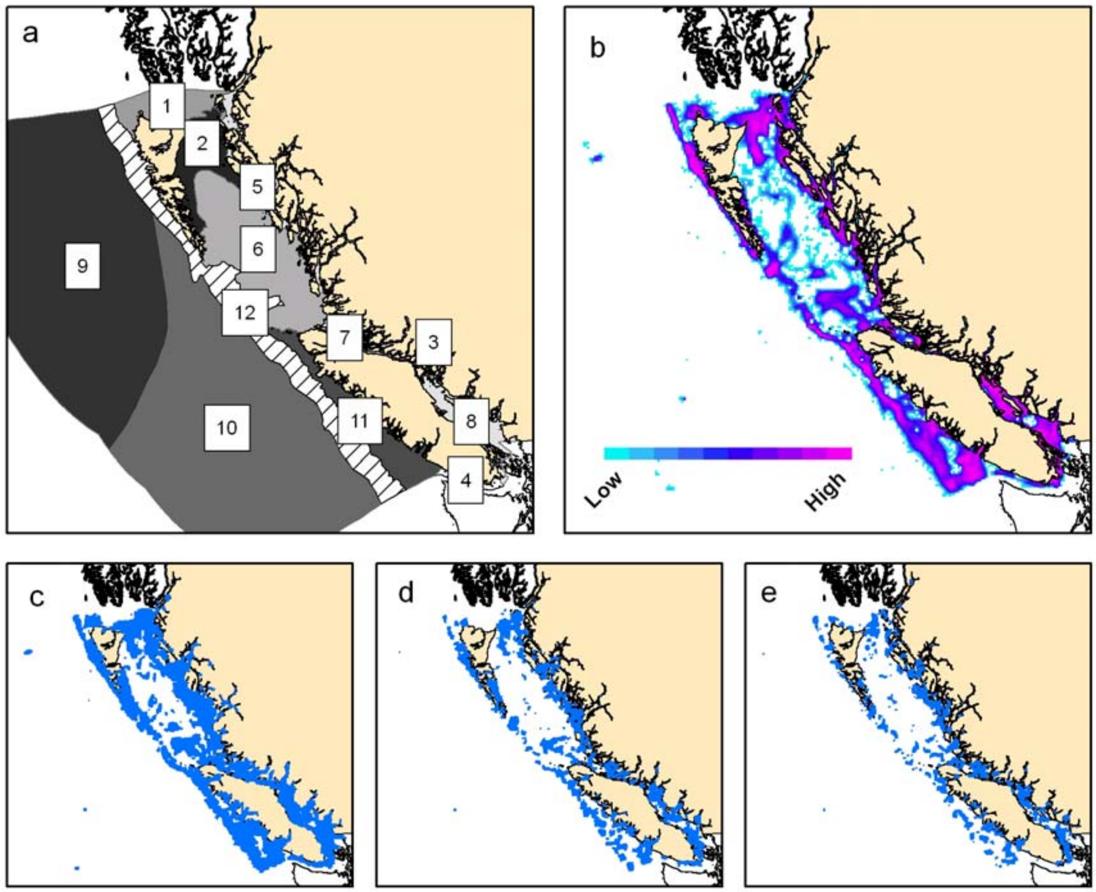

Figure 7.2. Marine ecosections in British Columbia and selected permitted fishing area solutions. The marine ecosections (a) are 1=Dixon Entrance; 2=Hecate Strait; 3=Johnstone Strait; 4=Juan de Fuca Strait; 5=North Coast Fjords; 6=Queen Charlotte Sound; 7=Queen Charlotte Strait; 8=Strait of Georgia; 9=Subarctic Pacific; 10=Transitional Pacific; 11=Vancouver Island Shelf; 12=Continental Slope. The selection frequency map (b) shows the importance of areas to commercial fisheries. The permitted fishing area solutions (in blue) are for a sample of the scenarios that minimize the area fished with the corresponding percent reduction in commercial fishing catches: (c) 5%, (d) 20%, (e) 40%.

Table 7.1. Gap analysis by ecosection for the most spatially limited result for each scenario.

	Area of ecosection (ha * 1000)	% of area fished	Percent reduction in catches (italics), resulting in percent protected (plain, in %)										
			2%	*5%*	*10%*	*20%*	*30%*	*40%*	*50%*	*60%*	*70%*	*80%*	*90%*
Continental Slope	3,330	53.8	55.3	60.0	64.8	74.5	78.6	79.2	84.5	88.5	91.9	89.6	97.2
Dixon Entrance	1,089	55.4	57.8	64.8	72.0	79.1	83.0	90.8	86.9	89.9	95.3	93.6	98.1
Hecate Strait	1,280	77.0	36.5	43.0	50.0	57.6	64.9	70.9	74.5	82.7	91.7	85.5	95.5
Johnstone Strait	239	98.0	11.4	19.2	24.1	46.3	58.1	65.1	98.0	73.0	81.7	81.8	96.1
Juan de Fuca Strait	150	90.8	15.5	27.0	67.4	56.7	74.6	72.8	69.5	96.6	100	97.7	100
North Coast Fjords	958	91.9	23.6	33.9	46.9	59.6	68.7	72.8	80.5	83.3	91.3	85.2	96.4
Queen Charlotte Sound	3,642	55.7	60.7	68.1	75.9	82.5	87.5	89.1	89.8	90.1	97.4	94.2	99.2
Queen Charlotte Strait	220	94.5	7.9	18.1	33.2	46.2	45.6	69.4	66.4	56.8	93.2	74.6	86.0
Strait of Georgia	815	94.8	7.9	63.0	14.6	27.3	31.2	51.7	50.7	64.5	79.4	77.8	92.7
Subarctic Pacific	17,098	0.3	99.8	99.9	99.9	99.9	99.9	100	99.6	100	100	100	100
Transitional Pacific	14,850	0.1	100	100	100	100	100	100	100	100	100	100	100
Vancouver Island Shelf	1,670	89.2	17.8	24.4	30.9	42.2	56.3	66.8	70.7	76.4	87.9	75.5	93.8

Table 7.2. Detailed analysis of the result of the 5% catch reduction scenario that produced the greatest area unfished, indicating ecosystem components that would be protected.

		Total area (ha) of each ecological feature	% outside permitted fishing areas
Depth	Shallow (0-20m)	743,853	40.6
	Photic (20-50m)	1,521,555	42.9
	Mid-depth (50-200m)	60,400,258	94.3
	Deep (200-1000m)	3,469,678	43.9
	Abyssal (>1000m)	33,627,695	99.7
Temperature (summer at seabed bottom)	Warm (9-15˚C)	2,438,557	32.9
	Cool (<9˚C)	42,820,022	88.0
Slope	Flat (0-5%)	40,556,889	87.2
	Sloping (5-20%)	4,737,411	67.4
	Steep (>20%)	42,749	43.0
Current	High (>3 knots)	212,713	39.0
	Low (<3 knots)	45,162,974	85.3
Substrate	Mud	2,295,529	27.6
	Sand	4,852,577	47.7
	Hard	3,631,788	53.5
Exposure	High	42,616,399	89.0
	Moderate	1,287,192	17.4
	Low	1,470,964	30.4
Relief	High	206,158	17.6
	Moderate	20,839,047	93.9
	Low	43,040,993	87.7
Salinity (annual average at surface)	Mesohaline (5-18ppt)	147,957	22.1
	Polyhaline (18-28 ppt)	11,279,517	91.7
	Euhaline (28-33 ppt)	43,945,636	87.0
Stratification	Mixed	4,931,996	36.8
	Weakly-mixed	2,083,666	42.3
	Stratified	37,823,783	94.4
Kelp		79,806	19.5
Eelgrass		10,449	28.7
Clam		18,978	22.5
Herring spawn		99,737	22.3
Sponge reefs		69,733	85.0

Table 7.3. Predicted catch reductions for each fishery under the scenario that (a) reduced overall catch by 2%, 5% and 10% and (b) produced the greatest area unfished at that level.

	Predicted catch reduction (%)		
Commercial fishery	2% catch reduction scenario	5% catch reduction scenario	10% catch reduction scenario
Prawn	2.0	5.0	10.0
Crab	2.0	5.0	10.0
Geoduck	2.0	5.0	10.0
Groundfish trawl	2.0	5.0	10.0
Krill	1.9	4.7	9.9
Green urchin	2.0	5.0	10.0
Shrimp trawl	2.0	4.2	7.2
ZN catch	2.0	5.0	10.0
Sea cucumber	2.0	4.7	10.0
Schedule two	2.0	5.0	10.0
Sablefish trap	2.0	5.0	10.0
Sablefish longline	1.2	1.5	3.4
Red urchin	2.0	5.0	10.0

The approach we employed for selecting Permitted Fishing Areas used catches averaged over multiple years as the input, yet the result of the 5% reduction scenario also performed well when analyzed using annual catches for geoduck, green urchin, red urchin and sea cucumber fisheries (Table 7.4). As expected, we found some inherent spatial and temporal variability in the proportion of catches that would fall within the Permitted Fishing Area each year. The greatest range for a target of 95% of catches retained across all fisheries was a 2-12% reduction in sea cucumber catches, depending on the year.

Table 7.4. Proportion of annual commercial fisheries catches that fall within the permitted fishing area result of the 95% target scenario.

Fishery	Annual data	Average	Standard deviation	Minimum	Maximum
Geoduck	2002-2004	95.04%	1.91%	92.96%	96.73%
Green urchin	1998-2003	94.29%	2.89%	90.49%	97.05%
Red urchin	1997-2003	95.07%	0.39%	94.72%	95.82%
Sea cucumber	1997-2004	94.81%	3.07%	88.25%	97.97%

Potential conservation and fisheries benefits of Permitted Fishing Areas

Conservation benefits

The practical approach used in this study allows for explicit analyses of trade-offs between small reductions in fisheries – in a spatially strategic manner – and large gains for marine conservation through spatial protection. Managing marine environments by selecting permitted fishing areas rather than marine reserves would represent a much-needed paradigm shift in areas where little headway is being made in marine reserve establishment. Instead of debating the merit of each potential marine reserve, the discourse could focus on analyses of the ecological benefits of small reductions in fishing, and the ecological costs of small increases in fishing that would only come by making much larger areas accessible to fishers.

This approach seems to offer real conservation benefits. At a minimum, the approach outlined here would protect the same proportion of fished populations as the target reduction in catches, assuming even catchability. Even small marine reserves that protect only a fraction of populations have been shown to increase the size, number, and diversity of fish within their boundaries (Halpern 2003; Halpern & Warner 2002; Tetreault & Ambrose 2007). Given the increase in fecundity of fishes that are able to grow larger and live longer within protected areas, protecting even a small proportion of the population could greatly enhance numbers in areas that continue to be fished. For larger species, no-fishing areas are predicted to exceed yields of traditional fisheries management by up to 60% in areas that remain open to fishing (Gaylord et al. 2005).

Even though ecological goals were not included *a priori* in the designation of the permitted fishing areas, the areas that fell outside permitted fishing areas included good representativeness across ecosections (Zacharias et al. 1998) (Table 7.2). Further detailed analysis of the scenario with 5% catch reduction showed that the areas outside the permitted fishing area represented key physical and habitat features, with some temporal variability for fisheries. The representation of habitats suggests that the proportion of populations protected through this approach is likely to be greater than the reduction in catches. Such results indicate that the approach may approximate the outcomes sought in designating MPAs.

Fisheries benefits

Even while protecting large (and representative) tracts of ocean, the proposed approach of designating permitted fishing areas could reasonably be expected to also strengthen fisheries in three ways. First, the removal of destructive fishing gear from the areas outside the permitted fishing areas should promote improved habitat quality (Collie et al. 2000), while also eliminating bycatch (Roberts et al. 2005). Second, given the benefits of even small reserves for population recovery (Russ & Alcala 1996), the areas outside the permitted fishing areas may enhance fish populations within permitted fishing areas (Polacheck 1990; Roberts *et al.* 2001). Third, many fisheries around the world are operating unsustainably (Pauly et al. 2002), such that reductions in catch while setting permitted fishing areas could also move these fisheries closer to desired biological reference points for sound management (Collie & Gislason 2001). Such changes might offset catch reductions over the long run.

The flexibility of the approach used here could help to enhance societal acceptance of and compliance with spatial planning, particularly among fishers. Marxan is a decision support tool that facilitates decision-making, without making decisions. Indeed, because it offers multiple solutions that may differ only slightly in their efficiency, the exact choice of permitted fishing areas can be adjusted for social acceptability and ecological viability (Fernandes et al. 2005). Fishers' input will be important in setting commercial catch targets by fishery, verifying formal data (Johannes 2000), mapping and scaling fisheries that lack formal spatial data, and in agreeing to the permitted fishing areas. A more advanced version of our analyses would incorporate other commercial fisheries, recreational fisheries, timing of fishing effort, and more detail on ecologically important areas. Ironically, launching the assessment process we propose – in a consultative fashion – might be a particularly effective way of eliciting or prompting the collection of just such important data, which are seldom available (or at least publicly accessible) in even the best resourced management jurisdictions.

The approach of selecting permitted fishing areas would be expected to yield useful results in other geographic areas. Gear types used in British Columbia are typical of commercial fisheries elsewhere – trawl, hook and line, gillnet, seine, trap, and dive – and bioeconomic models suggest consistency in fisher behavior across locations (Walters & Martell 2004). Moreover, modelling has previously shown that optimal harvesting strategies always include marine reserves when certain assumptions are met, even before consequent improvements in habitat recovery are considered (Hastings & Botsford 1999). Trials of this approach must, however, be taken elsewhere to determine whether, for example, the resultant no-fishing zones are generally ecologically representative.

Assess the costs and benefits

While optimistic about the potential of our approach, we are well aware that many challenges remain to be resolved. First, some fisheries that already operate sustainably might gain few benefits from the spatial management we propose, and would essentially be making concessions for other fisheries and/or for broader conservation principles. Second, our approach focuses only on fisheries (and only commercial fisheries in this trial study), whereas other marine and terrestrial uses also significantly impact the ocean. Fishing, however, is the main threat, and hence a tangible starting place for making conservation gains. Third, the large no-fishing zones arising from our approach, might lead to claims that no further areas need be protected, whatever their importance for conservation. Fourth, it remains to be determined whether the areas protected through this approach would provide the same conservation benefits as the same protection gained through conventional science-driven marine reserve selection. We do, however, know that both approaches tend to lead to protection for areas that are less valuable economically.

As ever, no single management measure will achieve all goals. The effectiveness of our approach in terms of accelerating protection will depend, in large measure, on the extent to which fishers gain yields in proportion to the benefits they cede in the no-fishing zones. Some conflict is still likely if, for example, the best fishing grounds – and hence the areas most likely to be included in permitted fishing areas – are also (a) the most sensitive habitats with the highest fish densities or (b) the most sensitive habitats. These areas would ideally be protected in no-fishing zones. Worse, by leaving them in the permitted fishing areas, they might come under more concentrated fishing unless quotas were reduced commensurate with the spatial contraction of the fishery. In terms of spatial management, the best approach is likely to combine the selection of permitted fishing areas with the identification and protection of sensitive habitats.

The designation of permitted fishing areas will have similar obstacles to the selection of marine reserves. First, there are data availability issues and knowledge gaps. In both cases, we usually lack spatial data for at least some fisheries, biological and range data for at least some species, and an appropriate understanding of dispersal and connectivity (Palumbi 2004). Second, similar implementation and management issues might arise for permitted fishing areas and marine reserves. Enforcement would still be a challenge, and the political will to proceed with establishment has to exist for advances to be made. One answer is that both approaches will need to take an adaptive management approach to ensure that objectives are being met, continuously acting, assessing and revising.

Conclusion

We have little to lose – and much to gain – in trying a new approach in areas where marine conservation advances have been inadequate. It appears, *ab initio*, that large areas that are representative of ecoregions and habitats might be protected at a small cost to fisheries (although particularly sensitive areas might may have to be included *a priori*). Moreover, the dependency of the approach on explicit commercial catch targets for each fishery forces us to define the trade-offs we are willing to make to ensure a healthy ocean. The alternative to the approach described here seems to be the continuation of the *status quo*, which has resulted in the sequential collapse of fisheries (Myers & Worm 2003; Pauly et al. 1998) with only a small proportion of the ocean protected by marine reserves.

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8. Conclusion

Summary of thesis and status of research objectives

The research presented in this thesis achieved my goal of making theoretical and practical contributions to planning for marine protected areas (MPAs). I provided the first direct comparison between – and integration of – community-based and science-based approaches to MPA establishment. I also made novel contributions to systematic conservation planning by mapping human stressors in the marine environment, determining the level of data needed for systematic conservation planning, and reversing the conventional MPA approach by selecting areas for fishing instead of protection. In this concluding chapter, I present a summary of my research, its strengths, applications, limitations, and implications for future research.

I achieved the first objective of this thesis – to develop techniques and carry out analyses to advance conservation planning in the marine environment – in chapters 3, 5, 6 and 7. I addressed my second objective – to compare and integrate science-based and community-based approaches to MPA prioritization – in Chapters 2, 4, 5 and (particularly) 6.

I first developed a literature-based context to marine protected area (MPA) prioritization and establishment (Chapter 2), arguing that science-based and socially-driven MPA approaches to MPA selection are more intertwined in practice than the literature acknowledges. This review set the stage for Chapters 4, 5, and 6, where I conducted primary research to compare and integrate the approaches.

In Chapter 3, I developed a GIS approach to mapping human stressors in the marine environment. Knowing the threats to the ocean, where they occur and their impacts, is a critical contribution to subsequently managing such pressures (Margules & Pressey 2000; Pressey et al. 2007), and has been identified as a gap in the literature (Hixon et al. 2001; Sarkar et al. 2006). My mapping approach combined information on stressors resulting from human activities (*e.g.,* Folt et al. 1999; Halpern et al. 2007; Hughes & Connell 1999; Johnson et al. 1998; Porter et al. 1999; Ruiz et al. 1999; Vinebrooke et al. 2004; Zacharias & Gregr 2005), the relative impact of different activities (*e.g.,* Halpern et al. 2007; Jamieson & Levings 2001), and the distance to which the effect of activities was felt (*e.g.,* Bolton et al. 2004; Foote et al. 2004; Hampton et al. 2003; Krkosek et al. 2007; Krkošek et al. 2005), with spatial information on the location of activities. This chapter used methods similar to Zacharias and Gregr (2005), but incorporated all known and mapped human activities rather than just two human threats. My approach is widely applicable, and will facilitate marine conservation elsewhere. The resulting human stressors maps for BC provided a powerful rationale for MPAs: Very little of the ocean, and almost no part of the continental shelf of British Columbia, lies beyond the reach of human stressors.

I developed the community-based approach to MPA planning in Chapter 4. Given that indigenous people in Canada have resource use rights (Ayers 2005; Garibaldi & Turner 2004; Harris 2002; Turner et al. 2000; Turner & Jones 2000), I engaged two First Nations in British Columbia for their views on marine planning and protected areas. I developed a three-phased approach for executing my research: building research partnerships, carrying out individual interviews, and holding community discussion sessions. My results showed that participants expressed a common goal of recovering depleted species and ensuring the sustainability of indigenous fishing. I found strong support for spatial protection measures, and significant overlap in the areas that different individuals suggested for protection. The overlap evident from individual interviews was validated during community meetings, at which participants agreed on preferred conservation areas. The most common type of protection recommended by participants was the exclusion of commercial and recreational fisheries (despite some First Nations involvement in these activities) while allowing for indigenous fishing; this stands in contrast to the emphasis on no-take MPAs in the literature. The congruence of the goal, and level and areas of protection among many people points to a gap in conservation approaches: the conservation of areas of importance to indigenous people, where they can continue to practice and adapt their culture. My study suggests that using a community-based approach to identifying priorities may indeed be a feasible option (as also seen in Berkes 2004; Brown 2003; Campbell & Vainio-Mattila 2003; Christie et al. 2002; Forgie et al. 2001; Pollnac et al. 2001; Pomeroy 1995; Salafsky et al. 2001; Wilson 1999).

My analyses in Chapter 5 again offered new approaches to marine conservation planning. To my knowledge, this was the first study that assessed the relative contributions of spatially limited datasets to a systematic MPA planning approach (but see Beger et al. 2007; Gladstone 2002; Ward et al. 1999 for a similar analysis of the effectiveness of surrogates). Conservation planning needs to proceed even in the absence of detailed information on the patterns of distribution of biota (*e.g.,* Banks et al. 2005). I therefore applied a decision support tool, Marxan

(Ball & Possingham 2000; Possingham et al. 2000), and commonly available biotic and abiotic data to help determine where MPAs should be placed in two regions of British Columbia, Canada. I next tested the robustness of this method by sequentially removing datasets with limited geographic distributions and applying surrogates. I found both that the reserve selection method was robust to some missing data, and that it was best to use a combination of abiotic and biotic data to ensure habitats and species were represented. Biotic data served as better surrogates for abiotic features than vice versa, and both represented more species or habitats (hereafter referred to as features) than occurred in randomly selected reserves. The results should provide encouragement to decision-makers engaged in MPA planning with limited spatial data.

In Chapter 6, I directly address my second objective by comparing and integrating the approaches from Chapters 4 (community selection) and 5 (scientific selection). A debate about the efficacy of community-based vs. science-based (sometime also referred to as ad hoc or opportunistic vs. systematic conservation planning) reserve selection has been attracting attention in the scientific literature (Cowling et al. 2003; Knight & Cowling 2007; Margules & Pressey 2000; Pressey 1994; Pullin et al. 2004; Roberts 2000; Sarkar et al. 2006; Smith et al. 2007; Stewart et al. 2007; Stewart et al. 2003). While past studies assessed the efficiency of social versus systematic selection approaches (*e.g.,* Gonzales et al. 2003; Klein et al. In Press; Stewart et al. 2003), none have more thoroughly compared and integrated the two approaches. My findings that the approaches verified each other lend credibility to both community-based and science-based approaches for prioritizing marine areas. Indeed, an integration of the two was preferred by participants and also achieved all conservation objectives. My study also provides empirical evidence that areas for protection selected socially can provide biodiversity benefits (as concluded by Roberts (2000) and Knight and Cowling (2007)).

In my final data chapter (Chapter 7), I took a step back from the conventional approach of selecting MPAs. This chapter was aimed at my first objective of advancing marine conservation by developing and testing the innovative approach of setting aside areas where fishing would be permitted, rather than prohibited. While MPAs hold promise for marine conservation, implementation is too slow to protect significant parts of the ocean (Wood et al. 2007). To meet international commitments, we must start to think creatively and look beyond existing approaches to marine conservation. My work, using spatial data for thirteen commercial fisheries on Canada's west coast, revealed that small reductions in fisheries yields, if strategically located, could allow creation of large unfished areas. Given that such unfished areas included diverse biophysical regions and habitat types, it appears that small reductions could achieve remarkable conservation gains. This approach had been suggested in the literature (Dayton 1998; Walters 1998), but had never been analyzed for its potential conservation contribution, although Walters and Martell (2004) carried out a similar analysis to maximize fishing profitability.

Strengths and applications of this thesis

One of the main strengths of this thesis as a contribution to the field of conservation biology is that it provides theoretical and applied contributions to the discipline. In this thesis, I have made the following original contributions and findings:

- I executed a pioneering study in mapping multiple human stressors in the marine environment (Chapter 3). I found that nearly all of the continental shelf of British Columbia is impacted by stressors resulting from human activities. This strengthens the argument that MPAs are needed.
- I developed a new framework of eliciting community preferences for protection (Chapter 4). The combination of individual interviews and community meetings ensured that a range of opinions were obtained. The emphasis on collaborations – by developing partnerships and obtaining explicit permission to carry out work in the communities – and leaving decisions on conservation actions with the communities, can lead to empowerment by the communities.
- I carried out the first study to assess the contribution of datasets with small geographic distributions to site prioritization (Chapter 5). I found that there are diminishing returns to including numerous datasets in a Marxan analysis. Based on my case studies, this means that it is not necessary to wait for additional datasets; including habitat data and some biological data will result in similar patterns of conservation importance as including more data.
- I developed and carried out the first study to directly compare and integrate community prioritization to prioritization using science-based systematic conservation planning

(Chapter 6). The results show that integrating community and science-based approaches are the preferred approach, and that the community-based approach would be a suitable proxy to a science-based approach if data are not available.

- I carried out the first study to explore the conservation potential of creating fishing areas rather than protected areas (Chapter 7). I showed that, from a conservation perspective, this could be a viable alternative to the current approach to protected areas.

Because conservation biology is a normative discipline, facilitating conservation actions is as important as, if not more than, making theoretical scientific contributions. Such application is relatively rare (Robinson 2006) but my thesis has made an applied, practical contribution to conservation actions in the following ways:

- The stressor mapping (Chapter 3) is already being applied to conservation planning in British Columbia. I have been working with WWF Canada and Parks Canada Agency to refine the methods developed in Chapter 3 in order to incorporate the results into systematic conservation planning. Parks Canada is planning on using the results of my model as a scenario in their National Marine Conservation Area feasibility and interim management planning, and WWF Canada intends to use the maps to engage in marine planning in BC.
- My community-based approach and integration approaches are providing input into the First Nations' marine planning efforts. In particular, the Gitga'at First Nation established a marine planning committee in 2006 that is using the community-based mapping results from my thesis and the integration approaches, to develop a marine use plan for their traditional territory. I am likewise working with the Huu-ay-aht First Nation to provide maps from my research to further their marine planning endeavours.

In addition, several of my chapters could result in an applied, practical contribution:

- For my work on Chapter 4, I developed research partnerships with two First Nations in BC. As part of my research agreements, I left a copy of all of my research data (individual identifiers removed), results and analyses with the partners to allow them to use the results in their planning efforts as they deem appropriate.
- Chapter 5 provides practical guidance of how much data are necessary to pursue a systematic conservation approach. This finding could be taken up by any MPA

planning process in the world, and could result in systematic conservation planning proceeding where it may otherwise be delayed.

In addition to the above contributions, my thesis has implications for marine planning in British Columbia. Establishment of MPAs has been an extremely slow process in British Columbia (Office of the Auditor General of Canada 2005), in part because of a reluctance by First Nations to relinquish their rights to fish within no-take areas (Ayers 2005; LeRoy 2002). My research has shown that First Nations (at least the ones I worked with) are keen to discuss marine conservation options if they are able to make the decisions of which places to protect, and what kind of protection to implement. The predominant recommendation from interviews and community meetings (Chapter 4) was to exclude commercial and recreational fisheries from areas chosen by participants, but allow aboriginal fishing. There was a general willingness, however, to consider no-take areas as well, as long as their placement could ultimately be decided by the First Nations. Thus, if the federal and provincial governments wanted to pursue MPAs in British Columbia, it would be in their best interest to form partnerships with First Nations, and to use an integrated approach – combining community-based and science-based MPA prioritization (Chapters 4, 5, 6) – in marine planning.

Limitations of thesis

The focus of this thesis was on carrying out the best possible MPA envisioning given the limitations of data availability and our incomplete knowledge of the marine environment. It is hardly surprising, then, that I was working with limited data and unable to explore all aspects of MPA design.

One limitation (that I recognised *ab initio*) was that I restricted the social scope of my thesis to the perspectives of indigenous peoples in BC. I chose this focus because of the special relationship indigenous peoples have with the environment (Ayers 2005; Garibaldi & Turner 2004; Turner et al. 2000), and because indigenous peoples are involved in the many activities that take place in the ocean (*e.g.,* shellfish farming, commercial and recreational fisheries). Nonetheless, such a focus creates limitations. For example, although some participants of this study were commercial fishermen, they do not necessarily represent the perspective of nonnative commercial fishermen. As well, I did not explore perspectives of MPAs with stakeholders such as anglers. (Buanes et al. 2004; Dalton 2005; Fraser et al. 2006; Kessler 2004; Lundquist & Granek 2005; Salmona & Verardi 2001; Scholz et al. 2004; Suman et al. 1999).

Limited fisheries data influenced the analyses I was able to undertake. A data limitation was that I did not have access to spatial catch records for recreational fisheries (but see Coleman et al. 2004; Cooke & Cowx 2004; Lynch 2006; Schroeder & Love 2002 for studies of recreational fisheries), and therefore only examined commercial fisheries (Chapters 5, 6 and 7). I used commercial catches as a proxy for the economic importance of areas to commercial fisheries. Yet many other factors contribute to the profitability of areas for commercial fisheries, and the effect of MPAs on commercial fisheries (Groeneveld 2005). For example, distance from port, the habitat type fished, bycatch generated, and subsidies may all contribute to the profitability of fisheries (Abernethy et al. 2007; Sumaila 2003; Sumaila et al. 2000; Whitmarsh et al. 2000). Economics was not a focus of this thesis, and therefore I did not attempt to test or develop spatial measures of profitability of fisheries.

A third limitation of this thesis was that I did not assess the ecological effectiveness of the MPA scenarios, particularly persistence of species and habitats. The proxy I used for the goal of persistence was to target a percentage of all conservation features when applying Marxan. This is the same approach as carried out in most applications of systematic planning tools like Marxan (Banks et al. 2005; Cabeza & Moilanen 2001; Cook & Auster 2006; Loos 2006; Munro Royle 2007; Possingham et al. 2000; Puniwai & Gibson 2005). However, using a percentage of representation of each conservation feature does not necessarily ensure the persistence of those features. Data do not exist to facilitate an assessment of persistence, and it was not within the scope of the thesis to collect the requisite species distributions, life history parameters, and food web interactions in order to model persistence (*e.g.,* as done by Ainsworth 2004; Baskett et al. 2007; Micheli et al. 2004; Wagner et al. 2007; Walters 1999; Zeller & Reinert 2004).

Comments on future research

The contributions of this thesis highlighted several themes for future research that would serve to further contribute to marine conservation:

- The marine stressor mapping (Chapter 3) served to provide an overview of human stressors in the marine environment. The methods used, however, were based on modelling of threats given our knowledge about the effect distance of stressors and their impacts. To assess the viability of this approach, empirical studies should be carried out to ground-truth the mapping approach. Such studies could use the modelled stressor maps as a basis for ground-truthing, selecting low, medium and high stressed locations, and measure conditions that represent the state of the ocean.
- The mapping of marine stressors (Chapter 3) emphasized the negative impact of human activities on the ocean. Focusing instead on studying and mapping the benefits humans receive from ecosystems (*i.e.*, ecosystem services) might allow us to better appreciate the negative impact of stressors on humans (direct or indirect) (Balvanera et al. 2001; Beaumont et al. 2007; Carpenter et al. 2006; Chan et al. 2006; Hixon et al. 2001; Palmer et al. 2004; Palmer et al. 2005). Such a positive approach might also be more appealing to decision-makers who can enact conservation measures.
- The indigenous perspectives of marine conservation as described in Chapter 4 highlight the need to study the ecological effectiveness of partially protected marine areas. Studies of the effectiveness of MPAs to date have focused primarily on no-take MPAs (*e.g.,* Bene & Tewfik 2003; Bohnsack 2000; Chiappone & Sullivan Sealey 2000; Edgar & Barrett 1997, 1999; Fryxell et al. 2006; Gell & Roberts 2003a, b; Gerber et al. 2003; Halpern 2003; Halpern & Warner 2002; Roberts et al. 2001; Russ & Alcala 2004; Tetreault & Ambrose 2007). Relatively few studies have examined the effectiveness of partial-take MPAs (but see Denny & Babcock 2004; McClanahan et al. 2006). Because the preference of indigenous people in this study was to allow some fishing inside protected areas, guidance on the ecological effectiveness of such areas is urgently needed, including recommendations on how much and what kind of extraction is possible to maintain some conservation benefits.
- My assessment of the data needs to carry out systematic prioritization was based on available data for BC (Chapter 5). Ideally, the study should be repeated in an area where a detailed, systematic survey of all marine habitats and biota has been carried out, if/when such a surveyed area exists. Then the same methods I used can be used to test the data needed to represent all biodiversity, rather than the limited mapped components of biodiversity I used for BC. This would serve to test my findings.

In addition to future research as highlighted by my thesis, there are many important avenues for further research to make MPAs as effective as possible. Many of these topics are the subject of active and important research by other scientists.

The success of MPAs hinges upon community support (Walmsley & White 2003), yet our understanding of the factors that lead to successful community engagement and support is incomplete. A thorough documentation of lessons learned from MPA successes and failures, with a specific focus on the social dimensions that contributed to successes and failures, could help MPA establishment efforts direct resources and attention to achieve a greater likelihood of success. We need, for example, to understand how communities and users of the marine environment react to the establishment of MPAs (National Centers for Coastal Ocean Science 2007; NOAA 2003). If fishers fish areas outside of the MPA more intensively following MPA establishment, then additional fisheries management measures – and appropriate compensation – might be needed. We also need to understand whether there are conditions that make communities more likely to want MPAs – such as education or familiarity with the marine environment. If we can achieve a better understanding of such conditions, then outreach, education and development efforts can be geared towards creating communities open to implementing conservation efforts.

Most current MPA design efforts do not properly account for the processes that underpin biodiversity, and further research into opportunities for incorporating processes into MPA design would improve the effectiveness of MPAs (Pressey et al. 2007). On land, for example, a method has emerged for incorporating patch dynamics and fire into protected area planning (Leroux et al. 2007). A guide for conservation planners to narrow down the processes to be considered in conservation planning is already available and suggests that we consider processes that (1) we know about, (2) are understood well enough for their spatial requirements, and (3) for which conservation planning can make a difference (Pressey et al. 2007). Understanding processes that are needed to ensure persistence of biodiversity is particularly important (Cabeza & Moilanen 2001; Possingham et al. 2006). Some scientists are carrying out research to allow future applications to plan for persistence more directly (Allison et al. 2003; Cabeza & Moilanen 2001; Nicholson et al. 2006; Salomon et al. 2006; Sarkar et al. 2006).

One of the most important processes in the marine environment is connectivity. Given our limited knowledge, further research to incorporate connectivity into MPA design may prove to be a fruitful area of future research. Many researchers are already focusing on connectivity (Gerber et al. 2005; Grantham et al. 2003; Klinger 2001; Largier 2003; Ogden 1997; Palumbi 2003, 2004; Robinson et al. 2005; Shanks et al. 2003; Stockhausen et al. 2000; Warner et al. 2000) but we need to know much more about species distributions, larval dispersal trajectories, and ocean current patterns in order to incorporate connectivity into MPA design for multiple species (Palumbi 2004).

Ultimately, given the degraded state of the ocean (Chapter 3), what matters most is the timely implementation of conservation actions. Research that facilitates this, and that can suggest the best options for conservation actions given the realities faced by decision-makers – *e.g.,* limited data and knowledge, uncertainties, trade-offs amongst various interest groups – should be emphasized. MPAs may be only one tactic in a global marine conservation strategy, but they have a higher chance of being used and useful - if we can get them right - than many other approaches.

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9. Appendices

<u>Appendix 1: Data used in spatial analyses</u>

	Data source	**Gitga'at study area**	**Huu-ay-aht study area**
Abiotic data	Province of British Columbia	Complexity (Ardron 2002) Current high Current low Depth deep Depth mid-deep Depth photic Depth shallow Exposure high Exposure low Exposure moderate Relief high Relief low Relief moderate Salinity euhaline Salinity polyhaline Slope flat Slope sloping Stratification mixed Stratification stratified Stratification weakly mixed Substrate hard Substrate mud Substrate sand Temperature cool Temperature warm	Complexity (Ardron 2002) Depth photic Depth shallow Exposure high Exposure low Exposure moderate Relief high Relief low Relief moderate Salinity euhaline Salinity polyhaline Slope flat Slope sloping Stratification mixed Stratification stratified Stratification weakly mixed Substrate hard Substrate mud Substrate sand Temperature cool Temperature warm
Biotic data	Province of British Columbia, Gitga'at First Nation	Humpback survey Orca survey-resident Orca survey-transient Sponge reefs Alcids Diving ducks Eagles Fulmar Geese Gulls Marbled murrelets Other pelagic birds Shorebirds Other waterfowl Eulachon Herring Eelgrass Kelp Gray whale Harbor porpoise Pacific white sided dolphin Sea lion haul-outs Dark brown kelps Soft brown kelps Bleached red algae Surf grasses Marsh grasses	Alcids Blue heron Black oystercatcher Cormorant Dabbling ducks Diving ducks Eagles Fulmar Geese Gulls Loons and grebes Shorebirds Herring Eelgrass Kelp California sea lion Dall's porpoise Grey whale Harbour porpoise Northern fur seal Sea lion haul-outs Sea otter Steller sea lion

	Data source	**Gitga’at study area**		**Huu-ay-aht study area**	
Coarse-scale biological data	NOAA	Arrowtooth flounder Dover sole English sole Flathead sole Lingcod Pacific Hake Pacific Halibut Pacific Ocean Perch Petrale Sole Starry Flounder Walleye Pollock Widow Rockfish Coonstripe shrimp Dungeness crab	Fat gaper clam Geoduck Manila clam Northern pink shrimp Ocean pink shrimp Pacific gaper clam Pacific Littleneck clam Pacific Razor Clam Pinto Abalone Red king crab Sidestripe shrimp Spot shrimp Sperm whale	Arrowtooth flounder Dover sole English sole Flathead sole Lingcod Pacirfic Cod Pacific Hake Pacific Ocean Perch Sablefish Spiny dogfish Starry Flounder Walleye Pollock Widow Rockfish Dungeness crab Fat gaper clam Geoduck	Manila clam Ocean pink shrimp Pacific gaper clam Pacific Littleneck clam Pacific Razor Clam Pinto Abalone Sidestripe shrimp Spot shrimp Bairds beaked whale blue whale Cuvier's beaked whale Fin whale Hubbs’ beaked whale Sperm whale Stejneger's beaked whale
Local ecological knowledge and traditional fishing area data	Fisheries and Oceans Canada, Gitga’at First Nation	Abalone – historical fishing areas Chinook fishing areas Chum fishing areas Clams fishing areas Cockles fishing areas Coho fishing areas Crabs fishing areas Eulachon fishing areas Halibut fishing areas Mussels fishing areas Octopus fishing areas Pinks fishing areas Prawns fishing areas Sablefish fishing areas	Scallops fishing areas Sea cucumber fishing areas Sea prunes fishing areas Seaweed fishing areas Slipper fishing areas Snapper fishing areas Sockeye fishing areas Urchins fishing areas Herring spawn Salmon holding Seals Dall’s porpoise Orca Pacific white-sided dolphin	*None available*	

	Data source	Gitga'at study area		Huu-ay-aht study area	
Fisheries data	Fisheries and oceans Canada	Crab catch Geoduck catch Groundfish catch Prawn catch Red urchin catch	Sablefish trap catch Schedule 2 (groundfish) catch Sea cucumber catch Shrimp trawl catch ZN catch (rockfish)	Crab catch Geoduck catch Groundfish catch Prawn catch Red urchin catch	Sablefish long line Sablefish trap catch Schedule 2 (groundfish) catch Shrimp trawl catch ZN catch (rockfish)

Appendix 2: Behavioural Research Ethics Board approval

The University of British Columbia
Office of Research Services
Behavioural Research Ethics Board
Suite 102, 6190 Agronomy Road,
Vancouver, B.C. V6T 1Z3

CERTIFICATE OF APPROVAL- MINIMAL RISK RENEWAL

PRINCIPAL INVESTIGATOR:	DEPARTMENT:	UBC BREB NUMBER:
Amanda C.J. Vincent	UBC/College for Interdisciplinary Studies/Fisheries	H05-80810

INSTITUTION(S) WHERE RESEARCH WILL BE CARRIED OUT:

Institution	Site
UBC	Vancouver (excludes UBC Hospital)

Other locations where the research will be conducted:
N/A

CO-INVESTIGATOR(S):
Natalie Ban
Chloe Shen

SPONSORING AGENCIES:
Mountain Equipment Co-op - "Does It Matter How We Select Marine Reserve Sites?"
Social Sciences and Humanities Research Council of Canada (SSHRC) - "Conservation Effectiveness of Stakeholder Selection of Marine Protected Areas (MPAs)" - "Does It Matter How We Select Marine Reserve Sites?" - "Comparing the Efficacy / Conservation Potential of Stakeholder-Driven vs Data-Driven Marine Reserve Selection in BC" - "Selecting Ecologically Viable and Socially Acceptable Parks"

PROJECT TITLE:
Does It Matter How We Select Marine Reserve Sites?

EXPIRY DATE OF THIS APPROVAL: October 25, 2008

APPROVAL DATE: October 25, 2007

The Annual Renewal for Study have been reviewed and the procedures were found to be acceptable on ethical grounds for research involving human subjects.

Approval is issued on behalf of the Behavioural Research Ethics Board