AN EVALUATION OF THE OPPORTUNITIES AND IMPEDIMENTS IN MANAGING QUOTA FISHERIES FOR BIODIVERSITY

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Abstract

Emerging political, ecological, and social priorities support inclusion of biodiversity conservation on national research and management agendas. Meeting biodiversity conservation objectives, however, will be difficult for fishery management systems that traditionally rely on the single-species, single-population "stock concept". My dissertation examines four scientific and institutional challenges to broadening the scope of fisheries management to include controlling fishery impacts on biodiversity.

First, despite broad recognition of its importance in ecosystems, there is no single definition of biodiversity that can be used in tactical fisheries management. I recommend extending single-species approaches to include diversity within populations across space as a first step toward biodiversity-based management.

Second, many existing data collection programs are not structured to account for spatial diversity within fish populations. I use the case of Pacific herring (*Clupea pallasi*) in the Strait of Georgia, British Columbia, to illustrate how a monitoring program designed to estimate biomass of a spatially structured population generates management vulnerabilities and opens the system to disputes over biodiversity conservation.

Third, many fisheries management systems knowingly ignore spatial diversity in fish populations. The management implications may include a loss of biodiversity and over-fishing of certain components of the population. I developed a closed-loop simulation model based on the dynamics of British Columbia herring populations and fisheries to evaluate the consequences of single-species management of spatially diverse fish populations. I demonstrate that the impact of this approach can not be inferred from the characteristics of the population or the scale of management. Depending on the nature of the population and the fishery, well mixed populations may be more vulnerable to overfishing than spatially discrete populations.

Fourth, reduced availability of funding for fisheries science may stifle innovation and reinforce the use of single-species approaches. I document shifts in Canadian science policy that have shifted the funding of public science in favor of oceans and ecosystems, and have required the fishing industry to offset cuts to fisheries science budgets. This funding model may restrict the nature and scope of fisheries research in Canada.

Keywords: Pacific herring · spatial diversity · simulation modelling · Canadian science policy · management

Subject terms: Fishery management· Stock assessment · Natural resources

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General Introduction

The Fisheries Management Problem

The term 'fisheries management' is frequently associated with phrases such as 'paradigm shift', 'holistic approach', 'precautionary principle', 'strategic planning', and 'ecosystem-based management' in the current literature (Sinclair et al. 2002). These phrases refer to the proposed solutions to what has come to be known as the 'failure' of modern fisheries management - the inability to meet the dual objectives of maximizing economic returns while conserving the resource for future generations (Stephenson and Lane 1995). These solutions are anchored in the concept of sustainable development, which recognizes that human well-being is both directly and indirectly dependent on the maintenance of ecological processes and components (FAO 1999). Management for sustainable development differs from traditional fisheries management in having more dimensions that are represented in the problem set: ecosystem, social, economic, and cultural objectives supercede maximum biological and economic yield in this new paradigm (Garcia and Staples 2000). Recognition of the need for a holistic and strategic approach to management emerged within fisheries science over 30 years ago (Larkin 1977), however, the management failure has continued and its effects may have amplified in recent years (Myers and Worm 2003). Progress has been hindered not by crippling scientific uncertainty, but by a singular focus on controlling fish stock production and removals, and by setting management targets that may be unrealistic and unlikely to achieve sustainable fisheries (e.g., the familiar goals of 'maximum sustainable yield' and 'maximum economic yield') (Holling 1973, Larkin 1977, Ludwig 2001, Wilen 2000).

The Shifting Scope of Fisheries Management

Fisheries management developed under a biologically-based paradigm, which is reinforced to a large degree by current management structures (Rice 2003). However, international and federal legislation is changing to highlight the need to place fisheries management within an integrated ecological, socio-economic, and institutional context (e.g., UN Convention on Biological Diversity, FAO Code of Conduct for Responsible Fisheries, Canada's Oceans Act, Canada's Fisheries Act, Jennings 2005). In practice only indicators of stock biomass and fishing mortality are regularly applied in fisheries management: these indicators reflect either a minimum level of stock biomass or a maximum acceptable fishing mortality rate. These indicators are often incorporated in harvest control rules or other forms of rule-based decision making (Cox and Kronlund 2008). Many control rules are based on biomass-related indicators of fish stock status, and in some cases this approach has been legislated or incorporated into national policy (Hilborn 2002). These institutional practices have fostered a narrow focus on biomass by fisheries scientists, often to the exclusion of other criteria related to separate dimensions of the system (Hilborn 2002).

The narrowing scope of fisheries management appears to stand in direct contrast to the general call for broader, holistic management, and it is difficult to see what role ecological or other indicators could play in the decision making process. However, apart from the control function, indicators can provide important information on system performance relative to the management objectives. This separate 'audit' function of such indicators has become obvious in recent years, as fisheries scientists have begun to incorporate evaluations of the monitoring and control systems (the management strategies) into their analyses (Cox and Kronlund 2008, de la Mare 1996, Butterworth and Punt 1999). While it is common to use the same (biomass-based) indicators in both the control and audit function in these types of evaluations, the indicators could (and perhaps should) be separated (Rice and Rivard 2007). By recognizing the dual role of indicators, one can begin to see how alternate sources of information might be explicitly used to guide management decisions. Rice and Rivard (2007) identify the British Columbia (B.C.) Pacific salmon (*Oncorhyncus nerka*) fishery as a case in which the information used to audit management performance (i.e. stock assessment estimates of adult returns) differs from the information used to set catches (i.e. test fishery catch rates). In the B.C. Pacific trawl fishery, multi-species interactions are incorporated in the control function as bycatch caps on non-target species (Rice and Rivard 2007). These examples suggest that suites of indicators are already used to manage certain fisheries, and that different indicators are applied in different ways. Separate from these established examples, the effects of fishing on populations can be measured using age composition, spatial distribution of the population, and perhaps even the status of other species (Hilborn and Walters 1992, Swain and Sinclair 2000). The challenge in terms of implementation of a more holistic management approach is to identify a small number of indicators that directly relate to the stated management goals, in order to ensure that the monitoring and evaluation system is tractable and relevant to the objectives.

Thesis Overview

In this thesis I proceed on the assumption that the conservation of biodiversity is fundamental to fisheries management, and will be a central component in future management strategies. Three lines of evidence support this assumption. First, the conservation of biodiversity has been adopted as a global benchmark for successful fisheries management by eco-certification organizations such as the Marine Stewardship Council, which seek to influence consumer purchasing power in order to promote environmentally sustainable harvesting practices by the fishing industry. The influence of such groups is intensifying as they gain economic power - ultimately they may set global-scale objectives and standards for fisheries science. Second, the shifting economic (and social) context within which fisheries management occurs has drawn the attention of politicians and national-level policy makers, as indicated by this statement made by a member of the Canadian House of Commons:

> *"My duty is to protect biological diversity, this principle is now clearly spelled out as a pillar of proper fisheries management."*
>
> *Hon. Loyola Hearn, Minister of Fisheries and Oceans Canada December 19, 2006*

The emerging economic and political imperatives are expected to redirect future national-level research agendas toward biodiversity conservation over the next 10 years, in order to better align fisheries science with public expectations for resource management, this shift will likely occur at the expense of research in support of single-species fisheries management (Rice 2003). The anticipated reduction in the availability of government funding for single-species management activities will likely be addressed by increasingly popular cost-recovery and cost-sharing agreements between government and the fishing industry (Schrank et al. 2003). These partnerships are intended to apply the 'user-pay' approach to fisheries research and to foster greater collaboration between stakeholders, managers, and scientists. It is therefore reasonable to expect that biodiversity will feature prominently on the publicly-funded national research agenda. The third line of evidence is that the theoretical basis for fisheries science recognizes diversity within fish populations across space (Ricker 1973, Stephenson et al. 2009), and this theory has recently been validated by empirical evidence, which demonstrated the long-term conservation and economic benefits of maintaining biodiversity in salmonid population aggregates (Schindler et al. 2010). Therefore, given the emerging importance of this topic, it is critically important to determine how existing aspects of fisheries science and management can be extended to generate, and more fully include, indicators of biodiversity, and to test the quality of management decisions with respect to biodiversity-based objectives. That is the overarching goal of my work.

My thesis is structured in 4 chapters. In Chapter 1, I review the theory and justification for preserving biodiversity. This chapter is intentionally broad, both in the literature examined and the evaluation of the rationale for including biodiversity-type objectives in fisheries management. By adopting this level of inquiry, my aim is to provide a common foundation for the subsequent chapters. In Chapter 2, I present an initial evaluation of the existing biomass-based management regime for Pacific herring (*Clupea pallasi*) in the Strait of Georgia, British Columbia (B.C.). Similar to many other fisheries, management of the B.C. herring fishery is focused on maintaining stock biomass within pre-specified limits - the spatial distribution of the stock is not explicitly treated in the management or assessment process. I explore the reasons for and possible implications of this management approach, given uncertainties in herring survey planning, population structure and spatial behaviour. In Chapter 3, I develop a closed-loop simulation model based on the key dynamics of B.C. herring populations and fisheries, and evaluate the consequences of applying a single, homogeneous-population assumption to monitoring, assessment, and biomass-based management of fisheries that target spatially diverse fish populations. Chapter 4 marks a departure from ecology into policy, and builds on the theme of user-pay approaches to fisheries management. In this chapter I examine how a shift in Canadian science policy toward industry co-management and cost recovery may have inadvertently biased the information provided to fisheries managers in favor of environmental explanations for fish population decline (as opposed to fishing) and/or failure to recover from low levels of abundance. I suggest that insufficient and variable funding for fisheries science in Canada is a significant problem that has emerged from policies intended to devolve responsibility for management activities (including science) to the private sector, while re-directing the focus of public science toward issues of ecosystems and climate change.

Chapter 1

Biodiversity and the Future of Fisheries Science

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1.1 Introduction

An immediate challenge facing fisheries scientists is to expand the scope of our research beyond single species in order to achieve a holistic, ecosystem-based approach to management. Central to ecosystem-based management is the notion of biodiversity conservation, which is the primary concern of groups such as the Marine Stewardship Council and other environmental non-governmental organisations. The influence of such groups can be expected to intensify as they gain political and economic power

- ultimately their influence may set global-scale objectives and standards for fisheries science. In these early years of shifting political context, I believe that fisheries scientists must think critically about what they are being asked to do, and identify inconsistencies between what is politically desirable and what is scientifically feasible. In my opinion, the management of marine biodiversity is a very good example of one such inconsistency. In this chapter I attempt to define biodiversity based on a directed review of the ecological and fisheries literature, in order to clarify the ecosystem-level objectives of fisheries management. However, it appears that the ecological theory of biodiversity conservation is not, and may never be, well developed. It is unclear how to reconcile single-species stock assessment, harvest management and 'biodiversity' conservation. I believe that the willingness of fisheries scientists to set aside a sophisticated (albeit, single species) theory of resource management in favour of a grander (and more complicated) approach may require further consideration. After all, as discussed later in this chapter, the theoretical basis of fishery science recognises diversity within fish populations across space (e.g., Ricker 1958), and as such, single-species theory is directly relevant to implementing biodiversity policy.

I expect that fisheries scientists in the future will revisit and adapt single-species theory to account for the fact that many fish populations are naturally spatially discontinuous and heterogeneous. This natural state is advantageous because it spreads exposure to external perturbations across space and time, enabling the population to maintain production in spite of changing environmental conditions (Hilborn et al. 2003). When viewed in this context, it is evident that important aspects of fish populations are not captured by existing, abundance-based assessment approaches, and some measure of spatial diversity within populations is required. This is the definition of biodiversity that requires the attention of fisheries scientists, and which will emerge as our understanding of marine populations evolves. By shifting emphasis to systems-level characteristics of fish populations such as spatially-complex dynamics, interactions, and distribution, we will see demonstrable improvements in both single-species and ecosystem-based fisheries management. At a minimum, this approach will indicate whether ecosystem-based management is achievable in principle.

1.2 What is Biodiversity?

In spite of prolific use of the term, there is no consensus on the definition or the value of biodiversity. The term 'biodiversity' has been used to describe various aspects of biological populations, including the number of species present in a community, genetic variability, and diversity among ecological systems (Harper and Hawksworth 1995). The lack of consistency exists for two reasons: first, biodiversity is a pseudocognate term in that most people assume the definition to be intuitive and that others automatically share their understanding; and secondly, several perspectives on the meaning of biodiversity have developed over time (Gaston 1996). These perspectives can be classified as those in which biodiversity is approached as a concept, those that consider biodiversity to be an entity that can be measured, and those that consider it to be a socio-political construct (Gaston 1996).

The competing definitions of biodiversity all emphasize its multi-dimensional nature, and build on a basic theme that equates biodiversity with the 'variety of life' (e.g., McNeely et al. 1990, Wilson 1992). A commonly cited definition is that of the Convention on Biological Diversity (UNEP 1992):

"the variability among living organisms from all sources, including inter alia, terrestrial, marine and other aquatic systems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems"

Although this characterization has certain intuitive appeal, it does not advance a precise understanding of the concept. This language unfortunately reflects the majority of biological studies on the topic. The 'variety of life' is frequently deconstructed into smaller categories of genetic, species or taxonomic diversity, and ecosystem diversity, in an effort to facilitate its study (Harper and Hawksworth 1995, McAllister 1991). Some authors divide the categories further into genes, populations, species, assemblages, and whole systems (Soule 1991). Regardless of the scheme, all classifications emphasise hierarchies both within and between levels in the system (e.g., Noss 1990). Biodiversity is commonly defined in terms of these hierarchical entities rather than the processes underlying the observed patterns (Smith et al. 1993). Viewed from this perspective, biodiversity is seen to be the singular end of the evolutionary process, rather than a by-product of evolution (Zeide 1997). In focusing exclusively on units such as species and genes, the basic concept of biodiversity is insufficient because it ignores the fact that biodiversity is the biological response to a variable environment (Hengeveld 1996).

Most biologists tend to avoid the problem of defining biodiversity and instead approach it as something that is generally understood to be both real and measurable. There are volumes written on the loss of biodiversity and how to rebuild and conserve what remains. Statements expressing concern over what biologists refer to as the sixth mass extinction on our planet are common (Perrings et al. 1995), as are biodiversity evaluations that are presented in terms of some numeric value (e.g., Wilson 1988).

Biodiversity is firmly tied to the notion of preserving the natural environment, and there is a growing public perception that biodiversity is 'good' and should therefore be maintained. In this sense, biodiversity is not a neutral scientific concept, rather, it is either imparted with a value or it is perceived as representing the value of nature (Gaston 1996). At its most basic level, the problem of nature conservation is fundamentally linked to human population growth and resource consumption. Thus, the conservation of nature is more of a political problem than an ecological one because it requires value judgements to be made regarding what biodiversity is good for and how best to allocate scarce conservation resources (Vane-Wright 1996).

1.3 The Ecological Justification for Conserving Biodiversity

The relationship between biodiversity and characteristics of ecosystems such as productivity, stability, resilience, and function, has been a key area of focus in ecological research for at least four decades, but relatively few unifying principles for management have emerged (Hughes and Petchey 2001). This may be because biodiversity is most often treated as an academic area of study instead of a broadly applied one. In addition, there is little research directed at predicting how human activities are likely to influence biodiversity, or at the importance of such an effect relative to other human impacts on ecosystems (Srivastava 2002). Much of the experimental work that has been conducted focuses on biodiversity in terrestrial plant communities, and there is substantial uncertainty as to how these results might be generalized for other species and ecosystems (Loreau et al. 2001). In addition, it is difficult to reconcile both the approach and the results of small-scale and large-scale observational experiments. However, taken in aggregate, the empirical evidence suggests that biodiversity is an important predictor of ecological processes at small scales, and can be expected to decrease in importance at regional and ecosystem-level scales (Loreau et al. 2001).

1.3.1 Biodiversity and stability

Our current understanding of the relationships between biodiversity and ecosystems emerged from the two branches of ecology: community ecology, which focuses on the interrelationships between species (competition, predation, parasitism, etc.), and ecosystem ecology which focuses on energy flows and nutrient cycling (Holling et al.

1995). The majority of historical work on the relationship between biodiversity and ecosystems is firmly cast in the community ecology approach. Prior to the 1970s ecologists believed that stability increased with the number of species in an ecosystem (species richness) (Elton 1958). This conclusion was supported by repeated observations that the population densities within simplified terrestrial communities were more variable than those within complex, diverse communities (McCann 2000). However, subsequent work showed that increased diversity tends to either destabilize community dynamics, or is of little importance to stability (May 1973). These and other disparate results have come to be known as the 'diversity-stability debate' and have created much uncertainty within the scientific community, because natural systems are observed to be both highly unpredictable and diverse. The field of ecology is currently replete with studies searching for diversity-stability relationships, and the results appear to depend greatly on the species and scale of examination (Hughes and Petchey 2001, McCann 2000). Although ecologists have yet to identify a universal diversity-stability relationship that can be applied across scale, species, and ecosystem, it appears that high diversity on average yields greater ecosystem stability (Loreau et al. 2001, McCann 2000). However, the persistence of an ecosystem results not from its biodiversity per se, but rather from the fact that it is comprised of species that can respond differently to change. This idea is referred to as the 'insurance hypothesis' - biodiversity provides a buffer against environmental variations because the mixed responses of different species mean that the aggregate characteristics of ecosystems are stabilized (Loreau et al. 2001). Confusion also arises from the fact that the definition of stability used in many analyses focuses on systems behaviour and the notion of achieving constancy, a tradition borrowed from classical physics and engineering (Holling 1973). In keeping with the classical approach, both the dynamics of populations and the physical environment are defined with respect to an equilibrium level, and the environment is seen to affect organisms but is not affected by them. In reality, this definition of stability is insufficient for marine systems which have been shown to shift rapidly between alternate states (e.g., Benson and Trites 2002).

1.3.2 Biodiversity and resilience

The relationship between resilience and biodiversity can be represented in terms of stability landscapes in which species-rich environments have landscapes with deep pits implying high local stability whereas species-poor environments have landscapes with shallow pits and low local stability (Peterson et al. 1998). However, ecosystem resilience is not a simple function of the number of species present at a given time (Elmqvist et al. 2003). There are competing models of how increases in species richness relate to increases in local stability of ecosystems. The first model of how ecosystems function and organize around species is the 'redundant species hypothesis', which proposes that there is a minimum level of biodiversity that is required to maintain ecosystem processes, and species additions or losses above this level have minor impacts (Walker 1992). Under this perspective a limited number of keystone organisms drive the critical ecosystem processes (drivers) and other species exist in the niches created by the keystone groups (passengers) (Holling et al. 1995). Most ecological function resides in the keystone species, whose presence or absence determines the state of an ecosystem (Walker 1995). Empirical evidence highlights the importance of keystone species in both natural and managed systems - they maintain critical processes in ecosystems under stress, even as the species composition changes (Folke et al. 1996). However, identification of keystone species is problematic because depending on the state of the ecosystem, species can be either a driver or a passenger (Lawton 1994, Lister 1998). The second model is the 'rivet hypothesis' which suggests that all species contribute evenly to ecosystem function. Similar to rivets on an airplane wing, the effects of losing a small number of species (rivets) are buffered by the overlapping functions of other species, but continuous loss ultimately leads to collapse of the system (the airplane wing falls apart) (Ehrlich and Ehrlich 1981). In this model, an ecological function is not lost until all species performing that function are removed from the system (Peterson et al. 1998). As such, compensation by the remaining species masks degradation of the ecosystem and a loss of adaptive capacity. The third model is the 'idiosyncratic response hypothesis', which proposes that it is not the number of species that is important, but the particulars of the species and the ecological history of a region that is important for ecosystem function (Lawton 1994, Kunin and Lawton 1996). However, ecosystem structure and function can be sustained at the regional scale, independent of the mix and relative abundance of species that are present (Schindler 1990). This suggests that groups of organisms are more critical than single species to the maintenance of ecosystem functions.

Ecosystems generally possess considerable functional redundancy (i.e. the number of alternative species that can provide a particular function following a disturbance), which acts to stabilize ecosystem processes. However the loss of a functional group (a decrease in functional diversity) can drastically alter ecosystem functioning (Folke et al. 2004, Peterson et al. 1998). In addition to diversity in the function of groups of species, the adaptability of the groups that are present (that is, the diversity of responses to environmental change that exists within a functional group) plays a key role in determining the resilience of ecosystems (Elmqvist et al. 2003). This 'response diversity' is governed by species and population diversity. For example, sockeye salmon in Bristol Bay, Alaska exhibit a broad range of population distributions and life history strategies whose productivity varies in relation to climate. In spite of high variability in relative abundance, the populations in aggregate have sustained high productivity over time because of high response diversity (Folke et al. 2004). In short, the stock aggregate has maintained its resilience.

1.4 The Economic Justification for Conserving Biodiversity

Ecologists and economists both recognise that human beings can not live in isolation - we are critically dependant on other organisms for our survival. In terms of value, plants and animals are important not only because they house the genetic library of the planet, but also because they provide food, timber, industrial resources, and medicines; in addition, they sustain the flow of ecological services (e.g., waste assimilation and filtering of pollutants) that are used by humans (Ehrlich and Ehrlich 1992, Kunin and Lawton 1996). In this sense, biodiversity serves both the direct and indirect needs of society, and ecosystems are fundamental 'factors of production' that are being threatened by human activities (Folke et al. 1996). The value of biodiversity from an economic perspective is therefore intimately linked with its functional role in maintaining ecosystem processes.

Economists view the preservation of biodiversity as a form of insurance against lost utility (in terms of use and services) of organisms at some future date (Williams and Humphries 1996). This notion is captured in the 'option value' of species (Faith 1995). Option value corresponds not just to the unknown future values of known species, but also to the unknown values of unknown species. A general view of the option value of biodiversity is that it maximizes the capacity of humans to adapt to ecological change (Reid 1994). As such, the basis for biodiversity conservation from an economic perspective is to maximize species' abilities to adjust to a changing environment (Williams and Humphries 1996). The value of biodiversity therefore lies in the capacity of organisms to adapt through natural selection thereby maintaining options for future generations of humans. Because selection acts at the level of character (feature), the focus turns from attempts to value species to attempts to value the features of species (Williams and Humphries 1996). Fundamental to the idea of an option value is a high degree of uncertainty about both the environmental conditions and the realized value of species under those conditions (Faith 1995). At any given time, an individual species that contributes more novel features to a protected subset is of greater value than another that contributes fewer features (Faith 1995). However, uncertainty with respect to the future value of organisms greatly complicates the rationale for differentially ranking them for conservation. This point has sparked a substantial, and somewhat circular, debate on the point of weighting and non-weighting of both the features and the species that own them. Most valuation systems are inherently subjective (Reid 1994), and in order to avoid any subjectivity it is necessary to consider all species as equal (Wilson 1992). However, if the features of species at lower levels (e.g., genetic and phenotypic characteristics) are weighted equally, it is inevitable that species at higher levels (e.g., species assemblages) will be weighted differentially because certain species will contribute relatively more to the preserved set of features than others (Faith 1995, Williams and Humphries 1996).

1.5 “Biodiversity” as a Management Objective

The biodiversity objectives for ecosystem management are somewhat ambiguous. According to some authors the objective is to protect the species and ecosystems most at risk of extinction (Hansen et al. 1999), others promote maximizing species richness within a particular geographic region (Smith et al. 1993), while others focus on maintaining a range of ecosystem services (Folke et al. 1996). In addition to a lack of clear objectives, the preservation of biodiversity is thus far hampered by the absence of a well defined unit of analysis that might be used to formulate an objective function for biodiversity management (Weitzman 1995).

In principle, biodiversity can be measured at a variety of hierarchical levels, from molecular and genetic (alpha diversity), to the ecosystem level (beta diversity) (Harper and Hawksworth 1995). However, there is no consensus on which is most important for assessment and conservation because a loss of biodiversity at any level may represent a loss of future opportunities (i.e., option value) (Agardy 2000). Conceptual and methodological problems exist even at the species level, where ecologists debate whether species counts (i.e. species richness) are sufficient, or whether each species should be weighted by its relative abundance in order to reveal the relative effects of rare and common species, as in Simpson's index and the Shannon-Weiner measure of 'effective number' (Baumgartner 2004). These abundance-based indices do not explicitly account for the uniqueness of species. In contrast, indices that have been developed by economists such as Weitzman 1992Weitzman 1995 and Solow et al. 1993, stem from the notion of product diversity and focus on measuring the differences between species using metrics that incorporate species richness as well as the features of species (Baumgartner 2004, Faith 1995). It is important to note that measures of species richness are non-stationary because of the taxonomic inflation phenomenon wherein scientific progress drives the elevation of lower taxa to the rank of species (Knapp et al. 2005).

No approach to measuring biodiversity fully captures all of the characteristics of species within an area. As such, the choice of index requires a prior judgement on the purpose of biodiversity and the value of its components (Baumgartner 2004). Further problems with quantifying species diversity in a given area include the biasing effect of sampling effort on biodiversity measures (Gotelli and Colwell 2001), the effects of species movements through invasion, migration, and changes in distribution (Hawksworth 1995), and questions pertaining to the boundaries of the system of interest (Gaston and Williams 1993). The measurement problem is particularly pronounced in the marine environment, where organisms have wider and more variable geographic ranges than on land (May 1995), and where significantly less effort has been directed at generating species inventories, particularly for deep-sea environments (Society 2003). The enormity of the problem has led many researchers to propose proxy indicators for biodiversity. The use of indicators is based on the idea that small subsets of species, habitats, or ecosystems can be used to expedite biodiversity assessments, and may remove the necessity for a full census of global diversity (Pearson 1995). However, the search for appropriate indicators is problematic (e.g., Gaston and Williams 1993, Pearson 1995) and in some cases debate on this point threatens to eclipse the underlying issue. For example, the criteria for selection of indicator species are usually based more on legal and political imperatives rather than on scientific assessments (Pearson 1995). This turns the focus to the species itself rather than what it is supposed to be indicating. In addition to these fundamental limitations of the analytical approach, many researchers conclude that there is no single spatial or temporal scale at which to describe the natural system because the scale of interest depends greatly on the chosen species (Bunnell and Huggard 1999, Levin 1992).

It is important to recognize that most conservation and recovery efforts to date focus on those species that have already been identified as being under threat of extinction. This raises fundamental questions related to prioritizing conservation efforts, and the choice and identification of critical species for conservation (e.g., keystone species versus charismatic species) (Walker 1995). Given these constraints, it is reasonable to question the practicality of focusing on species when tackling the greater problem of biodiversity conservation. The species-based endeavour is not without merit; however, if one accepts that the 'biodiversity crisis' extends beyond the massive extinction of species to include all ecologically and socially undesirable changes in the composition and functioning of ecosystems, then the scope of the traditional species-based approach is not sufficiently general to account for all objectives. In addition, the number and variety of definitions and measures of biodiversity that form the basis for conservation initiatives indicate that biodiversity remains a poorly conceived and immature concept (Hengeveld 1996). Furthermore, the level of biodiversity is constantly changing because it is bound to the state of the ecosystem. Therefore, even if there was an objective measure of biodiversity, fixing target levels for it would largely be an academic exercise (Kampinen and Walls 1999).

1.6 A Fisheries Science Perspective

Objectives for biodiversity conservation are increasingly identified in terms of population viability and probabilities of extinction for closed, identifiable populations of rare or endangered species (e.g., Nicholson and Possingham 2006). Application of this approach is hampered by insufficient understanding of the process of extinction of marine populations, by a lack of data on population abundance over time, and by the fact that fish populations are seldom closed to migration (Wainwright and Waples 1998). This extinction-centric approach to management is somewhat at odds with traditional fisheries science, which is concerned with preserving stock productivity and the maintenance of future fishing opportunities for relatively abundant fish species. Adopting the extinction-centric approach to conserving biodiversity requires a concomitant shift in focus to the recovery of rare, depleted fish species and may not be necessary or appropriate for populations that continue to sustain fisheries. The challenge that remains for fisheries science is to develop a tractable interpretation of biodiversity that is relevant for the management of commercially exploited fish species. The scale of biodiversity management will be constrained in the near term by the limited application of genetic techniques to fish populations - the preservation of genetic or even phenotypic diversity is as yet beyond the scope of fisheries management. Similarly, both the theory and application of ecosystem-based management are under development. Therefore, in an operational sense, the unit of interest for fisheries management remains the population. Defining a population for fisheries management is a difficult and research-intensive aspect of fisheries science. The 'stock concept' has been debated for over 100 years, and a variety of definitions exist in the literature (Begg et al. 1999). In general, the recent definitions admit that a stock is largely a construct of management, and does not represent a single, homogenous group of fish

(Begg et al. 1999). The key uncertainty in applying the stock concept is the ecological implication of incorrectly treating a diverse group of fish as a homogenous unit for assessment and management. For example, the biological limits within which fisheries can operate are determined by the production function, or stock-recruitment relationship, of the underlying 'stock' (Mace 2001). Stock assessments typically proceed on the assumption that this relationship is stationary across both space and time. However, for spatially structured populations, changes in productivity of the substocks that result from harvesting or natural disturbance can lead to changes in the aggregate recruitment relationship that can never be fully understood or anticipated (Walters 1987). The assessment and management issues stemming from the stock aggregate situation parallel those of mixed-stock fisheries, where sub-populations that contribute to the total recruitment have a range of productivities. When treated as a unit stock, the lower productivity populations will invariably be over-fished and decline in abundance, resulting in a lower weighting of the unproductive populations in the aggregate recruitment function. This leads to a pathology in which the recruitment function actually appears to show increasing productivity as the stock declines, leading to recommendations of increased exploitation rates (Ricker 1973).

Sub-structure within fish stocks is represented by either spatially-discrete units or different life history types within a population. Species that demonstrate within-population structure include sockeye salmon (*Oncorhynchus nerka*) (Hilborn et al. 2003, Pacific herring (*Clupea pallasi*) (Hay and McCarter 1997), Arctic char (*Salvelinus alpinus*) (Secor 1999), and Atlantic herring (*Clupea harengus*) and cod (*Gadus morhua*) (Smedbol and Stephenson 2001). This aspect of fish populations has been noted since the early 19th century, but the high cost of collecting data on the compositional complexity of a population tends to preclude explicit consideration of spatial and sub-population-level impacts (Walters 1987). This omission presents a problem for fisheries, because management that does not account for population structure within and among stock complexes can lead to overexploitation of the stock components, an erosion of within-species diversity, and ultimate depletion of the productive potential of the stock aggregate (de la Mare 1996, Hilborn et al. 2003, Stephenson 1999). Based on these arguments, preserving within-stock diversity (response diversity) is a biodiversity-type objective that can be operationalized, as demonstrated to a degree in certain fisheries (Hilborn et al. 2003), and is directly consistent with both the theory of fisheries science, and the current call to maintain the resilience of populations and ecosystems.

Recognition of the importance of stock identification is manifest in widespread interest in tagging and genetic research programs as well as an increasing focus on spatially-explicit population models. However, in many cases it is unclear how this information can or should be incorporated into management. For example, does detailed information on stock structure easily translate to stock-specific management? To what extent does the management approach depend on the stock structure? What is the appropriate spatial scale of management? How much within-stock diversity is 'enough'? The answers to these questions are likely to be case specific, and conditioned on the local goals of fisheries management.

Fisheries management involves making decisions that balance resource conservation and exploitation, given imperfect information about the resource, the environment, and the resource users. In recognition of the highly uncertain nature of fisheries, levels of 'risk' and 'precaution' are now frequently provided in stock assessments. However, the management implications of such measures tend to be highly uncertain themselves. The evaluation of management procedures addresses this problem by formally testing the implications of uncertainties on the quality of management decisions (de la Mare 1998). Fundamental to this approach is the specification of a minimally realistic model of the exploited population that incorporates uncertainties in the key biological processes. However, spatial and multi-species interactions are rarely explicitly incorporated into such models (Punt 2006). Stock structure can be added to the list of important characteristics that are seldom addressed in these evaluations. Given that many fisheries are now moving toward formalized management procedures, fisheries scientists have a strategic role to play in identifying the important aspects of exploited populations that govern the productive capacity of the stock, and in developing methods for including these features in models of 'true' stock dynamics. The key population processes will likely include more than age-structure and apparent spawner-recruit relationships, particularly for population 'rich' species such as salmon, herring, rockfish, and cod, which exhibit numerous spawning populations within a single management unit (Stephenson 1999).

1.7 Science Requirements for Managing Within-Stock Diversity

Several challenges must be addressed before biodiversity conservation can be incorporated into the management of commercially exploited fish populations. Foremost is the identification of a metric that reflects relevant changes in the population. This requires knowledge of the aspects of population dynamics that support the ability of the sub-populations to respond differentially to disturbance (i.e. the maintenance of stock resilience). Fisheries ecologists emphasize the importance of the spatial distribution of the spawning stock, as it reflects evolutionary adaptations to the marine environment that maximize the survival of progeny (Sinclair 1988). In fact, recruitment success may be determined as much by spawner abundance as by the spawning distribution (deYoung and Rose 1993). The spatial distribution of the spawning stock may therefore be a reasonable proxy for stock diversity, provided that a broad spawning distribution reflects the varied responses of the sub-populations to disturbance. Taken together with information on abundance and age-structure, this information can augment the stock assessment by adding a dimension of interpretation that does not exist under standard, non-spatial approaches. It is important to note that the metric should reflect within-stock diversity, but may not necessarily directly measure the mechanism that maintains diversity. For example, the population structure of many marine fish species is determined by dispersal and homing dynamics (McQuinn 1997). From a management perspective, measuring the details of dispersal (i.e. the age-specific rate of straying) may be less important than obtaining an accurate measure of the spatial complexity of the stock, in part because the relationship between straying rate and stock resilience may be less direct than the relationship between spawning distribution and stock diversity.

Methods for incorporating stock diversity into harvest decision rules present another challenge for managing the diversity of commercially important fish species. A fundamental question related to harvest decision planning is the optimal scale of stock assessment. As discussed earlier, there is a level of heterogeneity in fish stocks that is assumed not to exist for stock assessment and management, in large part because the sub-population boundaries have not been reliably determined. Additional, practical considerations for conducting aggregate stock assessments include a reduction in data quality that would go along with disaggregating the data, and increased annual survey costs arising from the intensified sampling that would be required for fine scale stock assessment. It is reasonable to assume that resources for moving in this direction will be scarce, particularly given the additional fisheries management expenditures that are required for implementing finer scale assessment and harvest management (e.g., enforcement costs).

Given the high probability that sub-populations will continue to be assessed in aggregate, is there a way to include the measure of biodiversity in a harvest control rule? Adding a dimension of constraint for setting allowable catch is a minor problem when compared to the issue of setting the appropriate threshold for biodiversity. This is the point at which fisheries scientists must carefully consider how to apply the language of biodiversity conservation. For example, if the overarching objective is to maintain biodiversity, does this translate to a minimum number of spawning sites?

How is this number related to the aggregate stock production? Most current fishery harvest control rules are based on relative measures of depletion - can a similar method be applied to stock diversity? Must the threshold be absolute? The difficulty in identifying a threshold level of diversity is matched by the problem of implementing harvest policies for spatially structured stocks. The management questions arising from this approach pertain to the appropriate allocation of fishing effort among the various sub-populations and the tractability of a spatial approach to management, which will be determined by the type of fishery (i.e., a terminal fishery versus a fishery that operates on migratory aggregations), the fishery dynamics, and ultimately, the increased costs of management.

1.8 Conclusion

A directed review of the literature indicates that the justifications for conserving biodiversity that have featured prominently in the development of international conventions and policies are characterized by widely ranging hypotheses and disparate results. Nonetheless, these lines of argument have yielded both political and global economic imperatives that now set the agenda for fisheries management. Fisheries science has yet to feature prominently in the development of biodiversity policies and objectives, but given its unique position at the interface between ecological theory and application, I believe that fisheries science has much to offer in this regard. It has been observed that environmental problems arise from the negative net impact of many small decisions, including the focus on single species management (Odum 1982). In my opinion, the apparent incompatibility of traditional fisheries science and biodiversity conservation does not arise from a myopic focus on single species per se; rather, problems arise from a preoccupation with the level of production of a population, and insufficient consideration of spatial and sub-population-level impacts of management policies. The imperative to conserve biodiversity is no longer up for debate: there are now political, ecological, and economic justifications to do so. The challenge that remains for fisheries science in the future is to develop a tractable interpretation of biodiversity that can be rendered operational in the management of commercial fisheries.

The preservation of within-stock diversity (response diversity) is a biodiversity-type objective that is immediately relevant for fisheries management. This aspect of fish stocks is seldom explicitly included in stock assessments and evaluations of management strategies, but it may determine the ability of the stock aggregate to maintain its productive capacity under exploitation. Managing for within-stock diversity requires a measure of diversity, a method for incorporating the measure into harvest control rules, and a spatial approach to management that accounts for differences in productivity among population components. Fisheries science has developed a sophisticated theory intended to address questions as they relate to fish stock production. The theory should be extended to include within-stock diversity. The implications of, and necessity for, moving in this direction can be tested using existing methodology such as management procedure evaluations, which can be used to prioritize information requirements based on the key uncertainties in the fishery system. Additionally, by explicitly considering the feasibility of various spatial management regimes in light of the objectives of those interested in the fishery, this approach may provide a strategy for addressing the potential institutional mismatch that may exist between current management arrangements and those required to conserve population diversity.

Chapter 2

Spatial Dynamics, Uncertainty and Monitoring of Pacific herring Populations

2.1 Introduction

Resilience is widely considered to be an important property of marine fish populations, signalling their ability to adapt and persist over a broad range of environmental conditions (Hilborn et al. 2003). Spatial diversity within populations is an attribute that contributes to population resilience by raising the probability of reproductive success within and across spawning seasons, and preventing recruitment failure (Kerr et al. 2010, Ruzzante et al. 2006, Secor 1999). Scientific recognition of the long-term value of resilience has begun to shift the management paradigm in order to promote this feature of marine populations. However, it is often unclear if and how current management should change in order to accommodate the overarching goal of maintaining resilient fish populations. In this chapter I present an initial evaluation of an existing, biomass-based management regime in light of this objective. I use the case of Pacific herring in the Strait of Georgia (SOG), British Columbia (B.C.), Canada (Figure 2.1) to illustrate how a monitoring program designed to estimate biomass of a spatially structured population has generated a management vulnerability that opens the system to disputes arising from conservation concerns and issues of scientific credibility.

I begin this chapter by presenting data on SOG herring spawn timing and distribution. SOG herring appear to have experienced a dramatic contraction of the area occupied at the time of spawning, and in the timing of spawning migrations over a period of 50 years. These trends have been interpreted by some observers as relatively benign, natural events because of known limitations in the sampling program (discussed in a subsequent section), and because they were not matched by strong evidence of eroding genetic structure and depleted stock biomass (Hay et al. 2009). This interpretation reflects a management paradigm that is characterized by: (i) a narrow definition of conservation that is based only on total biomass, and which may not capture all indicators relevant to the conservation of spatially structured populations (Stephenson 1999), and (ii) a view of biodiversity that is based exclusively on genetic differentiation. These features of the SOG herring management system are inconsistent with Canada's commitments to the Ecosystem Approach to Fisheries (Garcia and Cochrane 2003) and with the broader ecological literature on spatial dynamics and population structure, which collectively indicate that biomass is an insufficient indicator of the status of marine populations, and that variable migration trajectories *within* a population contribute to its ability to adapt to variable environments. Any loss of spatial diversity - signaled by genetic, phenotypic, or behavioral homogenization - may therefore reflect reduced resilience to anthropogenic and environmental change (Hutchings et al. 2007).

Analysis of the spawn monitoring data is unfortunately limited by potential confounding between trends in herring spawning dynamics and the locations and timing of the spawn survey. Rather than attempt to disentangle these effects, my objective is to highlight the management implications of persistent uncertainty in the data. I present three competing hypotheses to explain the concentrating patterns of spawn timing and distribution, each of which presents a unique set of demands for the science and management of herring. The competing hypotheses are: (i) the spatial and temporal trends in the spawn data are accurate ecological measurements that may represent a loss of resilience in the stock, (ii) the spatial and temporal trends in the spawn data merely represent an ongoing contraction of the spawn monitoring program rather than actual change in the spawning behavior of herring, or (iii) some combination of hypotheses (i) and (ii). The conservation risk associated with failing to account for patterns of population structuring and spatial dynamics is evident in the collapse and failed recovery of Atlantic cod (Hutchings 1996). A similar explanation might apply for herring populations in Haida Gwaii, the Central Coast, and the West Coast of Vancouver Island, which remain at low stock biomass levels, despite annual fishing closures (Figure 2.1) (Schweigert et al. 2010). The biomass trajectories of herring stocks in the Prince Rupert District and the SOG exhibit general declines over the past decade, but currently remain above the biomass limit reference point used to close the fisheries during periods of low abundance (Schweigert et al. 2009).

2.2 B.C. herring

2.2.1 Spawning dynamics

Pacific herring spawn on intertidal and sub-tidal substrates during the winter and spring months in British Columbia. Adult herring migrate from offshore feeding grounds to inshore 'holding' areas several weeks prior to spawning (Haegele and Schweigert 1985), and move closer to their spawning locations as they reach maturity (Hay 1985). The precise mechanisms that trigger spawning are not well understood, but once it begins, egg deposition generally occurs as a single, rapid event in which all of the fish in an area spawn over a period of a few hours (Schweigert and Stocker 1988). This process can yield very dense depositions of eggs (millions of eggs per square meter) (Haegele and Schweigert 1985). Males release large quantities of milt which discolor the water, and provide a temporary visual record of the location and areal extent of egg deposition. Because the eggs remain attached to the substrate until they hatch, they can be counted and combined with the area of egg deposition to generate an estimate of spawning biomass based on assumptions of a 1:1 sex ratio and 200 eggs/gram of female body weight (Schweigert and Stocker 1988). The adults leave the area immediately after spawning; most are believed to return to the offshore feeding areas, but some remain near the spawning grounds throughout the year (Haegele and Schweigert 1985).

2.2.2 Spawn monitoring data structure and uncertainty

Pacific herring spawn survey data (1928-present) are maintained in the Herring Stock Assessment Database, Fisheries and Oceans Canada, Pacific Region. The data structure is hierarchical, reflecting a nesting of spawning locations (individual beaches and bays) within geographically larger sections, statistical areas, and ultimately, stock assessment regions (Figure 2.2 (a)) (Haist and Rosenfeld 1988). The survey records the location of herring spawn deposition, the date of the spawn, and the approximate dimensions (length, width, and estimated number of egg layers deposited). Two distinct 'eras' characterize the history of herring surveys in B.C.: (i) the surface survey era (1928-1986), during which time observations were made from beaches, boats, and the air by Fisheries Officers during routine patrol, and (ii) the dive survey era (post-1986), during which divers are deployed to obtain accurate sub-surface measurements of herring egg depositions. The goal of both survey approaches is to estimate absolute stock size of adult herring, but they yield two distinct types of data: the surface survey generated broad-scale, imprecise estimates whereas the dive survey generates spatially restricted, but locally precise estimates of egg deposition (Schweigert and

Stocker 1988). Superimposed on these known changes to the survey procedure are periodic shifts in the spatial coverage and detail of observation that have not been quantified, but which are believed to have introduced substantial measurement error into the earliest estimates of spawn deposition (Hay and Kronlund 1987). Shifts in the survey coverage arise from the fact that there were few fishery officers in the field and logistical support was considerably lower in the earliest (pre-1950) years of the Era 1 survey than in the later years (late-1970s and 1980s). Additionally, the pattern of fishing changed after 1970 to concentrate on pre-spawning aggregations of herring. This change in the fishery meant that there were more fishery and survey vessels operating in the vicinity of spawning herring, which increased the probability of detection and reporting of spawning events in areas that were not previously surveyed (Schweigert and Stocker 1988). These changes in the sampling procedure are believed to have introduced systematic bias into the spawn survey data (for a thorough review see Schweigert and Stocker 1988).

The shift from survey Era 1 to Era 2 may have added further uncertainty to the herring spawn survey data because a '0' in the Era 1 (post-1970) data is more likely to reflect a location that was surveyed and found to have no spawn than in Era 2, where a '0' may reflect either a location with no spawn or a location that was not surveyed. No distinction is made between these options in the database. Compounding this uncertainty are two potential recording errors in the data: a spawning location may be assigned multiple names, and/or a location can be assigned to multiple sections, which further confuses the data hierarchy (Doug Hay, *pers. comm.*, Figures 2.2 (b) and (c)). This issue is not easily resolved because it is not documented. However, it is believed to be significant enough to limit the scale of "useable" spatial information to the level of spawning section. Such treatment of fine-scale spatial information reflects the regional-scale management paradigm under which the B.C. herring monitoring program was developed.

2.2.3 Trends in spawn distribution and timing

Catch and spawning stock biomass estimates are used to describe the dynamics of the SOG herring population for stock assessment and management (Figure 2.3 (a)). High catches were characteristic of the reduction fishery, which operated on large overwintering aggregations of herring until the late 1960s. This fishery harvested high volumes of herring that were processed into low-value fishmeal and oil. The high fishing mortality rates associated with the reduction fishery prompted a population collapse and fishery closure from 1967 to 1971. The sac roe fishery began in 1972. It uses gillnet and seine gear to target pre-spawning aggregations of herring and it is characterized by substantially lower catches than the reduction fishery. The spawning biomass trajectory reflects the initial decline and subsequent rebuilding of the SOG population under the roe-herring management system. However, the stock has been in decline for the past 10 years (Cleary et al. 2009). These trends in catch and biomass provide critical information on the temporal dynamics of herring in the SOG, but because they are aggregated across space and season, they provide no avenue for evaluating population-level resilience.

However, the additional consideration of indicators of herring population resilience (within-season and spatial information) does not permit a simple evaluation of population status. The range of observed spawn deposition times in the SOG narrowed in the early 1970s, eliminating the early-season spawns, it shows no indication of recovering to its initial temporal distribution (Figure 2.3 (b)). In addition, the spatial dimension of the spawn data shows a pattern of concentration into areas along the east coast of Vancouver Island (Figure 2.6). The location-level errors in the database that were described earlier confuse the interpretation of spawn frequency by region - for example, the increased number of locations with spawn that are evident in the Comox region may simply be an artifact of multiple names given to one location. However, there are patterns in both spawn timing and spawn distribution that require further evaluation. The apparent loss of early spawning herring and cessation of spawning in the eastern and southern SOG has sparked debate on the reasons for the change, with a focus on the potential for fishery-induced depletions of localized herring stocks. Some observers have dismissed this possibility based on a relatively coarse comparison of fishing and spawning locations (Hay et al. 2009), whereas others (mainly First Nations and local residents) believe that intense, localized fishing has decimated many small herring populations.

The key question within the context of this paper is not what caused the concentration of herring spawn in the SOG, but what are the implications of the phenomenon? To date, this line of inquiry has yielded research into the broader ecosystem-level impacts of herring availability as a prey resource (Therriault et al. 2009, Willson and Womble 2006). The potential for population-level impacts and an evaluation of the management vulnerabilities associated with uncertainty about herring spatial dynamics have yet to be explored.

2.3 Limitations of Biomass-Based Interpretations of Stock Status

The limitations of biomass-based interpretations of stock status are not restricted to Pacific herring management, thus I draw on the experience and understanding developed for fisheries elsewhere. For many species, the scale of management frequently does not agree with the scale of population structuring because of the practical compromises that have been made in defining management units and the uncertainties related to stock identification and patterns of migration within and between populations (Smedbol and Stephenson 2001, Waples et al. 2008). These uncertainties challenge interpretations of stock status based on total biomass because they violate the key assumption underlying traditional stock assessment and management, namely that there is a closed population structure supported by simple migration circuits to and from natal habitats (Secor 2002).

2.3.1 Stock identification

Stock identification research features prominently in the history and development of fisheries science. Motivated by technological advances and management failures, research in this area has driven the development of new perspectives on population dynamics and complexity that have sometimes placed the 'stock concept' either at odds or in step with the assumptions implicit in stock assessment models (Cadrin 2005). While the trajectory of conceptual development has been non-linear and often conflicting, the majority of research on stock identification demonstrates a clear preference for stock definitions that are based on genetic evidence of differentiation (Cadrin and Secor 2009). It is therefore somewhat ironic to note that while this preference has fueled the expansion of the field of fishery genetics in recent years, genetic information is seldom formally integrated into fisheries management (Waples et al. 2008). One reason for this discrepancy is that there is no universal, objective definition of what constitutes either a fish population or a fish stock (Cadrin 2005) - the choice of definition is strongly influenced by both the management objective (i.e. conservation vs. maximizing yield) and the time scale of consideration (i.e. evolutionary vs. ecological time, as well as short-term vs. long-term management goals) (Cope and Punt 2009, Waples and Gaggiotti 2006). Not surprisingly, many of the stock definitions that are currently employed are management constructs that reflect the historical objectives of maximizing short-term yield within a given spatial area (Jennings et al. 2001). Mis-matches between management scale and biological scale are common and appear to be one of the factors contributing to collapses of fisheries worldwide, renewing recognition that existing stock definitions are insufficient for conserving biodiversity (Stephenson 1999).

Fisheries management increasingly seeks to accommodate both conservation and harvesting objectives by merging the methodology and theory of endangered species management with fisheries science (Hammer et al. 2010). However, endangered species management focuses on questions related to extinction and loss of genetic diversity, which influence stock conservation on evolutionary time scales, whereas fisheries science considers stock productivity on ecological and shorter time scales (Cope and Punt 2009). Because evolutionary and ecological processes can occur across widely differing spatial scales, these broader management goals may represent different spatial scales on which stocks can be defined for management. Furthermore, it is important to recognize that the underlying assumption of stock assessment models is that stock dynamics are driven primarily by local demographic processes of birth and death and not by exchange of individuals from adjacent areas. As such, demographic, and not genetic, independence is the key assumption underlying fisheries management (Palsboll et al. 2007).

Defining management units based on what has been called demographic independence has been proposed as a holistic solution to stock identification that integrates a variety of types of information, including genetics (Palsboll et al. 2007, Waples and Gaggiotti 2006). Under this approach, the rate of mixing is the critical feature of stock definition. Existing stock units are insufficient if movement among populations occurs at rates high enough to influence critical demographic parameters such as the natural mortality rate. This method is limited by the fact that the mixing rate at which stocks switch from demographically independent to dependent is neither well studied nor well understood, but estimates of 10% have been proposed (Hastings 1993, Waples and Naish 2009). In terms of population genetics, mixing rates of 5-10% represent levels of gene flow that are sufficient to minimize the degree of genetic differentiation between populations, at these rates genetic analyses cannot distinguish between stocks that require separate management and those that can be managed as a unit (Waples et al. 2008).

A growing body of research suggests that the fundamental issue with stock identification is not whether it is approached from an ecological or evolutionary perspective, or what method is used to identify spatial patterns of variation among fish populations. Instead, the key issue lies in the stock concept itself, which envisions fish stocks as discrete entities with a single trajectory of migration between spawning, feeding, and nursery areas (Cadrin and Secor 2009). Advances in tracking technologies have enabled fisheries scientists to identify multiple behavioral groups within many fish populations that are associated with differences in migration timing and trajectory (Secor 1999). Such diversity is poorly accommodated by fisheries management because the existing institutions, data collection protocols, and assessment methods have evolved under the discrete stock assumption (Secor 2002). In order to overcome the barrier posed by that assumption, it is necessary to extend the stock concept in order to adequately explain patterns of fish distribution in space and time.

2.3.2 Migration theory

The observation that dispersal dominates at least one life history stage in marine fish populations has been instrumental in how fisheries scientists define a stock unit (Sinclair and Solemdal 1988). Early studies on schools of North Sea and Yellow Sea herring (*Clupea harengus* and *Clupea pallasi*, respectively) indicated that different migratory behaviors existed within species (Jung, 1816 and Heincke, 1898, cited in Secor 1999). Divergent patterns of migration were believed to be preserved by a common lineage, which led to increased emphasis on schools, rather than species, as the unit of study (Secor 1999). Subsequent work demonstrated that schools were commonly affected by pulsed recruitment events, and that these events were evident over broad spatial areas as good or bad year-classes (Hjort and Lea 1914). This new ability to track productivity across space and over time shifted the focus of fisheries science away from groups of fish that exhibit common migratory behavior to those with shared spawning behavior and locations (Smith 1988, Secor 1999). By increasing its focus on the common reproductive responses of fish within a 'stock' area, fisheries science fostered a management approach that overlooked diversity in favor of the aggregate or average response of a population. However, natural heterogeneity in age structure or in the spatial distribution of spawning biomass is a critical bet-hedging strategy intended to ensure individual reproductive success despite poor environmental conditions that can persist across both space and time (Berkeley et al. 2004, Secor 2007).

Recognition that the spatial extent of adult and juvenile distributions is typically broader than the spatial extent of spawning in migratory marine fish drove the development of theories to explain the concentration of adults on local spawning grounds. The 'parent stream theory' advanced the idea that fish populations are distinct entities that do not interact with other populations throughout their annual migrations (philopatry, Harden Jones 1968). In these populations, adults undertake extensive and active homing migrations to their natal spawning grounds. A corollary to philopatry is sympatry, in which individuals in a population undergo identical migrations but may overlap with other populations during this time. Both of these concepts suggest that fish populations are bounded biological units that undertake closed migration circuits, an idea that was central to the development of theories to explain both the distribution and regulation of abundance in marine populations (Secor 2002, Sinclair and Solemdal 1988). The concepts of philopatry and sympatry are codified in the 'migration triangle', which represents an ontogenetic and annual migration circuit between spawning, nursery, and adult feeding areas (Harden Jones 1968, Cushing 1975). Straying is considered to be an unsuccessful strategy under strict interpretations of this theory, because strays are lost to the population and presumed not to contribute to future production (*sensu* vagrants of the Sinclair 1988 member-vagrant hypothesis). However, as it was initially envisioned, straying was incorporated into migration theory in the form of 'biological insurance' against a changing environment (Harden Jones 1968, Secor 2002).

The dynamics of spatially complex fish taxa cannot be easily incorporated into standard models of fish migration and population structure. In recognition of this, the population concept has been extended to incorporate straying and natural, 'anomalous' migration patterns within populations - the key ideas are the contingent hypothesis (Clark 1968, Secor 1999) and the meta-population concept (Levins 1969, McQuinn 1997). Both recognize that polymorphism in behavior, physiology, and morphology exists within populations, but they differ in the importance assigned to the idea of stability when applied to divergent migration patterns and population structure. Contingent thinking is flexible, allowing for a number of mechanisms to explain the persistence of alternative migratory behaviors (i.e. contingents) within populations. It is inclusive of both reproductively isolated sub-populations and evolutionarily stable polymorphism in life cycles that is associated with other critical phenotypic traits such as body size (Secor 1999). Meta-population theory emphasizes the dual roles of straying and local extinction as integral to the persistence and expansion of the population aggregate. Under this theory, every local population has a probability of going extinct, and vacant sites are re-colonized by strays from within the meta-population. The prominence of extirpation as a natural process means that straying is not selected for at the local level, rather, the adaptive advantage of straying is conferred at the meta-population level (McQuinn 1997). However, in spite of its popularity as an explanation for spatial structure, there is little empirical evidence for the key meta-population processes of local extinction and re-colonization (Smedbol et al. 2002). As such, the primary benefit of the theory may lie in the fact that it has fostered recognition of spatial structure within populations that cannot be measured by patterns of genetic variation (Cadrin and Secor 2009).

2.4 Potential Interpretations of Trends in the Herring Spawn Data

2.4.1 Hypothesis 1 - reduced productivity and ecological diversity

My first hypothesis is that the spatial and temporal trends in the spawn data (Figures 2.3 and 2.4) represent the true spawning patterns of herring in the SOG. If this hypothesis is correct, it may signify a decline in the productive capacity of SOG herring. Spawning distributions are increasingly recognized as central features of fish stock dynamics, they are as important to future production as total biomass (Berkeley et al. 2004). However, the benefit of variable spawning distributions might be manifest in different ways by different species. For example, some pelagic species exhibit strong inter-annual variability in the distribution of spawning events (Planque et al. 2007), which appears to be influenced by both environmental factors and characteristics of the population such as total abundance or age- and size-structure (e.g., Pacific sardine, *Sardinops sagax*) (Bellier et al. 2007). In these species, population resilience may be imparted via a naturally high variability in distribution that spreads risks over many spatially discrete locations (Berkeley et al., 2004). In contrast, species such as rockfish that are characterized by low dispersal rates may benefit from spatially discrete spawning events because most spawners are unable to match their reproductive activity to ocean conditions favorable for larval survival (Cushing 1975). As such, the surviving year class may be produced by only a small percentage of spawning adults (Berkeley et al. 2004).

In addition to reduced productivity, the concentration of spawning in space and time may reflect a loss of ecological diversity within the SOG. B.C. herring exhibit differences in spawn timing, migration rates, growth rates, and life history characteristics throughout their range (e.g., Hay 1985, Hay and McCarter 1997, Schweigert

1991). A variety of data on morphology, movement, spawn timing and distribution, and genetics has been investigated in an effort to provide stable estimates of stock structure and to ensure the appropriate scale of management (for a full review see Stocker 1993 and more recent work by Beacham et al. 2002, 2008). As a result, the number of herring populations identified in B.C. has varied between 35 and most recently 7 since research began in 1937 (Stocker 1993). Tagging studies suggest that herring stray among the 5 major and 2 minor stocks, providing a degree of connectedness that may be consistent with metapopulation dynamics (Ware and Schweigert 2001). However, the pattern of straying is unclear and is highly dependent on the spatial scale over which the data are examined. Fidelity to previously used spawning sites may be low at the scale of a bay or inlet, and approach 100% within a large area such as the SOG (Hay et al. 2001).

This observation contrasts with the herring 'credo' used to explain population structure in Atlantic herring. Two elements of that credo relevant for this discussion are: (1) herring spawn in multiple discrete locations in a stock area and these are predictable in space and time, and (2) herring return to spawning grounds they have used previously and have fidelity to the spawning ground from which they originated (homing) (Stephenson et al. 2009). Another relevant observation is that tagging studies for Pacific herring show that herring tagged together tend to stay together (Hay and McKinnell 2002), implying a level of structuring similar to that of Atlantic herring.

The results of genetic and tagging analyses are used to support the current Pacific herring stock groupings, which form the basis for all stock assessment and management decisions. These studies indicate that there is little evidence for genetically distinct populations spawning in the individual bays and inlets within the 5 large stock areas (Beacham et al. 2002, (2008)). However, the broad-scale application of these results belies the remarkable diversity that exists within some stock areas, and which is present in the form of both genetic and phenotypic diversity. For example, populations that are genetically distinct from the main spawning aggregation have been identified in waters encompassed by or adjoining the Strait of Georgia - mainland inlets in Johnstone Strait, Portage Inlet on southern Vancouver Island, and Cherry Point in Washington State - as well as the populations found further south in Puget Sound (Beacham et al. 2008, Small et al. 2005). In addition to the genetic variation, SOG herring exhibit two distinct life-history types characterized by resident and migratory strategies. While not well understood, these types are believed to differ in their timing of spawning and use of both spawning and feeding habitats. Resident herring spawn both earlier and later than the major migratory schools, and at the heads of inlets, whereas migrants spawn in more accessible beaches in the coastal areas and move offshore to feed in the summer months (Taylor 1964, Beacham et al. 2008). Recent evidence suggests that that there may be geographic association of resident fish on a broad spatial scale. Beacham et al. (2008) found that resident herring in the eastern SOG originated from mainland inlets in that region, but that resident fish in the western SOG were a component of the large, migratory population that failed to undertake the summer feeding migration.

The concentration in timing and distribution of spawning reflects a homogenization of spawning behavior, and by extension, a loss of the suite of spawning behaviors specific to resident herring in the SOG. Two observations support this statement: the loss of early and late spawning fish, and cessation of spawning in the eastern SOG, both of which are behaviors that are associated with resident herring (Beacham et al. 2008). The coexistence of resident and migratory fish within the same population is referred to as 'partial migration', and has been documented for many fish species (Kerr and Secor 2009). Different migratory behaviors (referred to as contingents) can contribute to the development of genetically distinct populations, but these behaviors can also arise from multiple discrete phenotypes that develop within populations as a result of different environmental conditions experienced by individuals early in life via differences in spawn timing, initial density, or social factors (Jonsson and

Jonsson 1993, Secor 1999). In SOG herring, the migratory contingents are larger bodied and more abundant than the residents, a feature that may be explained by partial migration. Recent work on white perch in Chesapeake Bay found that this phenomenon arises from varying energetic tactics, such that perch with higher energy requirements are forced to migrate to a new habitat that can meet their needs (Kerr and Secor 2009). This life cycle diversity is believed to benefit the aggregate population by dramatically increasing the scope for individual growth and increasing population productivity (Kerr and Secor 2009). In addition to the growth benefits, different contingents may confer different types of benefits to the aggregate dynamics of populations. For example, the proportion of the population that recruits to either the resident or migratory perch contingents in Chesapeake Bay is determined by the strength of the spring freshet such that the proportion represented by the smaller bodied, lower-productivity resident contingent increases in low-flow and drought years (Kraus and Secor 2004). This suggests that the lower energetic needs of the resident contingent, which can be met by the local habitat, reflect an important adaptation that enables the white perch population to persist in poor environmental conditions (Kerr et al. 2010). Conversely, the ecological value of the highly productive migratory contingent lies in its contribution to population productivity which speeds recovery from a depleted state (Kerr et al. 2010). Given the differences in body size of resident and migratory herring in SOG, it is possible that the two contingents of herring provide different, but complementary behaviors that sustain the aggregate productivity of the stock.

2.4.2 Hypothesis 2 - spatial concentration of the spawn monitoring program

My second hypothesis to explain the concentrating pattern in SOG herring spawn timing and distribution (Figures 2.3 and 2.4) is that the spatial and temporal trends in the data represent an ongoing concentration of the spawn survey in space and time. This is a significant problem that has not been formally evaluated for Pacific herring. The herring survey is assumed to generate an absolute measure of spawner abundance, and to be a reliable source of fishery-independent information for the stock assessment model (Cleary et al. 2009). However, survey estimates of absolute abundance rely on the fulfillment of two criteria: (i) a pre-specified spatial coverage of the survey within a closed boundary, and (ii) systematic placement of transects within the survey area to ensure the resulting biomass estimates are spatially representative (McGarvey et al. 2008). There is ample evidence that the herring survey violates both of these criteria.

Considerable research has been directed at refining estimates of egg deposition within a spawning location in order to obtain accurate assessments of stock size at a small spatial scale (e.g., Schweigert and Fournier 1982, Schweigert and Stocker 1988, Schweigert 1993). An evaluation of the dive survey protocol found that the variation in egg density between individual spawning locations is so large that all spawning locations must be surveyed in order to achieve stable estimates of spawning biomass (i.e., it is not acceptable to survey a fraction of the spawning locations and apply a mean density estimate to the unsurveyed locations) (Schweigert et al. 1990). The natural spatial variability of herring means that the number of individual spawning locations in each stock region are not known a *priori,* and as such, the spatial coverage and representativeness of the survey cannot be verified. The 'on the ground' approach, where the greatest effort is concentrated on the largest areas of spawn, and the aim is to survey at least 30% of all spawning locations, is ad hoc (Schweigert et al. 1990). Local estimates are aggregated over the *surveyed* locations (not all of the potential spawning locations) to produce the total spawning biomass for each herring stock (Schweigert 1993). The herring survey therefore provides estimates of spawning biomass for an unknown fraction of the population, which is assumed to be consistent over time. In practice the sampled fraction is assumed to be 100% for all years of the dive survey, 1987 onward (i.e., survey catchability is assumed to be 1, Cleary et al.

2009). This is a requirement for obtaining reasonable statistical performance of the stock assessment model (Fu et al. 2004).

The interaction of the distribution of the survey and the spatial dynamics of herring has important implications for the quality of the estimates of spawning biomass. In particular, the survey is vulnerable to generating biomass estimates that are relatively constant even if actual biomass changes considerably (hyperstability) if it represents only areas of major fish concentration and the occupied range contracts as the stock biomass declines (Hilborn and Walters 1992, Hutchings 1996). Hyperstability is a well known phenomenon in which indices of abundance remain artificially high as population size decreases and the fish concentrate in the best habitats. A key recommendation to avoid this possibility is that surveys should encapsulate the entire geographic range of the population in order to ensure adequate spatial coverage during periods of both high and low abundance (Hilborn and Walters 1992). The acknowledged targeting of the largest areas of spawn deposition by the dive survey since its start in 1987 (Schweigert et al. 1990) introduces substantial uncertainty into the dive survey estimates of stock biomass, and raises a question regarding the validity of the upward trend in spawning biomass that occurred concurrently with a concentration of survey effort into the most productive spawning areas (Figures 2.3 (a) and 2.6).

2.4.3 Hypothesis 3 - a combination of processes

The third hypothesis is that the trends in the spawn data can be explained by some combination of hypotheses 1 and 2. Published summaries of the data (e.g., Hay et al. 2009, Therriault et al. 2009, Willson and Womble 2006) suggest that the spatial contraction in the spawning distribution of herring in the SOG is a real ecological phenomenon, but they are all limited by the problems with the survey data outlined in section 2.2.2. However, support for hypothesis 1 may be found in an alternative source of information: the Sliammon First Nation on the Sunshine Coast has documented a failure of herring to return to spawning grounds off Powell River since the mid-1980s. Traditional ecological knowledge (TEK), supported by archaeological research, suggests that these spawning beds supported large herring fisheries for at least 1000 years before that time (D. Lepofsky, *pers. comm.*). A similar pattern has been reported by residents of Gulf Island communities and the fishermen themselves (Penn 2009).

Support for the second hypothesis (a spatial concentration of survey effort) can be inferred from the objectives, time, and cost considerations of the dive survey, which make it reasonable to expect that the survey focused on areas reported to have large spawns (Schweigert et al. 1990). Building on this inference, it is also fair to assume that the survey would extend beyond the spatial coverage used from approximately 1987 to the present if substantial spawns were repeatedly occurring in 'abandoned' sites year after year. Collectively, all of these points above suggest that there may have been coinciding changes in spatial concentration in both the SOG spawning populations and the survey.

The issues presented in section 2.4.2 centered on how spatial contraction in the monitoring program can affect estimates of biomass. These problems are compounded by uncertainty about herring spatial dynamics, which is not acknowledged in the current survey design. Exploited fish stocks commonly originate from several spawning components, and the relative contribution of each harvested stock depends on its productivity and availability to the fishery (Begg et al. 1999). Such differences in productivity can arise from either distinct population units or different patterns of habitat use by components of the same population (Hilborn 1985). Ricker (1958) was among the first to recognize the challenge of monitoring and managing such populations, depicting how the less productive components are more vulnerable to depletion when they are exploited by a mixed-stock fishery. As a result, information on the relative contribution of the least productive population components to total production is considered critical for avoiding over-harvesting in such fisheries (Jonsdottir et al. 2007). Because the SOG roe herring fishery operates on spatially disaggregated schools of spawning herring, it may be immune to this problem. However, regardless of the pattern of fishing, the mixed- (or spatially structured) stock issue remains an issue because ignoring the diversity of productivities within a population can lead to underestimates of the total population size and inflated estimates of productivity (Hilborn 1985).

2.5 Discussion

2.5.1 Management implications

Uncertainties dominate resource management. Most fisheries scientists recognize this, and have begun to design management systems that are robust to a variety of types of uncertainty (de la Mare 1998). Therefore, the existence of uncertainty is not as large a problem as the failure to recognize it and take it into account explicitly in a management system. Failing to recognize uncertainty and to challenge the assumptions underpinning scientific analyses masks the fact that biological processes are seldom well understood and can often be described by a variety of equally valid models (Ludwig et al. 1993). This promotes false confidence on the part of managers and policy makers who look to scientists for advice (Harwood and Stokes 2003).

The spatial ecology of SOG herring is not well understood and the significance of this uncertainty has not been recognized by science nor adequately communicated to managers. As such, all management efforts to date have focused on total biomass within the putative stock area. In this chapter, I elaborated on the importance of within-population diversity for maintaining resilient fish populations, and outlined how failing to acknowledge, let alone actively manage, such diversity may affect the quality of advice provided to managers. This body of work indicates that a singular focus on biomass-based management is inappropriate for Pacific herring, and that overconfidence in the relatively simple stock structure assumption has yielded a significant but unrecognized vulnerability in the herring management system. The evidence presented thus far supports the hypothesis that the concentration of spawning herring into areas off the east coast of Vancouver Island was matched by a concentration of survey effort into locations with the largest spawns. Regardless of the underlying stock structure, any loss of spatial diversity in spawning may reflect a loss of resilience in the SOG herring population (Schindler et al. 2010, Stephenson 1999, Stephenson et al. 2009).

The SOG herring management system focuses on total biomass and does not formally address the spatial dynamics of the population. This has two key implications for management: first, the spawning biomass index used in the current management protocol may be subject to considerable bias, which has the potential to negatively affect the performance of both the stock assessment and the harvest control rule. The relationship between the spawn index and the SOG population cannot be verified, but it is assumed to be constant over time and to be independent of biomass. In theory however, this relationship is constantly changing, particularly for populations of small schooling fish which expand and contract their distributions rapidly within a survey or fishing season (Hilborn and Walters 1992). Spatial concentration of both the survey and the stock can therefore lead to an inverse relationship between biomass and the survey catchability q, which can generate overestimates of abundance as the stock concentrates in space, a feature that is well known for clupeid fish populations (for a review see Arreguin-Sanchez 1996). In addition, spatial bias in the survey may generate errors in the stock assessment model that propagate through to calculations of allowable catch that are based on the estimated adult stock biomass. It is difficult to draw specific conclusions about how fixing the survey catchability coefficient to a known overestimate of *1* affects the stock assessment, but confounding between q and the estimated natural mortality rate M has been documented for previous versions of the model (Fu et al. 2004), and is common to many stock assessments (Hilborn and

Walters 1992). Fixing q can therefore lead to erroneous estimates of M, which subsequently affect the estimated fishing mortality rate and generate biased abundance estimates (Clark 1999). Another issue is that the herring survey provides estimates of abundance-given-presence, i.e., it does not account for absence (true zero abundance). The validity of using this type of data to estimate absolute abundance, has not been demonstrated (Pearce and Boyce, 2006). If such an approach was developed for estimating abundance, it would require some method of dealing with pseudo-absence data in order to deal with the biases inherent in using presence-only data to draw conclusions about the distribution of abundance (or lack thereof) in non-sampled sites (Pearce and Boyce, 2006).

The second management implication is that mis-match between the spatial scale of monitoring/control and the dynamics of the population may lead to overexploitation or loss of some population components. In effect, SOG herring are treated as having simple spatial dynamics for the purposes of stock assessment and management (Cleary et al. 2010), but this is known to be false; both spawning and harvesting occur at fine spatial scales. What is unknown is how these fine-scale processes interact and affect each other. For example, harvest rates and allowable catch are calculated for the population aggregate and there are no spatial controls on exploitation (Cleary et al. 2010). It is therefore probable that the harvest rates realized by some spawning aggregations exceed the target harvest rate of 20%. The impact of this pattern of exploitation is unknown, but may result in overexploited or depleted components of the SOG population (Frank and Brickman 2000). A loss of within-population diversity may mean that the scientific models used to predict the consequences of alternative management actions do not reflect the true dynamics of the population or the fishery. As a result, the response of the SOG population aggregate to harvesting and climate impacts may be entirely unexpected (Stephenson 1999, Walters 1987).

2.5.2 Recommendations

2.5.2.1 Monitoring

The first recommendation is to improve the precision of the location-level observations in the survey by including better metadata for each record. This would provide the managers of the herring database with the information required to verify and control data quality at a fine spatial resolution. Beyond this, the required change is straightforward: modify the monitoring protocol to obtain estimates of *both* distribution and abundance. The implementation of this requirement is less clear. One feasible approach might be to conduct a 2-stage survey where the broad spawning distribution is obtained from aerial survey overflights (providing presence and absence information), and then based on the aerial estimates of the relative size of the spawns from this initial phase, DFO could dispatch dive survey crews to estimate the biomass in locations that meet some pre-established statistical criteria. Examples of aerial surveys designed to estimate distribution and abundance of Pacific herring exist in Alaska (Thomas and Thorne 2003). In addition, the apparent trend toward increasing concentration of the spawn distribution could be formally tested by conducting information interviews of longtime resource users in the SOG. Brown et al. 2002 designed a TEK interview protocol and geographical database to collect information on Pacific herring abundance, life-history, and distribution from First Nations and commercial fishermen in Alaska. This information has subsequently been used to design surveys, validate stock assessment model outputs, and to inform the development of theories on herring population structure in the region. A similar program of engaging the resource users in the SOG would provide valuable scientific information and could provide a process through which the users could meaningfully contribute to research and management.

2.5.2.2 Evaluating the impacts of spatial structure in stock assessment and management

As previously discussed, the key uncertainties related to the spawning distribution of SOG herring are the unknown effects of spatially and temporally biased sampling on the index of spawning biomass, and the unknown impact of ignoring the spatial distribution of both the fish and the fishery. These uncertainties have the potential to affect the perceptions of stock status as well as advice provided to herring managers because they can impact both the quality of the stock assessment and the predicted consequences of management actions (Punt and Donovan 2007). Simulation modeling is useful for evaluating the likelihood of success of fishery management strategies when observations, natural process dynamics, and implementation of management regulations are all uncertain (Butterworth and Punt 1999). Simulation studies have additionally demonstrated that ignoring spatial uncertainty in fisheries management can lead to a loss of sub-populations and erosion of spatial complexity within stock complexes (de la Mare 1996, Frank and Brickman 2000). The results of these studies imply that SOG herring conservation goals will not be achieved if the scale of management does not match the biological scale of the population. This should be tested using closed-loop simulation of the management system (i.e. a management procedure evaluation, MPE), which incorporates alternative hypotheses about the dynamics of the population, the quality of the monitoring program, and the fishery dynamics (de la Mare 1998). The MPE approach can be used to identify the key uncertainties in SOG herring management, and to inform the selection of a combination of monitoring protocol, stock assessment model, and harvest control rule (collectively referred to as the management procedure) that is robust to these uncertainties. Such an evaluation was conducted by the International Whaling Commission, which found that spatial and stock structure uncertainties significantly limited the performance of management procedures that were otherwise considered perform well with respect to the fishery objectives (Punt and Donovan 2007). I conduct such an evaluation in the next chapter.

2.6 Conclusion

Spatially dynamic populations present a problem for traditional fisheries management, which focuses on maintaining biomass within a set of pre-determined limits. However, recent ecological, political, and economic imperatives have prompted fisheries scientists to recognize the value of biodiversity and to begin to change the paradigm under which fisheries research and management are conducted (Benson, 2009). Under the biodiversity-based paradigm, fish populations are viewed as containing diverse responses to external forcing, which collectively enable the population to persist over a broad range of environmental states, i.e., to demonstrate resilience (Elmqvist et al. 2003, Hilborn et al. 2003).

The case of SOG herring suggests a disconnect between a broad-scale recognition of the value of maintaining resilient fish populations and the ability to change the management system in order to ensure progress toward the broader set of spatial objectives implied by this goal. In particular, inclusion of spatial criteria in management requires some effort to map where the spawning fish are and are not each year. In a fishery facing constraints of time and money, this change may require a shift of sampling effort away from sites known to contain fish, which would presumably result in a loss of survey precision due to reduced sampling effort in locations that contain spawn over time. The sampling trade-off between biomass and distribution as indicators of population status is a difficult one, and in the absence of evidence of poor management performance, it is tempting to continue to focus on biomass. However, the spatial trends in the SOG spawn survey data show a marked concentration of spawning activity and survey effort into areas off the east coast of Vancouver Island. It is necessary to look beyond biomass in order to begin to evaluate the management implications of this trend.

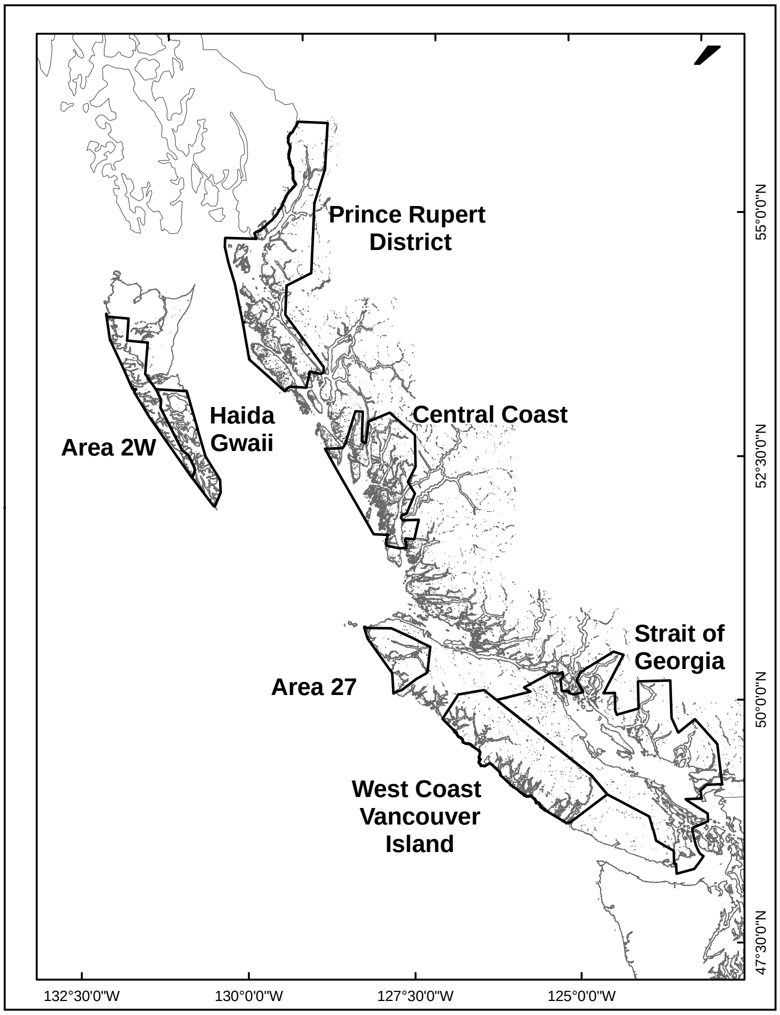

Figure 2.1: Map of the British Columbia herring management regions, courtesy of Jaclyn Cleary, Fisheries and Oceans Canada.

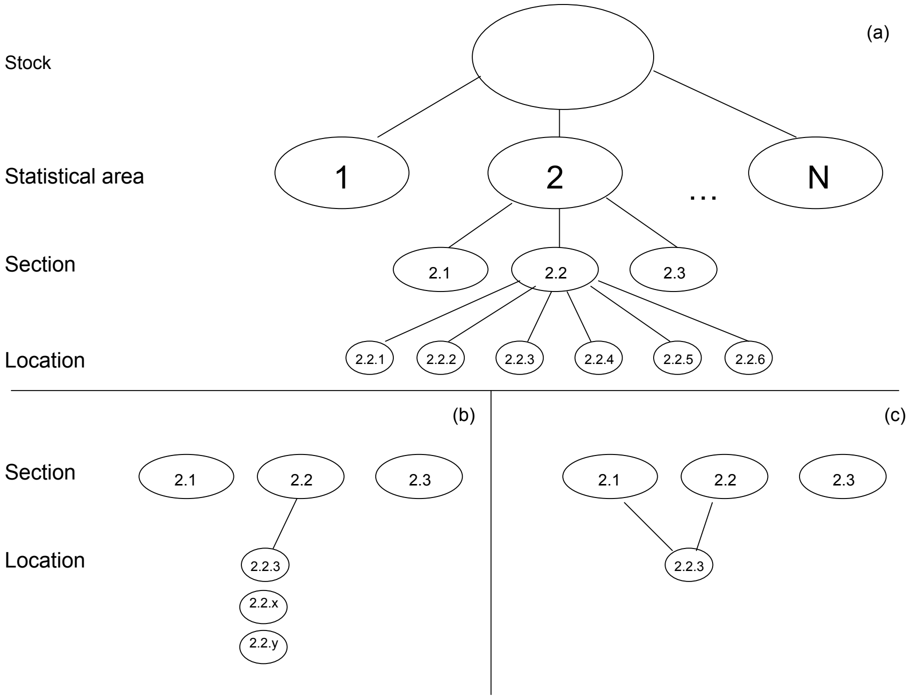

Figure 2.2: The hierarchical structure of spawn monitoring data in the DFO herring spawn database. The data structure reflects a nesting of spawning locations (individual beaches and bays) within geographically larger sections, statistical areas, and ultimately, stock assessment area. The lower two plots depict potential, undocumented errors at the scale of spawning 'location' - multiple names for one location (b), and locations that are associated with multiple sections (c).

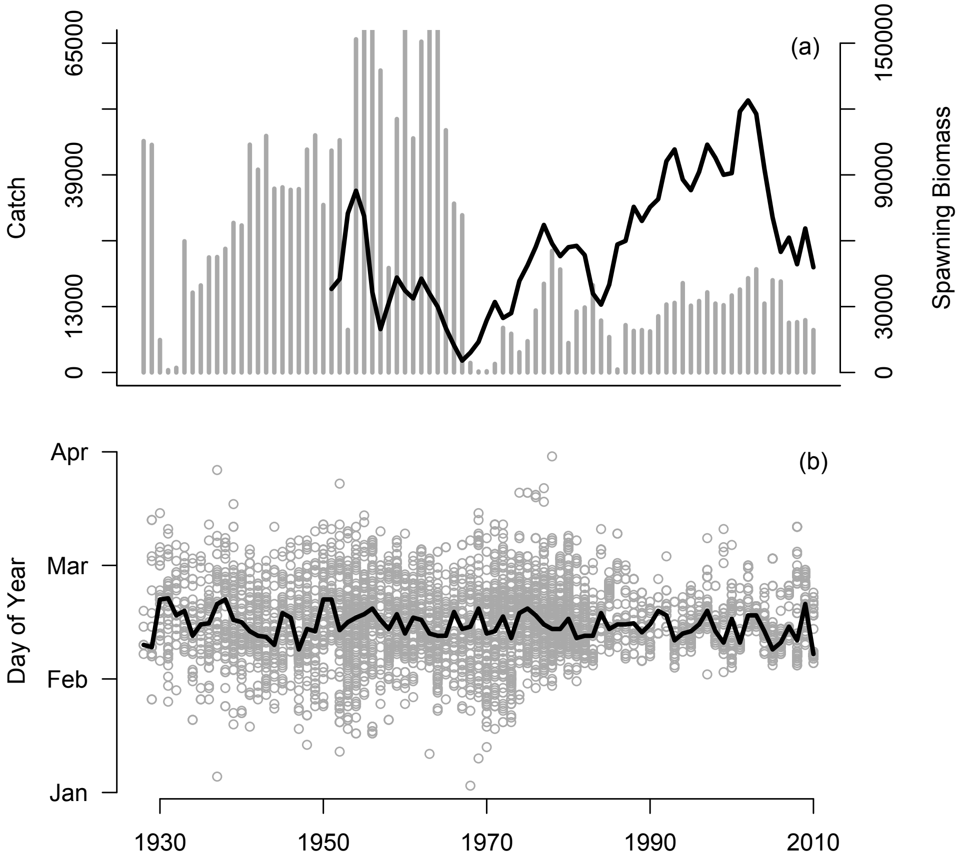

Figure 2.3: Time series of catch (1928-2010) and estimated spawning biomass (1951-2010), depicting changes in absolute removals and stock abundance in the Strait of Georgia (a). Plot (b) illustrates the changes in annual spawn timing (1928-2010). The grey dots show the distribution of spawn dates of herring in different sections within the Strait, and the solid line is the median annual spawn date, which shows no trend over time. Data were obtained from the Herring Stock Assessment Database, Fisheries and Oceans Canada, Pacific Region.

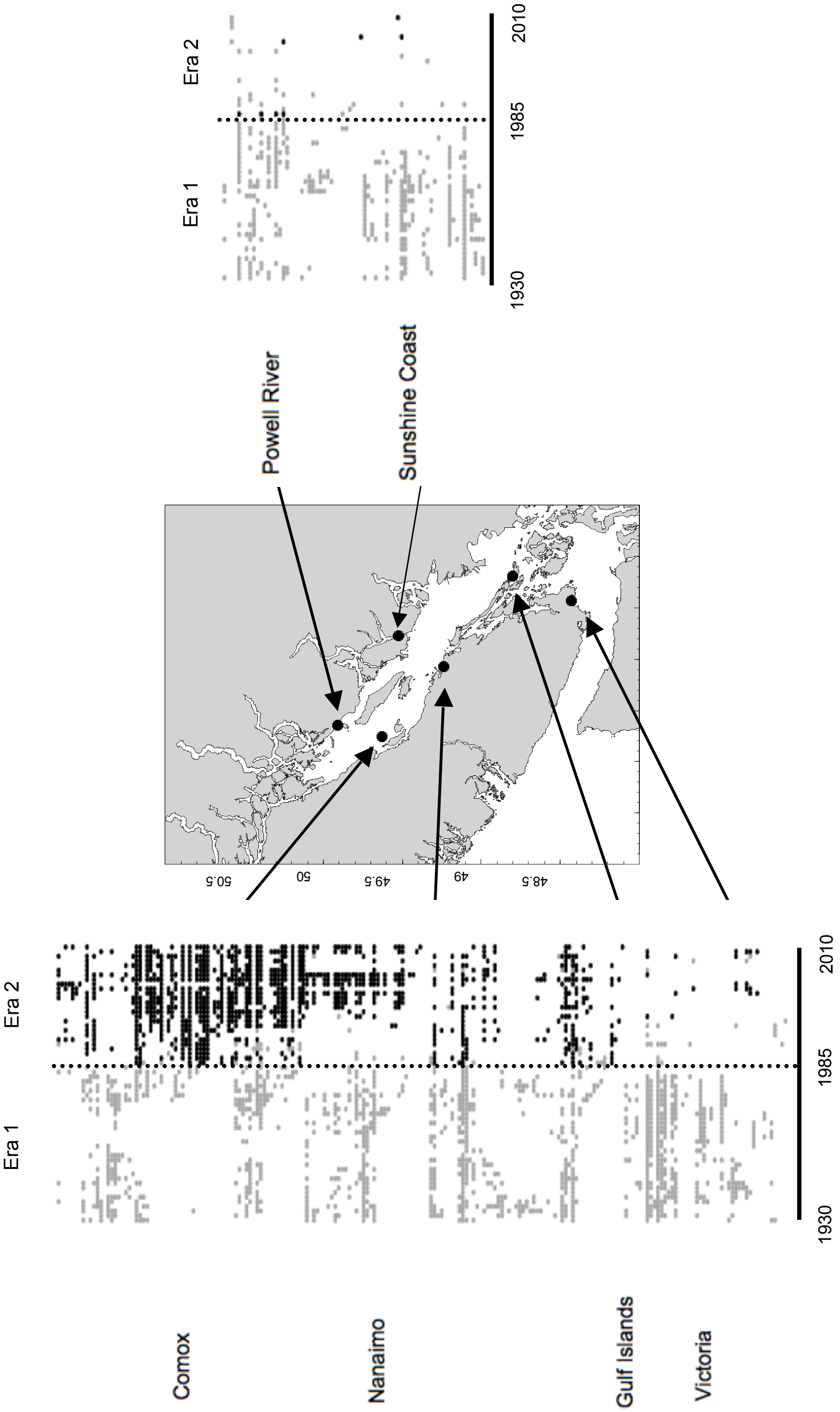
Powell River
Sunshine Coast
Era 1
Era 2
1930
1985
2010
50.5
50
49.5
49
48.5
Comox
Nanaimo
Gulf Islands
Victoria

Figure 2.4: Map of herring spawn survey locations in the Strait of Georgia, 1930-2010. Each row on the plot represents a 'location', and columns represent individual years. Grey and black dots signify observations of spawn recorded during the surface and dive-survey eras, respectively (Eras 1 and 2). The Era 1 survey may be more representative of the spatial distribution of spawn than the Era 2 survey, which is restricted to areas with moderate-to-large deposits of spawn. Data for this latter era are therefore presence-only, meaning that the white space might represent either the absence of spawn or 'unknown abundance' because it is not covered by the survey. The white space in Era 2 can therefore be interpreted as a map of the uncertainty associated with the spatial distribution of herring at the time of spawning. This uncertainty appears to be increasing dramatically over time in management areas on the mainland (Powell River and Sunshine Coast) and in the southern Strait of Georgia (Victoria and Gulf Islands). Data were obtained from the Herring Stock Assessment Database, Fisheries and Oceans Canada, Pacific Region.

Chapter 3

Identifying conservation leverage points in a fishery system facing practical constraints on the scale of management

3.1 Introduction

Fisheries stock assessment and management are predicated on the assumption of an idealized 'unit stock' - a coherent group of fish governed primarily by its own demographic processes, and for which the impact of fishing can be isolated in time or space (Cadrin 2005). However, there is growing recognition that the stock concept is insufficient for describing the spatial diversity in genetic, behavioural, and phenotypic characteristics (referred to as spatial complexity hereafter) that has been found within individual species and populations (Hilborn et al. 2003, Schindler et al. 2010, Kerr et al. 2010). Unfortunately that spatial complexity is routinely ignored in stock assessment and management (Cope and Punt 2011, Punt and Donovan 2007, Rothschild 2007). The rationale for assuming homogeneity of spatially diverse populations is justified by uncertainty about the ecological processes that govern spatial structure within, and connectivity between, local and regional populations (Corten 2002, McQuinn 1997, Sinclair 1988). Scientists' understanding of these ecological processes is confounded by the fact that spatially-explicit stock assessment models do not consistently perform better than aggregate-area assessments because the parameter estimation procedure is affected by data availability and quality (Butterworth and Geromont 2000, Cope and Punt 2011). The high cost of quantifying potential spatial complexity, combined with limited statistical power of many methods that are currently used to identify fine-scale spatial structure, are additional factors that reinforce the incorrect application of assumptions of homogeneity in stock assessment and management (O'Connell and Wright 1997, Waples and Gaggiotti 2006).

In order to overcome these limitations and move to a finer spatial scale of management, scientists feel the need to demonstrate 'appreciable levels' of spatial structure, but the definition of an 'appreciable level' of structure for fisheries management is frequently unclear and rarely articulated (Cope and Punt 2011, Waples and Gaggiotti 2006). Thus, fine-scale assessment and management appears unlikely to occur in the near future for many spatially structured populations. Faced with this situation, it is helpful to identify the conditions under which aggregate management of what is actually a spatially complex population can successfully balance conservation and exploitation objectives.

I use a management procedure evaluation (MPE) approach to evaluate the potential consequences of ignoring spatial complexity in fisheries stock assessment and management. MPE uses closed-loop simulation to evaluate the 'tools' of fisheries management (the combination of data collection, stock assessment modelling, and rules for determining allowable catch) against their ability to achieve conservation and economic objectives for specific fisheries (Cox and Kronlund 2008, de la Mare 1998, Punt 2006). The most widely recognized application of MPE was the development of the revised management procedure of the International Whaling Commission (Punt and Donovan 2007). I use MPE to evaluate management in the following, broader sense: (1) to test the hypothesis that a management procedure that ignores spatial complexity will lead to management failures, and (2) to identify general management formulations that work to achieve conservation and economic objectives in spite of a mis-match between population spatial complexity and management.

Building on these outcomes, I seek to identify the points in the system at which intervention can achieve a large impact on management performance; these 'leverage points' (Meadows 1999) may ultimately be used to identify and prioritize research and management needs for spatially complex populations. In this chapter I only examine potential leverage points as parts of the system that are under management control. Common examples of leverage points in fisheries science and management include the level of observation error in the survey, and the structural assumptions of the stock assessment model, either of which could have a large impact on the ability to achieve the management objectives.

The fishery system simulated in this chapter is based on general characteristics of the British Columbia (B.C.), Canada, Pacific herring (*Clupea pallasi*) fishery (described below). The indicators that I use to measure management performance reflect the common goals of fisheries management, which are to maintain stock biomass at B_{MSY} or higher (i.e., not overfished), achieve a fishing mortality rate of F_{MSY} or lower (i.e., overfishing is not occurring), generate reasonable economic returns from the fishery, and ensure that the spatial structure of the population aggregate is not compromised.

MPE separates the fisheries management function into three stages (Figure 3.1): (1) information generation (data collection and stock assessment, the first two elements of the management procedure), (2) provision of advice from stock assessment scientists to managers (harvest control rule, the final element of the management procedure (MP)), and (3) application of management advice. Ignoring spatial complexity at any stage has the potential to affect the performance of the entire management system (Stephenson 1999). The purpose of stage-1 is to generate a perception, or estimate, of the true state of the resource based on the data collection protocol and stock assessment model. Errors in perception of population status arise from a variety of sources, including, among other things, low-power survey designs for data collection, inappropriate assessment models, and mis-specification of the spatial complexity in those two steps. Stage-2 represents the decision-making process that links the estimated stock status with policy to generate harvest advice. Stage-1 errors can propagate through this stage and affect the quality of advice provided to managers in the form of recommended catch limits or acceptable harvest rates (Cope and Punt 2011, Punt and Donovan 2007). Fishing dominates stage-3, because it is the process by which fishing regulations are implemented. Deviation between the management target (i.e., the recommended catch) and the actual outcome is called 'implementation uncertainty', and is increasingly recognized in evaluations of fisheries management systems (Holt and Peterman 2006). Implementation uncertainty exists in fishery systems because managers are unable to predict precisely how fishermen will alter fishing patterns in response to changes in fishing regulations (Holt and Peterman 2006). Critical errors in implementation can also occur when management recommendations ignore spatial complexity. For example, previous simulation work has shown that failing to place spatial controls on fishing leads to overfishing of sub-populations and erosion of diversity within stock complexes (de la Mare 1996, Frank and Brickman 2000).

3.1.1 Synopsis of the B.C. Pacific herring fishery system

Pacific herring exhibit many characteristics and behaviours that complicate clear delineation of a “stock”:

1. Spawning occurs in large bays and inlets throughout their distribution from B.C. to Korea. However, within this region, the majority of herring undertake

seasonal migrations from these inshore spawning and nursery areas to offshore feeding grounds along the continental shelf, while a smaller portion remains close to spawning areas year round (Stevenson 1946),

2. Spawning sites are spatially discrete, with an unknown rate and spatial pattern of straying among sites (Hay 1985),
3. Herring exhibit phenotypic diversity in both morphology and spawn timing (Haegele and Schweigert 1985, Schweigert 1991),
4. Herring abandon and recolonize spawning locations (Ware and Tovey 2004)

Uncertainty generated by spatial diversity in herring spawning populations has prompted long-term monitoring of herring spawning behaviour, location, and timing (for a review see Hay et al. 2009). However, despite application of a variety of stock identification techniques (tagging, statistical analyses of spawn timing, location, and phenotype), no clear signal of population structure has emerged (Hay et al. 2009). The most recent stock identification methods focus on identifying genetic differentiation among herring populations, presumably with the expectation that a statistically robust and repeatable measure of population structure will be more useful for management (Beacham et al. 2008). These studies indicate that there are pockets of genetically and behaviourally distinct herring within the putative managed 'stocks' (Beacham et al. 2008). However, identification of genetic diversity has not prompted a shift to fine-scale management; instead B.C. herring are managed as 5 large 'stocks' within which discrete populations are known to exist, but are ignored in management (Beacham et al. 2008, Cleary et al. 2009). The 5 stocks are assessed separately, with a specific harvest control rule that is used provide advice on stock-specific TACs (Cleary et al. 2009). No spatial controls on fishing are applied within the boundaries of the 5 stock areas. Instead, the fishery is timed to optimize the quality of herring roe as fish move into and out of spawning locations during the fishing season.

3.2 Methods

3.2.1 General features of the simulation model

My simulation model has two main components: (i) an operating model that stochastically simulates the "true" spatial and temporal responses of multiple fish populations and fisheries to management regulations (Figure 3.1), and (ii) a management procedure (MP) that determines annual total allowable catch limits (TAC) based on survey data, stock assessment analyses, and a harvest control rule that are applied to the "true" population states (parts 1 and 2, Figure 3.1). The first key feature of this simulation is that the MP assumes a single, homogeneously-mixed fish population or aggregate stock, which implies that multiple spawning populations are irrelevant to the overall stock production dynamics. The assumption is analogous to the assumptions in each one of the 5 B.C. herring stock areas, and is manifest in each component of the MP. For instance, for each of these 5 areas, the survey does not account for differences in herring density between spawning locations, the stock assessment analysis uses a single, closed population model to estimate the management parameter (e.g., B_0) and the harvest control rule sets a single annual TAC that can be taken from any combination of spawning locations. As discussed earlier, violation of the aggregate stock assumption may lead to inappropriate calculations within any component of the MP.

The operating model (Figure 3.1) represents possible underlying dynamics of the fish population and fishery. I test the aggregate-stock MP against a set of operating model scenarios that encompass a range of assumptions about the mechanisms governing spatial complexity and the capacity of local spawning populations to recover from a depleted state. Spatial complexity is included in the operating model via local variation in spawning habitat quality, spawn timing, and in alternate hypotheses about within-population connectivity. The connectivity scenarios include: (1) discrete sub-populations that are closed to immigration and emigration, and (2) mixed populations - multiple populations that are mixed by a high rate of density-dependent habitat selection by adult fish. The discrete sub-populations scenarios are an extreme violation of the management procedure's assumption because over-fishing of any stock has a direct negative impact on total population production. The mixed-populations scenarios most closely match the management assumption and are therefore expected to achieve good management outcomes. The consequences of mis-match between the assumption of a single stock in the MP and the "true" multiple-populations state represented in the operating model may depend on the interaction between the spatial complexity scenario and how TACs derived from the procedure are taken from the populations. For example, in the absence of spatial controls on fishing effort, the *a priori* expectation is for the fishery to take the TAC by fishing in the highest-abundance spawning areas first, and move on to fish lower-abundance areas as the season progresses. Such a fishing pattern may result in unsustainable fishing mortality rates on some components of the population because the fleet operates without constraints on the catch derived from individual sub-populations.

3.2.2 Representing spatial complexity

The operating model uses a discrete delay-difference approach to provide a simple description of the exploited population that is based on biologically realistic parameters, but which accounts for biomass growth and loss and the lag effects in age-structured populations (Deriso 1980, Schnute 1985). The model generates biomass and numbers for P sub-populations. All recruited fish are fully vulnerable to the fishery and are assumed to contribute equally to the spawning stock. Model notation and parameter values are provided in Table 3.1. The equations governing the population dynamics are provided in Table 3.2. The parameters of the weight-at-age function were estimated based on 2005 herring fishery data from the Strait of Georgia. I assume that all herring recruit to the spawning population at age-3, and that the sequence of the natural population processes, harvesting, and the survey mimics the B.C. herring fishery, which targets schools of fish on or near the spawning grounds. Biomass and numbers are separated into pre-fishery ($n_{p,t}$ and $y_{p,t}$) and post-fishery ($N_{p,t}$ and $B_{p,t}$) numbers and biomass, respectively, in order to accommodate this timing. Numbers of fish are more typically represented by stock assessment scientists as the stock biomass divided by the mean weight of fish in the population in delay-difference models, however, I calculated numbers separately from biomass because the addition of movement greatly complicates the calculation of mean weight, which is a function of both natural survival and fishing mortality. The latter depends on the population of origin of the dispersing fish, which is an unnecessary complication to include in this model.

I follow previous authors (Walters et al. 2007) and assume that habitat quality and dispersal govern the local production dynamics of herring. I use habitat area A_p to represent local variations in the quality of spawning/nursery habitat. The total area $A\cdot$ is distributed among sub-populations in proportion to habitat H_p. Spatial differences in population productivity are reflected in the sub-population recruitment compensation coefficients K_p, which are estimated by solving the set of equilibrium equations to achieve either $F_{opt} = 0.6M$ or $= M$ for each sub-population. These represent low and high recruitment compensation scenarios, respectively (Walters 1998). Recruitment compensation refers to the relative improvement in juvenile survival rate as spawning stock size approaches zero, with higher compensation implying greater resilience to fishing or other disturbance because of the ability of the population to respond positively to a reduction in adult biomass (Goodyear 1977, Forrest et al. 2008). Annual recruitments of age-3 fish to each sub-population are generated using Beverton-Holt recruitment functions of local spawning biomass, recruitment compensation K_p and the local unfished biomass $B_{p,0}$ (Equations3.9, 3.10, and 3.12 in Table 3.2). The sub-population equilibrium conditions (Equations 3.5 - 3.10) are approximated in the presence of dispersal among sub-populations. In scenarios where there is exchange among sub-populations, equilibrium states are obtained by simulating the dynamic model (Equations 3.11 - 3.16) to convergence after initializing with the equilibrium approximations.

High values of the proportion of total biomass that disperses in a single year (E), represent a highly inter-connected set of spawning populations, whereas low values reflect more independent populations. The discrete-populations scenarios are therefore achieved by setting $E = 0$, which means that there is no straying among populations. The mixed populations scenarios use a proportional emigration rate of $E = 0.8$ to represent adult dispersal that occurs before spawning and harvest (Equations 3.17 - 3.22). In these scenarios, the proportion of the total dispersing fish that joins each sub-population is determined by applying E equally across sub-populations in order to generate a pool of dispersing fish in biomass (3.17) or numbers (3.18). This dispersing pool is then multiplied by the immigration rate $\gamma_{p,t}$, to obtain the sub-population immigrants, again either in biomass or numbers (Equations 3.21 and 3.22, respectively). Dispersal in spatial population models is frequently assumed to be density-independent, however, evolutionary and ecological theory suggest that disperal depends on local population density and other factors that affect the expected reproductive fitness of individuals in a particular site (Ruxton and Rohani 1999). Therefore, I assume that dispersing fish seek to maximize their reproductive fitness, and choose between potential spawning sites based on the quality of habitat and the local biomass of spawning fish. Additionally, I assume that Pacific herring are capable of moving between all spawning sites within the SOG, and ignore distance in this calculation. These dynamics are represented by an optimization routine that generates immigration rates that forgo large recruitments in the most productive sub-populations in favor of generating moderate recruitment across all sub-populations. Solving for the optimal immigration rate (Equation 3.20) involves maximizing the logarithm of the product of sub-population recruitments, i.e.:

$$f = log\left(\prod_{p=1}^{P} \frac{a_p(h_{p,t} + M_{p,t})}{1 + b_p(h_{p,t} + M_{p,t})}\right) \tag{3.1}$$

All operating model scenarios assume that the aggregate spawning biomass was at unfished, deterministic equilibrium prior to the directed fisheries that start in year $t = 1$. I use a single harvest trajectory to generate historical fishery data and to initialize the management procedure beginning in year 21. This initialization period is required because the stock assessment model requires a time series of data in order to achieve a good fit. The assumed harvest trajectory for the first 20 years is the average harvest rate applied across populations (i.e., $u_t = \frac{c_t}{\sum_p B_{p,t}}$), and it remains fixed across simulations (Figure 3.2). However, the stochastic dispersal and the effort-allocation sub-models produce realized local annual harvest rates that can differ dramatically from the average trajectory.

3.2.3 Management Procedures

Components of the MP are the survey, stock assessment and harvest control rule (Figure 3.1). The survey generates an unbiased fishery-independent measure of total population biomass $I_t = B_t e^{\tau \epsilon_t}$, where $\epsilon_t \sim N(0, 1)$. The MPs tested in this analysis are described in a later section and relate to alternative assumptions about the degree of error in the survey (reflected in three levels of τ), and in the quality of the management recommendations provided by the harvest control rule. All MPs exhibit perfect output control, in which the fishery is closed when the TAC is reached.

3.2.3.1 Stock assessment

The stock assessment model generates an estimate of the total pre-fishery biomass, which is later used to set annual catch limits. I elected to use a full stock assessment model instead of a simple approximation for two reasons: (1) stock assessments are an integral part of the Pacific herring management system, and (2) stock assessment errors are not independent over time, potentially leading to periods of over- or under-estimation of biomass, the implications of which are only apparent when the assessment is embedded in a simulation model of the management system (Walters 2004).

The stock assessment model is based on the delay-difference equations outlined in Table 3.3. This model assumes that the survey and catch data are collected from a single population. In each of the projection years $t \geq 21$, the assessment model generates a forecast biomass for the upcoming year (B_{T+1}), which leads to the catch limit calculation for year $t+1$ based on the harvest control rule. Generating the unfished and forecast biomass requires first fitting the model to t years of survey data in order to estimate the long-term average age-3 recruitment and the annual recruitment deviations (δ_t), and then projecting the total biomass forward one year, assuming that recruitment will be equal to the long-term average. B_0 is calculated from the stock assessment estimate of $\overline{logR}$ ($B_0 = \frac{\overline{logR}}{1-S}\overline{w}$). This value is used to parametrize the harvest control rule.

I use a penalized maximum likelihood approach to estimate the delay-difference model parameters ($\overline{logR}$, δ_t, t=*1, 2, ...*T) based on catch and biomass index data. The observation model for the biomass survey is assumed to be of the form: $I_t = \hat{B}_t e^{\xi_t}$ where $\xi_t \sim N(0, \tau^{*2})$. Inferences about the dynamics of fish stocks depend upon uncertainty in both the observations and the underlying population dynamics processes, but the maximum likelihood approach can not simultaneously estimate both process and observation error variances. An assumption must therefore be made about how to allocate random deviations in the data to either the stock dynamics or the observations, and different assumptions lead to different model estimators. I use an error-in-variables estimator that assigns an equal proportion of the total error variance to observation and process errors; this is a common default assumption when the ratio of errors is not known (Schnute and Richards 1995).

Lack of convergence of stock assessment estimators can arise during simulations because of extreme data points that create inconsistencies between the index and the catch. When these problems occur in my simulations (approximately 5-45% of the time, depending on the scenario), I generate a biomass estimate for the current year by projecting biomass from the most recent successful estimation using Equation 3.31 and the catch for the intervening years.

3.2.3.2 Harvest control rule

The harvest control rule (HCR) specifies the total allowable catch in the following year based on the forecast stock biomass ($\hat{B}_{T+1}$) and the current estimate of unfished biomass $\hat{B}_{0,t}$, where the latter is used to determine the HCR parameters: $\hat{B}_{lim,t} = 0.2\hat{B}_{0,t}$ and$\hat{B}_{targ,t} = 0.4\hat{B}_{0,t}$. These reference points were selected for the management procedure simulations because they are commonly used approximations for $0.5B_{MSY}$ and B_{MSY} (e.g Froese et al. 2010, Shelton and Sinclair 2008). Basing the HCR on estimated rather than true biomass propagates errors in the survey and stock assessment estimates through to the management decision-making function, which represents the real-world situation. The final parameter of the HCR is the reference harvest rate U_{ref}. The HCR is used to compute TAC in two steps:

$$u_{t+1} = \begin{cases} 0 & \hat{B}_{T+1} < \hat{B}_{lim,t} \\ \frac{U_{ref}}{\hat{B}_{targ,t} - B_{lim,t}}(\hat{B}_{F,t+1} - \hat{B}_{lim,t}) & \hat{B}_{lim,t} \leq \hat{B}_{T+1} < \hat{B}_{targ,t} \\ U_{ref} & \hat{B}_{T+1} \geq \hat{B}_{targ,t} \end{cases} \tag{3.2}$$

$$TAC_{t+1} = u_{t+1}\hat{B}_{T+1} \tag{3.3}$$

3.2.4 Harvest Dynamics

Information on in-season fishing dynamics is not available for the B.C. herring fishery. I therefore construct a model of the fishery dynamics based on the ideal free distribution (IFD), which characterises the relationship between fish movement, their habitat, and the fishery (Gillis 2003). For species that are distributed among habitat patches of varying quality, IFD theory predicts that harvesters will distribute themselves so that the average reward (e.g. catch per unit of effort or profit) will be equal among sites, and that fishing pressure in any patch will increase with fish availability (Fretwell and Lucas 1970). The main assumptions are that fishers have 'ideal' knowledge of the distribution of their target species, that there are no barriers (such as cost) to moving freely between patches, and that there is no limit on the amount of effort (Cox and Walters 2002, Gillis and Peterman 1998). The first two assumptions are plausible for the SOG herring fishery which relies on in-season monitoring of spawning aggregations to determine where and when to harvest. Although the final assumption suggests that effort will increase without limit over time on a given population, typical simulated daily effort during the fishing season ranges from 1 to 82 boats. This range appears to be reasonable, and perhaps underestimated, given that there are approximately 440 vessels licensed to capture herring in B.C., and that most of the catch is taken out of the SOG[1].

The harvesting dynamics sub-model (part 3 in Figure 3.1) implements a quota-based fishery in which no spatial restrictions are placed on the removal of the TAC. The sub-model outputs the daily catch by sub-population using the predicted fish availability on the spawning grounds and the fishing effort response to that availability. Derivation of this model is based on the following three assumptions: (1) on any given day, only a fraction of herring are available to the fishery, (2) the daily available biomass differs among spawning areas, and (3) the daily distribution of fishing effort among spawning areas is a function of the distribution of available biomass. Two lines of evidence support these assumptions and the resulting model structure. First, herring spawn in waves over a period of approximately 4-5 days in a single spawning area, and these events consist of early, peak, and late spawners (Hay 1985). The

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[1] Analysis of Commercial Fishing License, Quota, and Vessel Values, March 2010. Report Prepared for DFO Pacific Region, Stuart Nelson. Available online at: www.pac.dfo-mpo.gc.ca/fm-gp/picfi-ipcip/docs/2010-value-valeur.pdf

simplest way to characterise fish availability on the spawning grounds is in terms of normally-distributed daily available biomass, similar to arrival timing patterns that are used to model salmon abundance dynamics through fisheries and onto spawning grounds (e.g., Flynn et al. 2006). Second, herring exhibit considerable differences in mean spawn timing among areas, which suggests that the quality of fishing (in terms of available biomass) varies among locations, and when combined with the daily availability dynamics, implies that the quality of fishing is likely to change from day to day.

Daily availability patterns for the P simulated sub-populations are based on spawn timing records collected by DFO between 1928 and 2006 in the SOG (Pacific herring database, Pacific Biological Station, Nanaimo B.C.). Spawn records are collected by spawning location (individual bay or beach) and aggregated in a hierarchical manner into spawning 'sections', management areas, and region. The distribution of recorded spawn times varies considerably within and among sections over the time series (Figure 3.3). I reflect this variability by drawing a random mean day of spawning ($m_{p,t}$) for each of the simulated populations from the section-specific data each year. Thus, each of the simulated sub-populations corresponds to a herring spawning section. I assume that the daily biomass of herring arriving at each spawning section follows a bell-shaped curve (Equation 3.40), where $y_{p,t}$ is the total biomass and l determines the spread of arrival over days d.

Following previous authors, fishing effort is assumed to vary linearly with changes in biomass with a slope that is inversely proportional to the IFD target catch rate c_o (Cox and Walters 2002). The daily catch calculation (Equation 3.41) incorporates a minimum available biomass (A_{min}, Equation 3.38) below which fishing effort is not attracted, and a maximum exploitation rate (U_{max}, Equation 3.39) on each population (Cox and Walters 2002). The daily (and annual) catch can differ substantially among spawning areas because of variation in available biomass and the harvesting response. Therefore, the total annual harvest rate applied to each sub-population during the fishing season in year t ($f_{p,t}$) may differ substantially from the target U_{ref}. Parameters c_o and φ are used to set the harvest dynamics component of the simulation scenarios (see below).

3.2.5 Simulation Scenarios

All simulation scenarios reflect a management system that ignores spatial complexity. Three components of this system are used to structure the scenarios: the operating model, management procedure, and harvest dynamics model. The latter is technically a component of the operating model, but is separated here for clarity and because aspects of this model can be influenced by management. The operating model sets the form of connectivity among sub-populations (discrete, $E = 0$ and mixed, $E = 0.8$) as well as the degree of recruitment compensation in the sub-populations determined by $F_{opt} = 0.6M$ or M, for low and high productivity, respectively. The MP options involve high, medium, and low survey precision, $\tau^* = 0.1$, 0.25, 0.50, respectively, and two levels of reference harvest rate in the HCR: $U_{ref} = 1 - e^{-F_{MSY}}$ and $U_{ref} = 1 - e^{-2.5F_{MSY}}$, where F_{MSY} is the true (operating model) fishing mortality rate that yields B_{MSY} on average for the population aggregate. These HCRs reflect different scenarios of the quality of management recommendations. The harvest dynamics model includes alternative hypotheses about the profit required to attract fishing effort to a sub-population, the maximum local harvest rate on a single sub-population, and the duration of the fishing season within the SOG. I use three levels of the IFD target catch rate c_o (1, 10, and 20 t/boat) to reflect a range of profitability requirements, where at the lower limit ($c_o = 1$) the distribution of fishing effort is essentially unconstrained by profit and at the upper limit ($c_o = 20$) the fishery is constrained by a requirement to achieve a relatively high profit. These values were selected in the context of the simulation model and do not apply specifically to the B.C. herring fishery. In the real fishery, the catch per boat can vary dramatically, and depends on the gear type: common amounts are *125* tonnes per set in the purse seine fishery, and

3 tonnes per skiff in the gillnet fishery[2].

The harvest rate is determined by the number of days (φ) that the fishery is allowed to operate on a single sub-population. Values of $\varphi = 1$, 3, and 10 days reflect maximum local harvest rates of 0.4, 0.8, 1, respectively. The total duration of the fishing season over all sub-populations is set to either 40 or 100 days.

A "simulation" consists of a particular combination of each of the parameter settings (Table 3.5). All simulations represent a modification of one baseline parameter from the baseline case, which permits evaluation of the sensitivity of the results to each parameter. Simulations are grouped into sets based on the operating model and fishing season scenario (i.e., discrete, low compensation, 40-day fishery). Within each set, the baseline case settings are: $\tau = 0.25$, $F_{scale} = 1$, $c_o = 20$, and $\varphi = 10$.

3.2.6 Performance Indicators

Performance indicators are used to determine the ability of a management strategy to achieve the stated objectives for the fishery. The objectives used to develop the indicators for this analysis are generic but generally accepted objectives of maintaining stock biomass at B_{MSY} or higher (i.e., avoid depleting the stock to an overfished state), achieving a fishing mortality rate of F_{MSY} or lower (i.e., avoid overfishing), generating reasonable economic returns from the fishery, and ensuring that the spatial structure of the population aggregate is not compromised. In this study, the population scenarios influence stock productivity and equilibrium yield, so it is inappropriate to use total catch as an economic performance indicator. The catch comparisons for each population scenario are therefore presented with respect to a catch trajectory that is based on the 'optimal' harvesting strategy of F_{MSY} for each sub-population. I use the ratio of sub-population biomass to sub-population-specific $B_{MSY,p}$ to reflect the state of spatial structure in the population aggregate.

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[2] Integrated Fisheries Management, Roe Herring. 2010 Coastwide Season Review. Available online at: www.pac.dfo-mpo.gc.ca/consultation/fisheries.../2010-04-29-roe-post.pdf

The analyses involve 100 Monte Carlo trials for each scenario (Table 3.5), in which the operating model is projected 55 years into the future following 20 years of a fixed harvest rate trajectory. Indicators are calculated for the final 25 years of each simulation to avoid short-term transient patterns that might provide a misleading impression of the sustainability of some options compared to the long term performance. Performance indicators computed over this time period are:

1. Average annual ratio of total biomass to B_{MSY} for the population aggregate,
2. Average annual ratio of realized total harvest rate to the harvest rate at F_{MSY} for the population aggregate,
3. Average annual ratio of the total realized catch to the optimal total catch for the population aggregate, and
4. The average proportion of sub-populations that are overfished ($B_p < B_{MSY,p}$).

3.3 Results

3.3.1 Equilibrium Properties: Relationships Between Connectivity and Recruitment

Variation in recruitment among sub-populations arises from differences in habitat quality (H_p) that govern the parameters of the sub-population recruitment functions (Figures 3.4 and 3.5). Habitat quality also determines the magnitude of maximum sustainable yield (MSY) in the discrete populations (Figure 3.4); however, the impact of habitat is moderated in the mixed populations by density-dependent dispersal (Figure 3.5), which evens out the distribution of spawning biomass among habitats and increases their MSY relative to the discrete populations.

The effect of recruitment compensation on recruitment and MSY is minor relative to the combined effect of habitat quality and (for the mixed sub-populations) dispersal, but MSY increases with the degree of compensation for both population scenarios (Figures 3.4 (b) and 3.5 (b)). In addition, the redistribution of biomass between sub-populations depends on the recruitment compensation scenario. Under low compensation the highest proportion of migratory biomass is allocated to sub-populations that have either high- or low-quality spawning habitats (marked with an asterisk on Figure 3.5 (a)). In contrast, under high compensation, the highest proportion of migratory biomass is allocated to the highest quality spawning habitats (marked with an asterisk on Figure 3.5 (b)).

3.3.2 Patterns of Migration and Fishing Effort Allocation

This section is focused on understanding the interaction between connectivity, arrival timing on the spawn/fishing grounds, and fishing. MP options are therefore not explored here, instead I examine how changes in fishery dynamics affect the populations. The results of one example simulation using the baseline settings under low recruitment compensation in both the 40- and 100-day fishing seasons are shown on Figures 3.6 and 3.7 for the discrete and mixed populations scenarios, respectively. The baseline settings represent an MP with moderate levels of survey error, a stock assessment model that overestimates total biomass, and an HCR that sets the TAC based on a combination of forecast biomass, the estimated B_0, and the 'true' F_{MSY}. Fishery scenarios under the baseline allow daily exploitation rates of up to 99% on each sub-population, and the fishery requires relatively high profits in order to go fishing (i.e., the sub-population biomass must be high enough to achieve the IFD target catch rate $c_o = 20$ tonnes per boat).

Differences between discrete and mixed populations that were evident in the equilibrium analysis (Figures 3.4 and 3.5) are maintained in stochastic projections: the mixed sub-populations achieve a higher average biomass than the discrete sub-populations because high rates of movement redistribute production among spawning sites to achieve a relatively homogenous distribution. Within the mixed-populations scenarios, migrants account for approximately half of the biomass moving onto the spawning grounds (Figure 3.7).

Spawn timing plays an important role in determining the impact of the fishery on the sub-populations. For example, populations that migrate onto the spawning grounds outside of the fishing season realize substantially lower harvest rates than populations whose spawn timing corresponds more closely with the fishing season opening (Figures 3.6 (a), 3.7 (a)). When the fishing season is extended to 100 days, the populations with higher biomass and early spawn timing are more heavily exploited by the fishery (Figures 3.6 (b), 3.7 (b)).

Reducing the IFD catch rate to $c_o = 1$ tonne per boat indicates that the profit threshold of the fishery is an important determinant of the realized harvest rate on the sub-populations (compare Figures 3.6 and 3.7 with 3.8 and 3.9). Lowering the profitability requirement under a 40 day fishing scenario results in the removal of all biomass available to the fishery due to high exploitation rates on sub-populations whose spawn timing overlaps with the timing of the fishing season. Under this scenario, the early-spawning, discrete populations are maintained at relatively high levels of biomass because spawn timing confers a refuge from harvesting (Figure 3.8 (a)). The migratory populations exhibit an additional, indirect impact of high exploitation rates for the 40 day fishery, where the early-spawning stocks are depleted to low levels of abundance despite low harvest rates (Figure 3.9 (a)). This indirect impact is a result of the redistribution of production from the early-spawning populations to the heavily fished, late-spawning populations. The scenario of a long 100-day fishing season and low profitability threshold removes all spatial and temporal inseason controls on fishing, and depletes both the discrete and mixed sub-populations to very low levels (Figures 3.8 (b) and 3.9 (b)).

3.3.3 Factors That Affect Management Performance

3.3.3.1 Stock assessment

The stock assessment is an important source of error in the management system. Its impact on management performance is evident in the 100-day fishing season, low-profitability threshold scenario, which allows the impact of unsustainably high TACs to be realized on all sub-populations (Figures 3.8 (b) and 3.9 (b)). This length of season increases the vulnerability of the management system to errors that are pervasive in fisheries management, namely, biased stock assessment estimates of stock size and errors in the estimates of management parameters used to generate TAC recommendations (B_0 in this analysis). Eliminating the stock assessment and using the true B_0 and true biomass in the HCR under the same simulation settings (i.e., mixed populations, 100 day fishing season, $c_o = 1$) improves the conservation outcomes substantially, yielding a 20-fold increase in average total biomass relative to the simulations that included a stock assessment model (compare Figures 3.9 (b) and 3.10). This result also demonstrates that the stock assessment process introduced substantial error into the fishery management system.

3.3.3.2 Baseline results

Baseline settings of the MP and harvest dynamics sub-model achieve a high ratio of total biomass to total B_{MSY} (i.e., the population aggregate is not overfished) and a desirable ratio of fishing mortality to F_{MSY} (i.e., overfishing is not occurring) over all scenarios of population connectivity and recruitment compensation for the 40-day fishery (Figure 3.11). In contrast, performance measured for the sub-populations is substantially different from the aggregate in terms of biomass conservation, an average of 50% and 82% of discrete and mixed sub-populations are overfished under the baseline model settings across Monte Carlo trials for both scenarios of recruitment compensation (calculated using baseline values shown on Figure 3.11 (a), (b) and (c), (d)). Extending the fishing season to 100 days degrades the performance in terms of increasing the median baseline value ${}^{F}\!/_{F_{MSY}} > 1$, but the population aggregate is not overfished (indicated by the median baseline values of ${}^{B}\!/_{B_{MSY}} > 1$, Figure 3.12). The average percentage of sub-populations that are overfished in the 100 day fishing scenario increases significantly for both discrete and mixed populations with the latter averaging close to 100% across both scenarios of recruitment compensation (Figure 3.12 (c) and (d) histograms).

3.3.3.3 Simulation testing of MPs and harvest dynamics

Simulation testing of the MP simulations achieved the same outcomes as the baseline across both scenarios of population connectivity and recruitment compensation for the 40 day fishery scenario (Figure 3.11). This result indicates that changes in the quality of both the survey and catch recommendations obtained from the HCR have little impact on management performance given the harvesting conditions represented by the baseline scenario. The simulation set was reduced for the 100 day fishery scenario based on these results to exclude any further consideration of the MP options.

Parameters of the harvest dynamics sub-model had large impacts on management performance when compared to the baseline for both the 40-day and 100-day fishery scenarios (Figures 3.11 and 3.12). This sub-model controls different features of the fishery: the number of days the fishery is allowed to operate on a sub-population (φ) limits the maximum local exploitation rate (U_{max}), whereas changes in the IFD target catch rate (c_o) determine whether a sub-population will attract fishing effort on a given day. Holding all other baseline settings constant and reducing φ to 1 from the baseline of 10 moves the fishery system toward a more biologically conservative state, as reflected by the increased biomass ratio, reduced fishing mortality rate, and reduced number of sub-populations that are overfished, regardless of population or fishing season scenario (Figures 3.11 and 3.12). The impact of the non-linear relationship between φ and U_{max} (Equation 3.39) is evident when comparing the $\varphi = 3$ and $\varphi = 1$

options with the baseline. Reducing φ from 10 to 3 causes relatively little change in the fishery system, but a reduction to 1 day results in a system that is substantially more biologically conservative for both the population aggregate and sub-populations, at the cost of lost fishing opportunities (lower C/C_{opt}) in the 40 day fishery scenario. This imbalance is rectified by increasing the duration of the fishery to 100 days, which increases fishing opportunities. This is reflected in a higher C/C_{opt} for the $\varphi = 1$ option under the 100 day fishery scenario (for example, compare C/C_{opt} on Figures 3.11 (a) and 3.12 (a)).

The IFD target catch rate (c_o) scenarios isolate the impact of changes in profitability of fishing the sub-populations. The baseline setting of 20 tonnes per boat represents a high-profit scenario within the context of the simulation model. A comparison of single simulation outcomes for $c_o = 20$ (Figures 3.6 and 3.7) and $c_o = 1$ (Figures 3.8 and 3.9) indicated that lowering the profit requirement to fish on a sub-population yielded large increases in harvest rates, and substantially depleted the populations. This result is supported in the performance measured over all simulations, in which c_o values of 10 and 1 lead to reduced total biomass, increased fishing mortality on the population aggregate, and an increase in the number of sub-populations that are overfished compared to the baseline (Figures 3.11 and 3.12). Additionally, reducing the profitability requirement leads to sub-optimal long term economic performance, as reflected in the ratio of catch to optimal catch.

3.3.4 Performance Tradeoffs Among Management Options

The results thus far point to three system features that have large impacts on management performance: the duration of the fishing season (40 days versus 100 days), the number of days the fishery is open on individual sub-populations (φ), and the IFD target catch rate (c_o), which determines whether a given sub-population will be exploited. Each feature has benefits and risks in terms of conservation and exploitation; the performance tradeoffs among the management options represented by these features are summarized in this section. For example, a long fishing season spreads exploitation among sub-populations, and increases the average harvest rate on the population aggregate (e.g., Figure 3.7). Reducing the number of days open on individual sub-populations leads to more biologically conservative management outcomes, but leads to sub-optimal economic performance and under-utilization of the available biomass (Figures 3.11, 3.12, 3.13). Finally, reducing the target catch rate (c_o) increases exploitation rate of sub-populations and over-exploits the total biomass, which also leads to sub-optimal economic performance over the long term.

Tradeoffs between conservation and exploitation for the harvest dynamics options related to the duration of local fishery openings, φ reflect the previous finding that reducing the number of days leads to conservative outcomes as reflected in ratios ${}^{B}/_{B_{MSY}} > 1.0$ and ratios ${}^{F}/_{F_{MSY}} < 1.0$, regardless of the duration of the length of the fishing season (Figures 3.13 and 3.14 (a), (b)). This general pattern is maintained across both recruitment compensation scenarios, but increasing compensation shifts the tradeoffs for all scenarios toward lower ${}^{B}/_{B_{MSY}}$. This suggests that similar results can be obtained for different local fishery durations, depending on the degree of compensation (i.e., similar biomass ratios are obtained for a 3 day opening on low compensation populations and a 10 day opening on high compensation populations (Figures 3.13 and 3.14 (a), (b)). Relative to compensation, the difference in ${}^{B}/_{B_{MSY}}$. and ${}^{F}/_{F_{MSY}}$. for the discrete and mixed populations is minor, which suggests that changing the duration of local fishery openings will affect spatially structured populations similarly, regardless of the pattern of connectivity.

Results for the harvest dynamics options that explore the impact of alternative levels of the IFD target catch rate (c_o) show that management performance is strongly dependent on this parameter. Low profit thresholds produce populations that are both overfished and subject to overfishing, while high thresholds maintain the aggregate population at more conservative levels of biomass and fishing mortality. This trend is independent of the fishing season duration (Figures 3.13 and 3.14 (c), (d)). Similar to the results for local fishery openings, the pattern in median outcomes is shifted toward lower ${}^{B}/_{B_{MSY}}$. in the high recruitment compensation simulations. However, unlike the previous scenarios, connectivity appears to be an important determinant of the impact of changing c_o, with mixed populations realizing higher fishing mortality rates and being depleted to lower biomass levels than the discrete populations in the 40 day fishing season (Figure 3.13 (c), (d)). This difference is eliminated under the 100 day season, when all populations are depleted to low levels of biomass, and the degree of recruitment compensation determines the intensity of overfishing, with high compensation populations realizing higher ${}^{F}/_{F_{MSY}}$ than low compensation populations (Figure 3.14 (c), (d)).

3.4 Discussion

3.4.1 Evaluation of the Simulation Model

Previous simulation work suggests that ignoring spatial structure in the application of management advice (i.e., placing no spatial controls on fishing effort) may lead to a loss of sub-populations and erosion of spatial complexity within stock complexes (de la Mare 1996, Frank and Brickman 2000). However, many of these studies are based on unrealistic, simplifying assumptions about the dynamics of the fish populations, the fishery, or both, which makes it difficult to gain a clear understanding of the potential impacts of a mis-match between scales of management and population dynamics. For example, existing studies allocate biomass and recruitment equally among separate stock areas, allow for little or no exchange of individuals between stock areas, and assume that fishing mortality is distributed homogeneously across space (Cope and Punt 2011, Frank and Brickman 2000, Ralston and O'Farrell 2008). The model developed here extends this body of work by relaxing those restrictive assumptions by explicitly simulating these processes and evaluating their impact on management outcomes. In addition, by including an effort-dynamics component, the model represents a social-ecological system of a mobile fishing fleet operating on biologically and spatially diverse fish populations.

3.4.2 Insights Gained From Simulations

The main result of the simulations is that dynamics of the fishing fleet/harvesting component included in the simulation model more strongly influence management performance than the components of spatial complexity (connectivity and recruitment compensation). This result suggests that leverage in the management system exists in the incentives driving the spatial distribution of the fishery, and in the controls placed on the spatial distribution and magnitude of effort, both of which have been recognized by previous authors (e.g. Sanchirico and Wilen 2005, Walters and Pearse 1996). However the additional insights provided here, and discussed in subsequent sections, are:

1. Management success is not always governed by the spatial resolution (scale) of stock assessment,
2. Dispersal may actually increase the vulnerability of spatially structured populations to overfishing,
3. Spatial complexity in populations and fisheries provides a buffer against errors in management.

3.4.2.1 Management success and the scale of stock assessment

Persistent uncertainty about fish population dynamics and structure, the high cost of quantifying spatial structure, and issues of data sufficiency represent practical constraints on the scale of management for many spatially structured fish populations (e.g. Cope and Punt 2011, Punt and Donovan 2007, Waples and Gaggiotti 2006).

However, there is a growing concern that failing to manage spatially structured fish populations on the appropriate scale - i.e., not assessing and setting TACs for individual sub-populations - will promote overfishing, contribute to a loss of spatial diversity, and restrict the ability of populations to adapt and persist across a broad range of environmental conditions (Stephenson 1999). in contrast, my results indicate that overfishing and stock depletion do not automatically follow from a mis-match between the scale of population structuring and the scale of assessment and management. Instead, management success may be more determined by: (1) the incentives for the fishery to harvest a particular population (profitability threshold), and (2) fishing opportunity, which is governed by the spatial and temporal overlap between the populations and the fishery. The interaction of these factors is an important determinant of the state of the fishery. For example, the simulated populations varied in both their timing and distribution of biomass on the spawning/fishing grounds, which controlled the daily availability of each population to the fishery. In a system where the movement of fishing vessels to harvest sub-populations is driven by profit, low levels of available biomass do not attract high levels of fishing effort. This dynamic creates refugia from fishing at the sub-population level and also within a fishing season because daily availability dynamics expose only a fraction of each sub-population to fishing. Such refugia are strongest for the case where vessels need a high profit and short-fishing-season simulation. This scenario combines spatial and temporal refuges to achieve the most biologically conservative outcomes. In contrast, processes that intensify the overlap between the spatial distribution of fishing effort and the harvested populations promote high levels of fishing effort on a local scale, and contribute to overfishing of the aggregate population. Examples of overlap-intensifying processes include: (1) a low profit threshold in the effort dynamics model, which increases harvesting at low levels of biomass, (2) an extended fishing season, which permits harvesting on all spawn timing groups both within and among sub-populations, and (3) mixing between sub-populations. The latter process is discussed in the next section.

3.4.2.2 Dispersal and relative vulnerability to overfishing

The collapse of spatially complex fish populations such as Atlantic cod (*Gadus morhua*) and North Sea herring (*Clupea harengus*) has been widely attributed to the failure of fisheries management to protect and maintain within-population complexity (Cury et al. 2000, **?**). Recognition of this failure has permeated the thinking of fisheries scientists, and an increasing number of studies consider the management implications of ignoring spatial complexity. An interesting feature of much of this work is that it focuses on the threat that aggregate management poses for discrete populations (e.g. Frank and Brickman 2000, Wilson et al. 1999). These concerns appear to arise from a widely-held assumption that high rates of mixing confer resilience to a population complex via the "rescue effect", wherein high rates of straying decrease local extinction rates in spatially structured meta-populations (Dulvy et al. 2003). From a management perspective, a high rate of mixing is therefore viewed as removing the requirement to manage at a fine spatial scale (Cope and Punt 2011). The results of this study do not support this assertion; the mixed populations fared worse than the discrete populations under most fishing scenarios.

The primary difference between the connectivity scenarios explored in this chapter is that the dispersal function in the mixed-populations scenarios favors a homogeneous distribution of biomass, and re-allocates biomass from high productivity populations to low productivity populations. Thus, the impact of fishing (or not fishing) is shared among sub-populations in the high connectivity scenario. In constrast, the impact of fishing is localized in the discrete-populations scenarios. The difference between outcomes for discrete- and mixed-populations indicate that the spatial and temporal dynamics of contingents define the fraction of the population that constitutes the exploited stock. Therefore, as illustrated in this study, the impact of fishing on spatially complex populations can not be inferred simply from the characteristics of the population or the scale of management. Furthermore, depending on the nature of the population and the fishery, mixed populations may be more vulnerable to overfishing than discrete populations.

3.4.2.3 Spatial complexity buffers errors in management

Complex spatial structure is increasingly viewed as an important mechanism that promotes long-term persistence and productivity of marine fish populations (Kerr et al. 2010, Schindler et al. 2010, Stephenson 1999). The benefits of spatial complexity may extend beyond the population to the broader ecosystem and the fishery, as was shown for sockeye salmon (*Oncorhynchus nerka*) in Alaska, where variation in spawn timing and location stabilized fishing opportunities by enhancing the in-season availability of salmon (Schindler et al. 2010). My results further suggest that spatial complexity can act as a buffer against common errors in fishery management. Stock assessment estimates of abundance are often highly unreliable because the fundamental assumptions underlying the models are flawed (common examples include stationary production relationships and catch per unit effort proportional to abundance) (Hilborn and Walters 1992). In addition, different stock assessment methods can differ dramatically in their ability to estimate HCR reference points such as B_0 (Haltuch et al. 2008). My results indicate that the impact of stock assessment errors on management performance depends on the degree of overlap between the populations and the fishery. Removing all spatial and temporal controls on fishing enabled the fishery to capture the entire TAC by fishing on all populations. If the TAC was set to an appropriate level (e.g. the scenario in which the HCR used 'true' biomass and B_0), good management outcomes were achieved. However, if the TAC was based on biased stock assessment estimates (the realistic scenario), this management strategy promoted overfishing and poor long-term catch performance. Imposing spatial and temporal controls on fishing restricted harvesting to a subset of the populations and greatly improved conservation and fishery outcomes, even in the presence of biased assessments. Given the pervasiveness of stock assessment problems in fisheries agencies, recognizing and managing fisheries for spatial complexity may increase the degree of precaution in quota-based management systems.

3.4.3 Caveats and Limitations

The structure of the simulation model omits some important dynamics and processes that could affect the results. Foremost, the model is spatially implicit and therefore the harvest dynamics component does not account for the costs associated with travel time and distance between fishing locations, which could influence the distribution of the fishery such that locations that are close to each other and to fishing ports realize higher exploitation rates than more distant locations (Hilborn and Walters 1987). However, because the modelled region of the SOG is small relative to the dispersive capacity of the BC herring fleet, travel costs are justifiably ignored in this study. Another important feature of the model structure is that the population dynamics do not generate patterns of abandonment and recolonization of spawning locations arising from local extirpation or natural changes in patterns of dispersal, as is expected under some models of spatial population dynamics (Hanski 1998). This means that permanent loss of spatial diversity is not represented by the model. Instead, spatial diversity is 'eroded' as the sub-populations are depleted to low levels of biomass under exploitation rates that vary significantly within and between years.

3.4.4 Management Implications

This study provides a good starting point for additional, fishery-specific simulation analyses on the impact of ignoring spatial complexity in management. For example, the fishing effort dynamics model in this analysis was based on parameters and assumptions that may not apply to the B.C. herring fishery. This limitation arises largely because effort dynamics information is not collected for the fishery. However, future work could be informed by interviews with managers, industry members, and fishery scientists to develop a more realistic model of the fishery. A similar limitation exists for the population simulator, which is based on generalizations of herring dynamics and makes no attempt to reproduce historical trends in biomass and exploitation specifically for B.C. herring. However, the objective of the study was to explore a general fishery management system that ignores spatial complexity in both the fish populations and fishery, and to identify points at which intervention achieves large changes in management performance. These general management implications are not expected to change with a more specific model.

My analysis suggests that the combined spatial dynamics of fish populations and fisheries have powerful implications for fisheries management. In particular, the aspects of spatial structure that appear to be most important for management (i.e., the processes governing the overlap between populations and fisheries) may not be reflected in standard measures of stock structure such as genetic diversity. This issue is acknowledged by researchers in stock identification (Cadrin 2005), who note that the definition of a 'fish stock' is fluid, and depends on the reason for defining a stock in the first place (Carvalho and Hauser 1994). My results indicate that the fishery plays an important role in defining a stock for management, which is a perspective that echoes the "harvest stock" concept that identifies stocks based on the accessibility of fish to the fishery (Gauldie 1988). This implies that the spatial and temporal overlap between the fish and the fisheries is a fundamentally important characteristic of spatially complex populations. Furthermore, the economic aspects of the fishery appear to be important for defining a "stock" because they determine the profit incentives and opportunities for harvesting. Unfortunately, the information required for such economic evaluations (e.g. vessel searching activity, catch locations, catch rate, fishing costs) are not routinely collected for many fish populations, including B.C. herring. If such information was available, the relationship between the spatial distribution of fishing effort and profit could be used to create (or preserve) refuges from harvest within seasons and across space. Management tactics involving spatio-temporal harvest refugia were historically successful for Pacific salmon and Australian rock lobster (*Panulirus cygnus*) (Walters and Pearse 1996).

The number of days the fishery is permitted to harvest an individual sub-population had a significant impact on management performance in my analysis. This result indicates that in the absence of other types of intervention, controlling the local impact of the fishery is an important point of leverage for managing complex populations. In-season management has been used to successfully manage fisheries when fish availability and abundance vary dramatically throughout the fishing season, and when fisheries are managed for spawning escapement and exploitation rate goals (Walters and Martell 2004). It has also been used to manage Atlantic herring fisheries in order to prevent the loss or erosion of individual spawning populations in an aggregate, quota-based management system (Stephenson et al. 1999). An additional benefit of in-season management is that it creates an opportunity to understand the dynamics of the fish populations and the fishery because it requires intensive sampling of both in order to inform decisions on a daily or weekly basis. It may therefore be a necessary precursor to incorporating economic incentives and profit into management decisions.

Table 3.1: Notation for the fishery operating model components. Bold text denotes fixed parameters common to the population simulator and stock assessment model used in the MP, p indexes sub-populations, multiple values for variables are enclosed in braces.

Symbol	**Value**	**Description**
Indices		
t	1, 2,..., T	Annual time step (T = 40)
p	1, 2,...,P	Sub-population unit (P = 12)
d		Day of year
Model parameters		
B_0	100000	Aggregate unfished biomass (tonnes)
H	$\{H_1, H_2, ..., H_P\}$	Proportion of suitable spawning habitat in sub-population area
$A\cdot$	1000	Total spawning area occupied by all sub-populations
S	**0.63**	Natural annual survival rate
E	$\{0, 0.8\}$	Proportional annual emigration rate
F_{opt}	$\{0.6M, M\}$	Target optimal fishing mortality rate (yr^{-1})
k	**3**	Age at recruitment (yr)
w_k	**0.000091**	Weight per fish at recruitment ($tonnes$)
σ	0.25	Standard error of log-recruitment deviations
ρ	**0.7067**	Walford growth function slope parameter
α	**0.00004**	Walford growth function intercept ($tonnes$)
φ	$\{1, 3, 10\}$	Duration of local fishery opening ($days$)
c_o	$\{1, 10, 20\}$	Target catch rate ($tonnes\ boat^{-1} day^{-1}$)

(Table 3.1 continued)

Symbol	Value	Description
q	0.35	Fishery catchability (yr^{-1})
m	$\{m_2, m_2, ..., m_P\}$	Sub-population mean spawning day
l	5	Residence time on spawning ground ($days$)
v	0.5	Rate at which fish become available to the fishery *(day^{-1})*
Population aggregate		
s		Total survival rate accounting for emigration
$\bar{w}$		Equilibrium unfished mean body weight ($tonnes$)
δ_t		Estimated log-recruitment deviations in year t
$R_{\cdot,t}$		Total recruitment
$N_{\cdot,t}$		Total post-fishery numbers
$B_{\cdot,t}$		Total post-fishery biomass ($tonnes$)
$\hat{B}_{T+1}$		Stock assessment total biomass forecast ($tonnes$)
Sub-population (p)		
K_p		Recruitment compensation coefficient
B_p		Unfished biomass
$R_{p,0}$		Unfished recruitment
$N_{p,0}$		Unfished numbers
a_p		Beverton-Holt slope parameter
b_p		Beverton-Holt capacity parameter
State variables		
$R_{p,t}$		Annual recruitment
$n_{p,t}$		Pre-fishery numbers

(Table 3.1 continued)

Symbol	Description
Dispersal	
$N_{p,t}$	Post-fishery numbers
$y_{p,t}$	Pre-fishery biomass (*tonnes*)
$B_{p,t}$	Post-fishery biomass (*tonnes*)
D_t	Pool of total dispersing biomass (*tonnes*)
$\tilde{D}_t$	Pool of total dispersing numbers
$h_{p,t}$	Resident biomass (*tonnes*)
$M_{p,t}$	Immigrant biomass (*tonnes*)
$\acute{M}_{p,t}$	Immigrant numbers
$\gamma_{p,t}$	Proportion of dispersing pool immigrating to sub-population p
Observations	
I_t	Index of total population biomass (*tonnes*)
Fishery controls	
$C\cdot_t$	Total catch (*tonnes*)
$C_{p,t}$	Sub-population catch (*tonnes*)
u_t	Total annual exploitation rate
$f_{p,t}$	Sub-population exploitation rate

Table 3.2: Delay-difference model used to evaluate management procedures. Equations govern the recruited biomass and numbers.

Parameters

(3.4) $$\Theta = \{T, B_0, A\cdot, H, E, u, \sigma, \rho, \alpha, S, w_k, k\}$$

Initial State

(3.5) $$\bar{w} = \frac{s\alpha + w_k\,(1-s)}{1-\rho s}$$

(3.6) $$B_{p,0} = B_0\left(\frac{A_p}{A\cdot}\right)$$

(3.7) $$N_{p,0} = \tfrac{B_0}{\overline{w}}\left(\frac{A_p}{A\cdot}\right)$$

(3.8) $$R_{p,0} = N_{p,0}(1-s)$$

(3.9) $$a_p = K_p\frac{R_{p,0}}{B_{p,0}}$$

(3.10) $$b_p = \frac{K_p - 1}{B_{p,0}}$$

State Dynamics

(3.11) $$\omega_t \sim N(0,1)$$

(3.12) $$R_{p,t} = \begin{cases} R_{p,0} & 1 \leq t < k \\ \frac{a_p B_{p,t-k+1}}{1+b_p B_{p,t-k+1}} e^{(\omega_t\sigma - 0.5\sigma^2)} & k \leq t \leq T \end{cases}$$

(3.13) $$n_{p,t} = \left[S N_{p,t-1} + R_{p,t}\right](1-E) + \tilde{M}_{p,t}$$

(3.14) $$N_{p,t} = (1-f_{p,t})n_{p,t}$$

(3.15) $$y_{p,t} = \left[S(\alpha N_{p,t-1} + \rho B_{p,t-1}) + w_k R_{p,t}\right](1-E) + M_{p,t}$$

(3.16) $$B_{p,t} = (1-f_{p,t})y_{p,t}$$

(Table 3.2 continued)

Dispersal

(3.17) $$D_t = E\sum_{p=1}^{P}(S(\alpha N_{p,t-1} + \rho B_{p,t-1}) + w_k R_{p,t})$$

(3.18) $$\tilde{D}_t = E\sum_{p=1}^{P}(SN_{p,t-1} + R_{p,t})$$

(3.19) $$h_{p,t} = (1 - E)D_t$$

(3.20) $$\gamma_{p,t} = \frac{e^{(\beta_{j,t})}}{\sum_{j=1}^{P} e^{(\beta_j,t)}}$$

(3.21) $$M_{p,t} = \gamma_{p,t}D_t$$

(3.22) $$\acute{M}_{p,t} = \gamma_{p,t}\tilde{D}_t$$

Table 3.3: Delay-difference stock assessment model and likelihood function used in the simulated management procedure for the aggregate population.

Estimated Parameters

$$\Theta = \left\{\overline{logR}, \hat{\delta}_{2\ldots T}\right\} \tag{3.23}$$

Assessment

Initial Conditions ($t = 1$)

$$\bar{w} = \frac{s\alpha + w_k\left(1 - s\right)}{1 - \rho s} \tag{3.24}$$

$$N_0 = \frac{\bar{R}}{1 - s} \tag{3.25}$$

$$B_0 = N_0\bar{w} \tag{3.26}$$

State Dynamics ($t \geq 2$)

$$R_t = \bar{R}e^{\widehat{\delta_t}} \tag{3.27}$$

$$u_t = \frac{C_t}{B_t} \tag{3.28}$$

$$N_t = sN_{t-1}(1 - u_{t-1}) + R_t \tag{3.29}$$

$$B_t = s(1 - u_{t-1})(\alpha N_{t-1} + \rho B_{t-1}) + w_k R_t \tag{3.30}$$

Forecast

$$B_{T+1} = s(1 - u_T)(\alpha N_T + \rho B_T) + w_k\bar{R} \tag{3.31}$$

Likelihood

Conditional maximum likelihood estimates

$$\widehat{\log q} = \frac{1}{T}\sum_t \log\left(\frac{I_t}{\widehat{B_t}}\right) \tag{3.32}$$

$$\xi_t = \log\left(\frac{I_t}{\widehat{B_t}}\right) - \widehat{\log q} \tag{3.33}$$

(Table 3.3 continued)

(3.34) $$\tau^{*2} = \frac{1}{T}\sum_{t=1}^{T}\xi_t^2$$

(3.35) $$\sigma^2 = \frac{1}{(T-1)}\sum_{t=2}^{T}\hat{\delta}_t^2$$

Negative Log-likelihood

(3.36) $$\ell = \frac{2T-1}{2}\log\left(\frac{1}{0.5}\sum_{t=1}^{T}\xi_t^2 + \frac{1}{(1-0.5)}\sum_{t=2}^{T}\widehat{\delta_t}^2\right)$$

Table 3.4: The harvest dynamics component of the operating model, that is used to generate exploitation rates for the sub-populations.

Parameters

(3.37) $\Theta_t = \{y_{p,t}, c_o, l, m_{p,t}, \varphi, q, v\}$

(3.38) $A_{min} = \frac{2c_o}{q}$

(3.39) $U_{max} = 1 - e^{\varphi v}$

Fish availability

(3.40) $A_{p,d,t} = y_{p,t} \frac{e^{\frac{-(d-m_{p,t})^2}{2l^2}}}{\sum\limits_k e^{\frac{-(k-m_{p,t})^2}{2l^2}}}$

Daily catch by sub-population

(3.41) $C_{p,d,t} = (A_{p,d,t} - A_{min})\, U_{max}$

Sub-population exploitation rate

(3.42) $f_{p,t} = \frac{1}{y_{p,t}} \sum\limits_d C_{p,d,t}$

Table 3.5: Parameter settings used to represent the sets of simulations (Baseline and S1...S7) explored in this analysis. One baseline parameter is changed per simulation (boldface in the table) in order to permit an evaluation of the sensitivity of the baseline results to each parameter. All sets of simulations in this table were run for the 40-day fishing season scenario; a subset was run for the 100-day scenario.

Connectivity	Compensation	Parameter	Baseline	S1	S2	S3	S4	S5	S6	S7
Discrete	Low	τ	0.25	0.25	0.25	0.25	0.25	***0.1***	***0.5***	0.25
		F_{scale}	1	1	1	1	1	1	1	***2.5***
		c_o	20	***1***	***10***	20	20	20	20	20
		φ	10	10	10	***1***	***3***	10	10	10
Discrete	High	τ	0.25	0.25	0.25	0.25	0.25	***0.1***	***0.5***	0.25
		F_{scale}	1	1	1	1	1	1	1	***2.5***
		c_o	20	***1***	***10***	20	20	20	20	20
		φ	10	10	10	***1***	***3***	10	10	10
Mixed	Low	τ	0.25	0.25	0.25	0.25	0.25	***0.1***	***0.5***	0.25
		F_{scale}	1	1	1	1	1	1	1	***2.5***
		c_o	20	***1***	***10***	20	20	20	20	20
		φ	10	10	10	***1***	***3***	10	10	10
Mixed	High	τ	0.25	0.25	0.25	0.25	0.25	***0.1***	***0.5***	0.25
		F_{scale}	1	1	1	1	1	1	1	***2.5***
		c_o	20	***1***	***10***	20	20	20	20	20
		φ	10	10	10	***1***	***3***	10	10	10

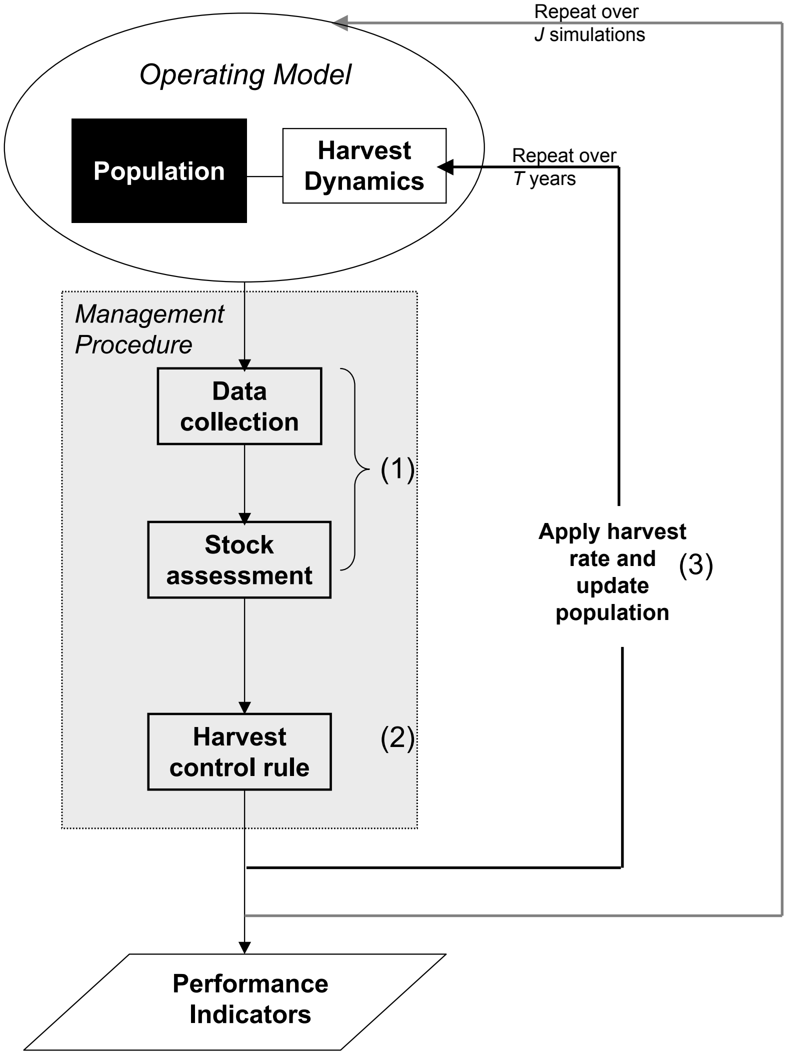

Figure 3.1: Schematic of the simulation model showing the components of the operating model and management procedure. The model separates the annual fisheries management function into three stages: (1) information generation, (2) provision of harvest advice to managers, and (3) application of management advice. Performance indicators measure performance of the simulated management system with respect to the stated fishery objectives.

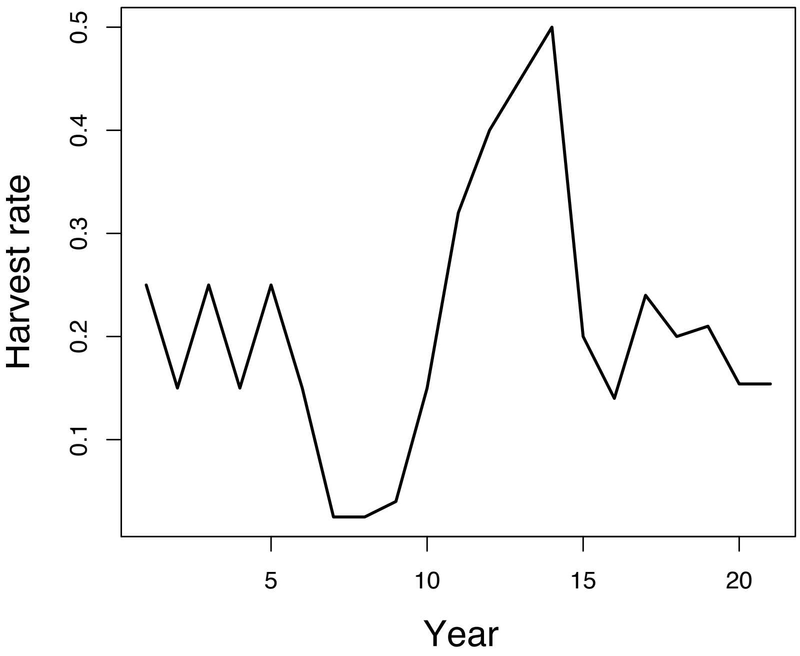

Figure 3.2: The time series of historical average harvest rates that is applied to the population aggregate and is used to generate catch data to initialize the stock assessment component of the management procedure.

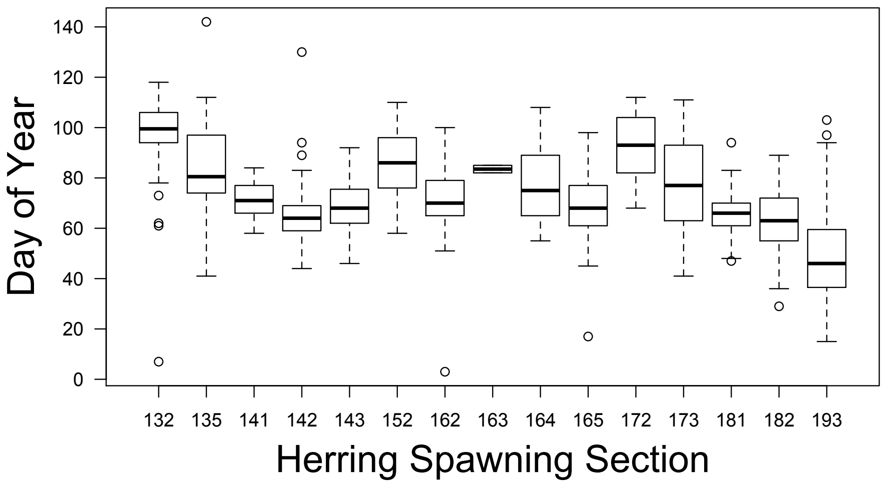

Figure 3.3: Pacific herring spawn timing by section, Strait of Georgia 1928-2006. The boxplots summarize the median, 25th, and 75th percentiles of spawning dates, the whiskers correspond to the 5th and 95th percentiles, and the open circles are outliers. Data provided by Jake Schweigert Pacific Biological Station, Nanaimo B.C.

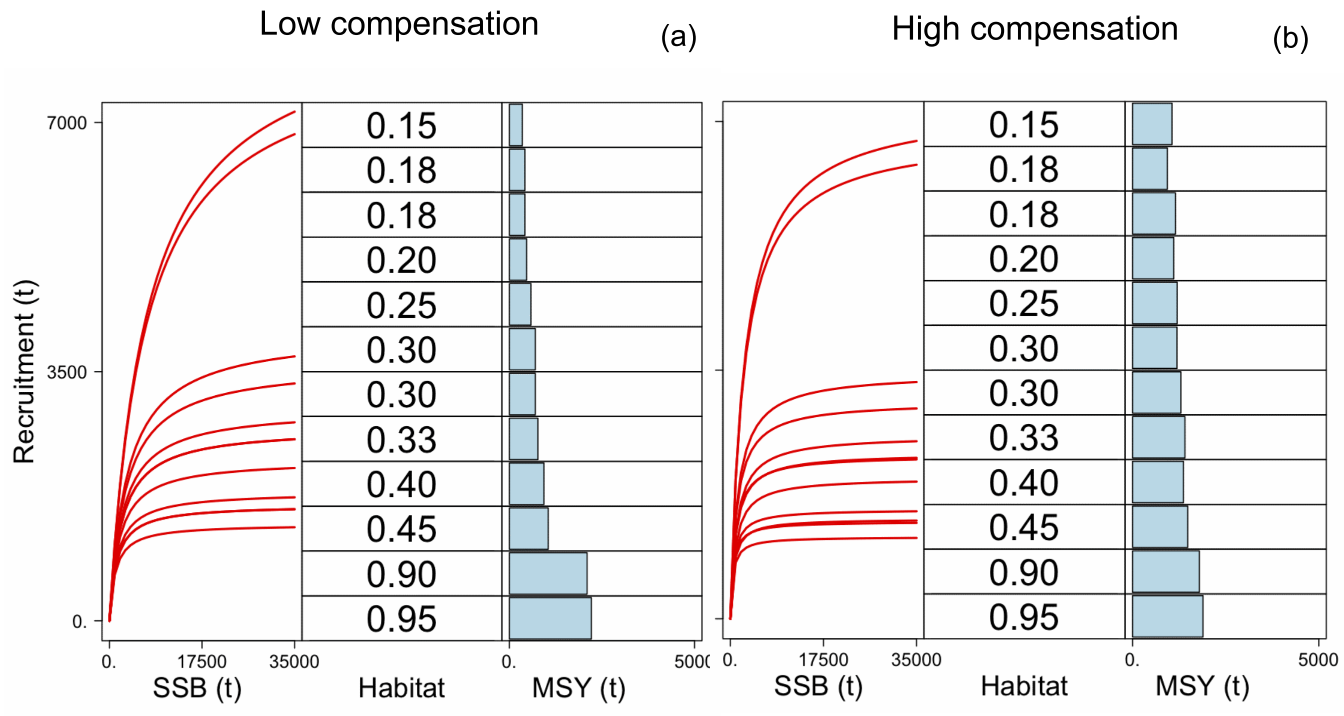

Figure 3.4: Recruitment functions, habitat quality (H_p), and equilibrium yield for the *discrete* population scenarios under low (a) and high (b) recruitment compensation.

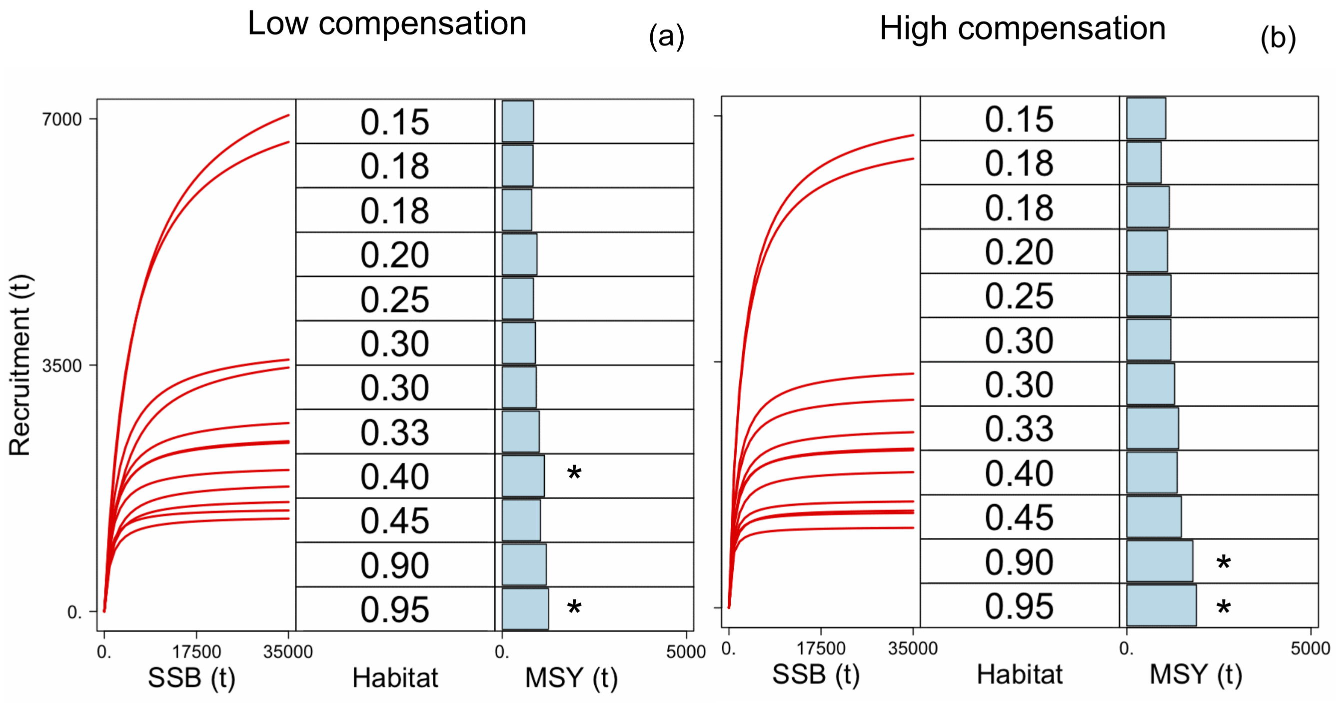

Figure 3.5: Recruitment functions, habitat quality (H_p), and equilibrium yield for the *mixed* population scenarios under low (a) and high (b) recruitment compensation. Asterisks denote sub-populations that receive the highest proportion of migratory biomass.

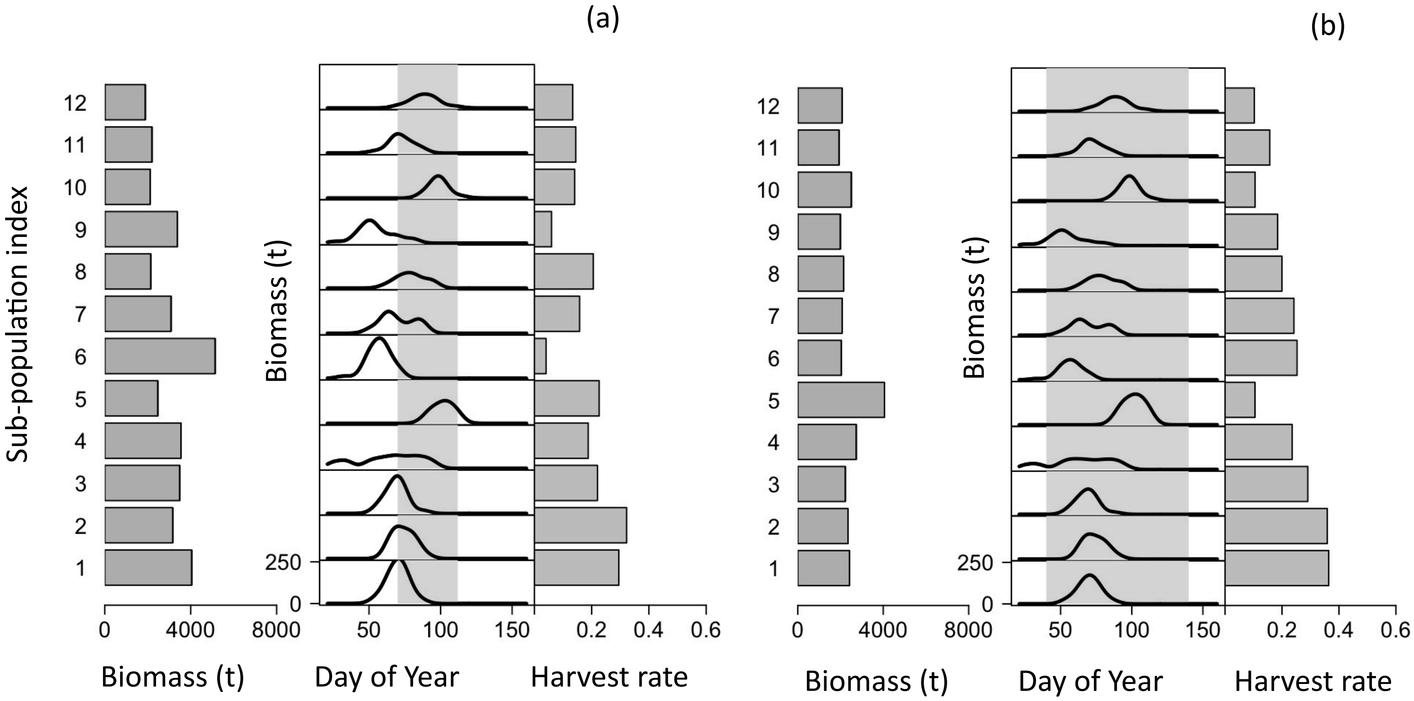

Figure 3.6: Average pre-fishery biomass, daily pre-fishery biomass arrival on the spawning grounds, and realized harvest rate by sub-population for the *discrete* populations scenario for (a) 40-day and (b) 100-day fishing seasons. Results are averaged over years 50-75 for one simulation run using the baseline parameter settings. The grey shaded region on the daily biomass plots indicates the timing of the fishing season. All daily biomass plots within a panel are on the same scale as sub-population 1.

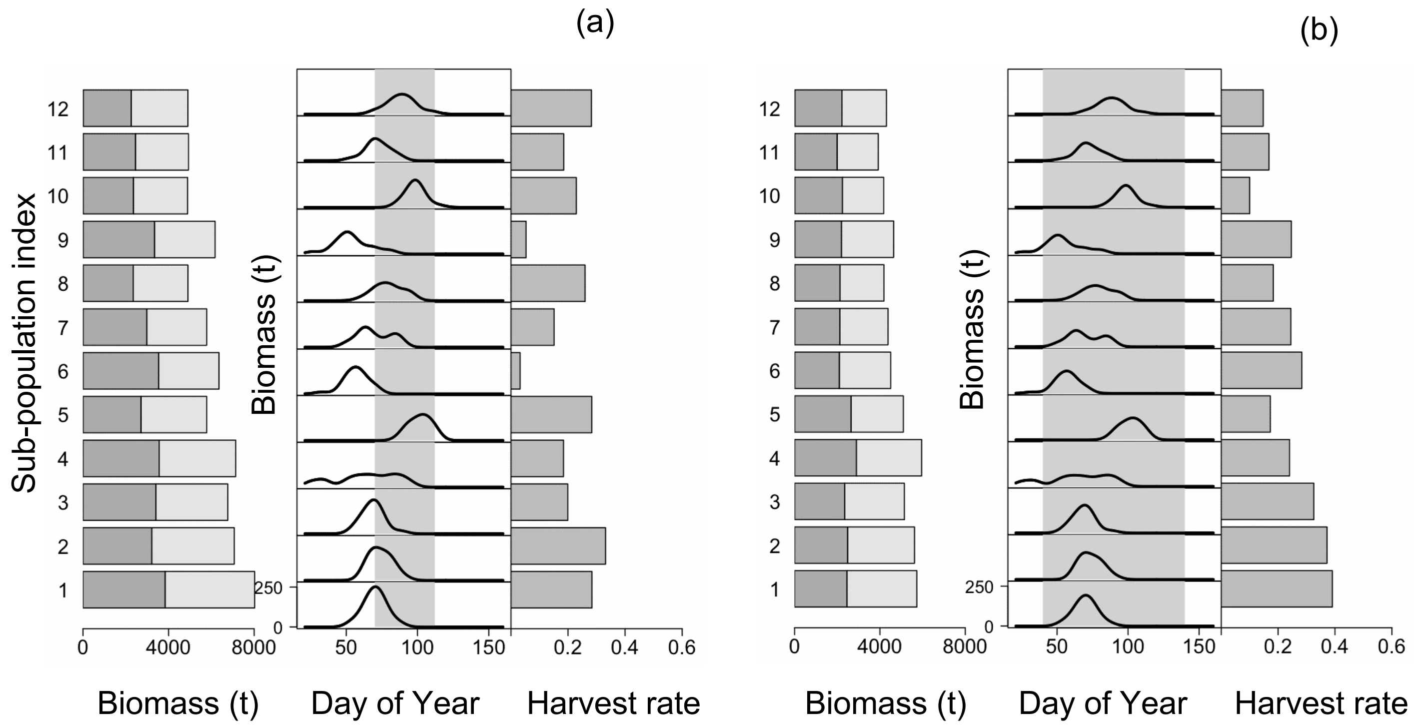

Figure 3.7: Average pre-fishery biomass, daily pre-fishery biomass arrival on the spawning grounds, and realized harvest rate by sub-population for the *mixed* populations scenario for (a) 40-day and (b) 100-day fishing seasons. Results are averaged over years 50-75 for one simulation run using the baseline parameter settings. Light and dark grey bars indicate the resident and migratory biomass, respectively. The grey shaded region on the daily biomass plots indicates the timing of the fishing season. All daily biomass plots are on the same scale as sub-population 1. All daily biomass plots within a panel are on the same scale as sub-population 1.

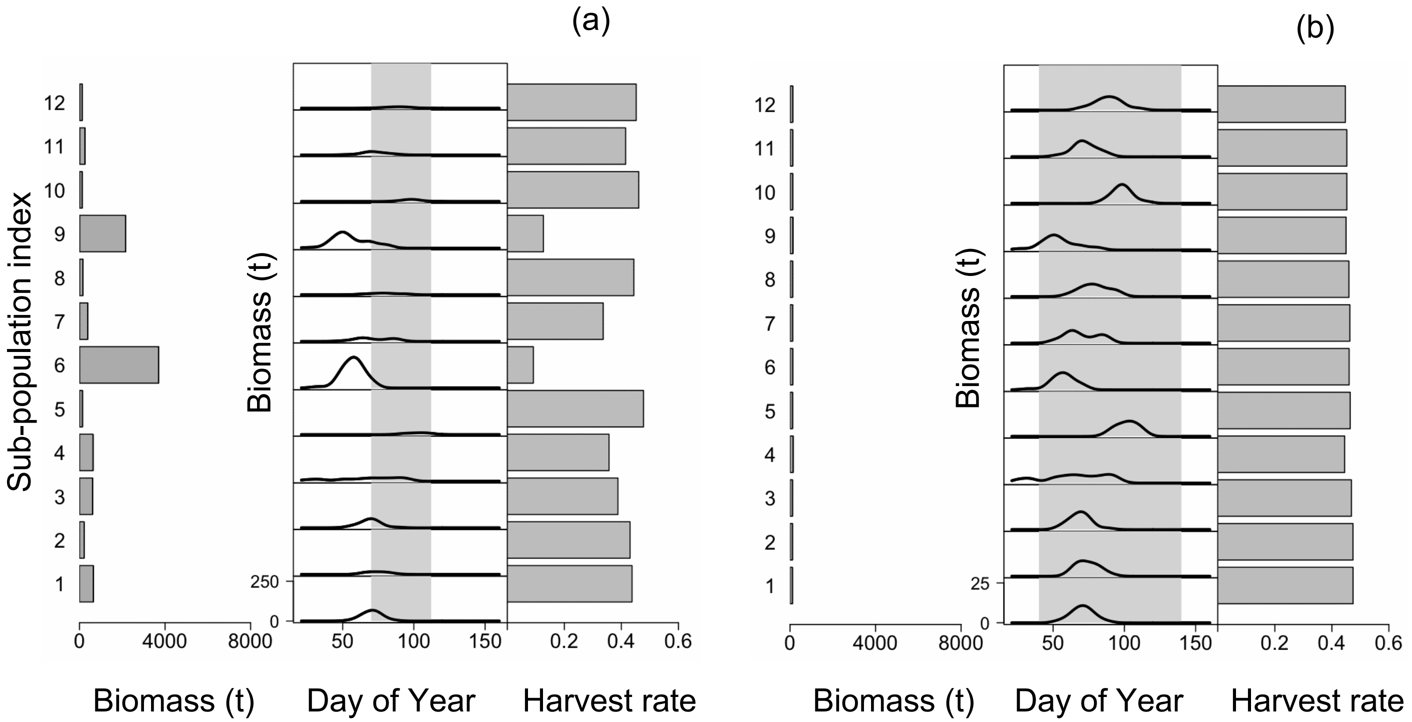

Figure 3.8: Same as Figure 3.6 (*discrete* populations) except using the IFD target catch rate $c_o = 1$ instead of the baseline value of 20 tonnes/boat.

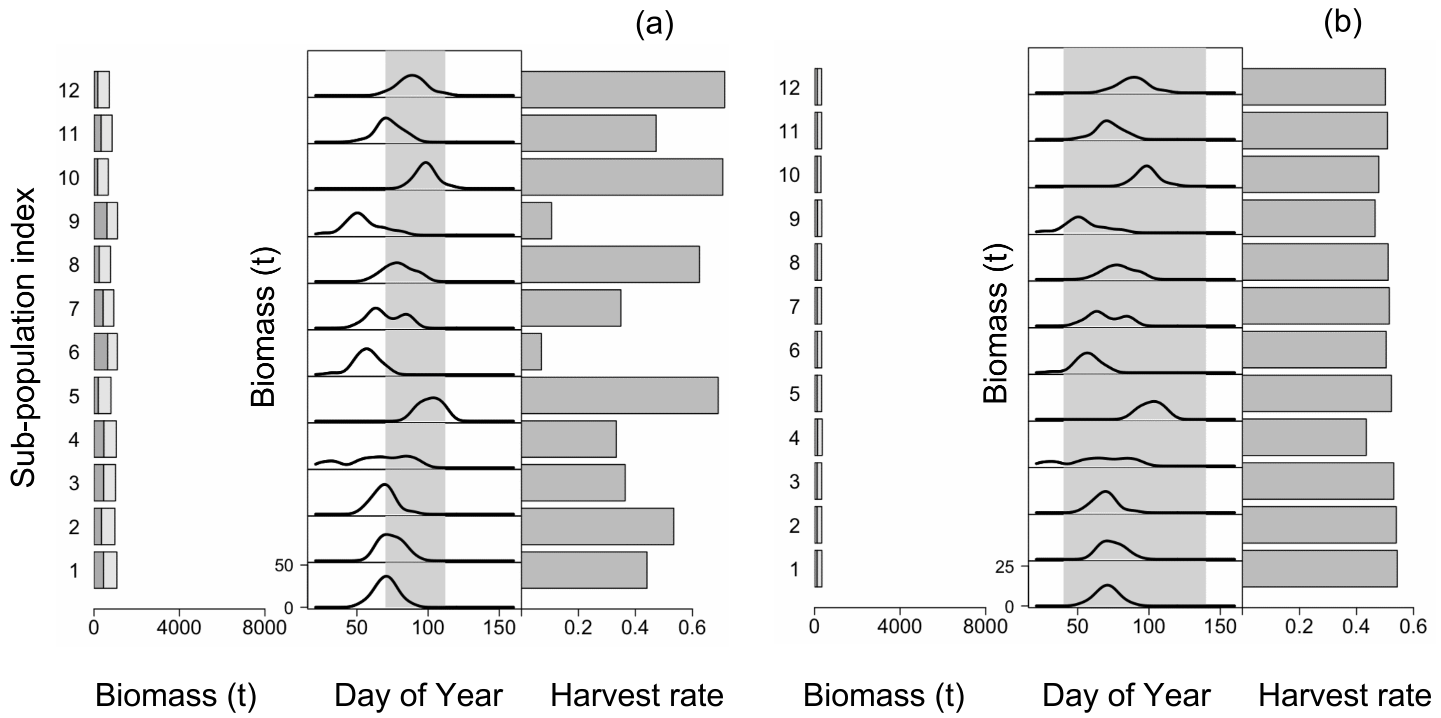

Figure 3.9: Same as Figure 3.7 (*mixed* populations) except using the IFD target catch rate $c_o = 1$ instead of the baseline value of 20 tonnes/boat.

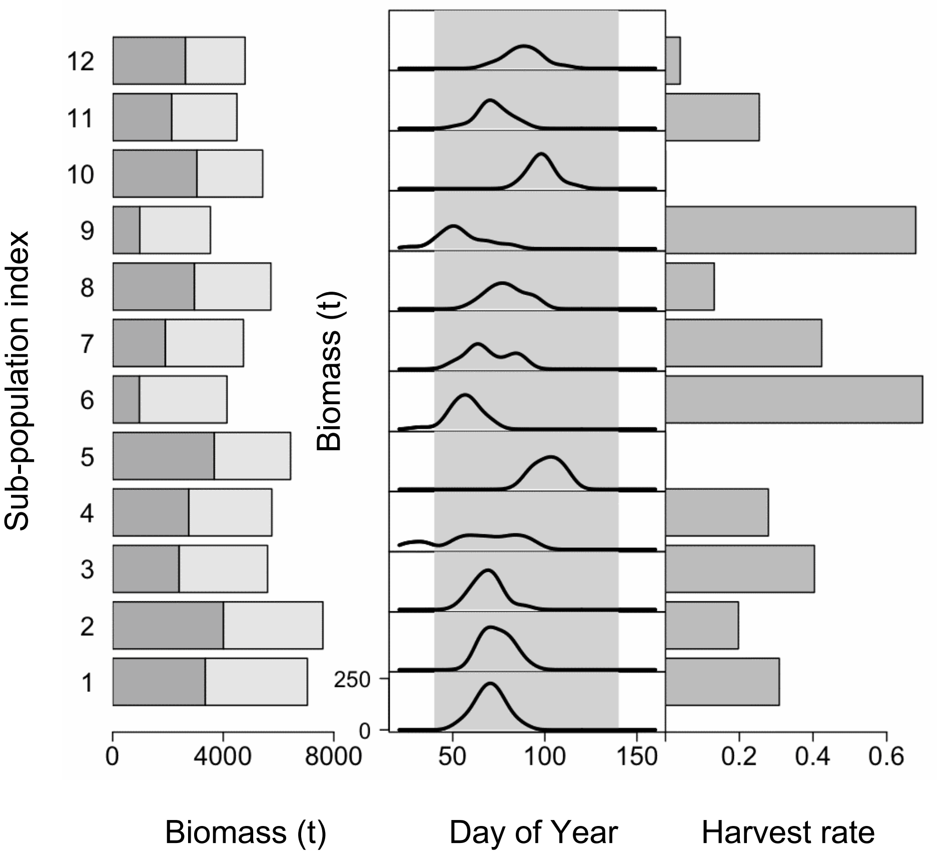

Figure 3.10: Average pre-fishery biomass, daily pre-fishery biomass arrival on the spawning grounds, and realized harvest rate by sub-population for the mixed populations scenario. Results are averaged over years 50-75 for one simulation which was run for a 100-day fishing season and ideal free distribution target catch rate $c_o = 1$. Light and dark grey bars indicate the resident and migratory biomass, respectively. TAC recommendations are based on the true biomass and true B_0 instead of their stock assessment estimates. The grey shaded region on the daily biomass plots indicates the timing of the fishing season. All daily biomass plots are on the same scale as sub-population 1.

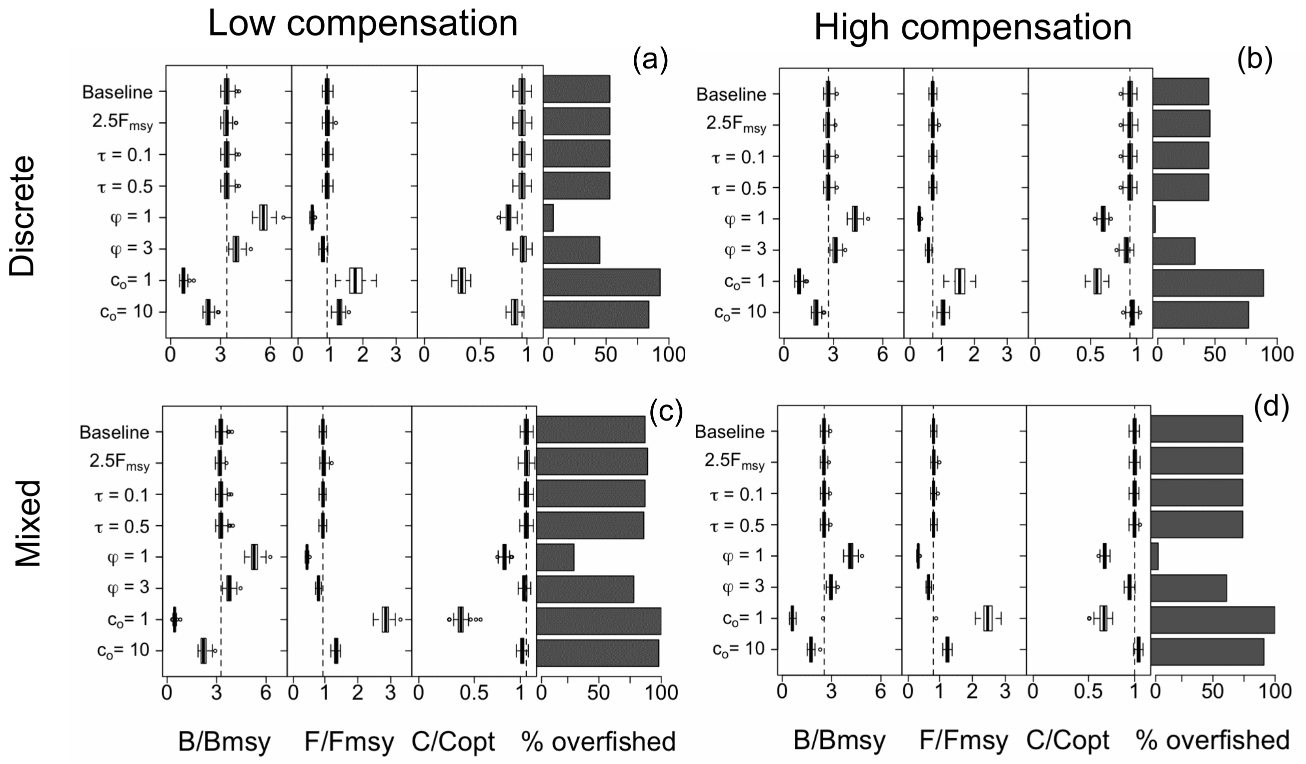

Figure 3.11: Performance indicators for the simulations based on the 40-day fishing season, for scenarios of low- and high-recruitment compensation (left- and right-hand panels, respectively) and discrete and mixed populations (top and bottom rows, respectively). Boxplots and histograms summarize performance indicators across Monte Carlo trials for the aggregate population and sub-populations, respectively. Y-axis labels indicate the parameter that differs from the baseline in each simulation (summarized in Table 3.2). Dashed lines represent the median baseline results.

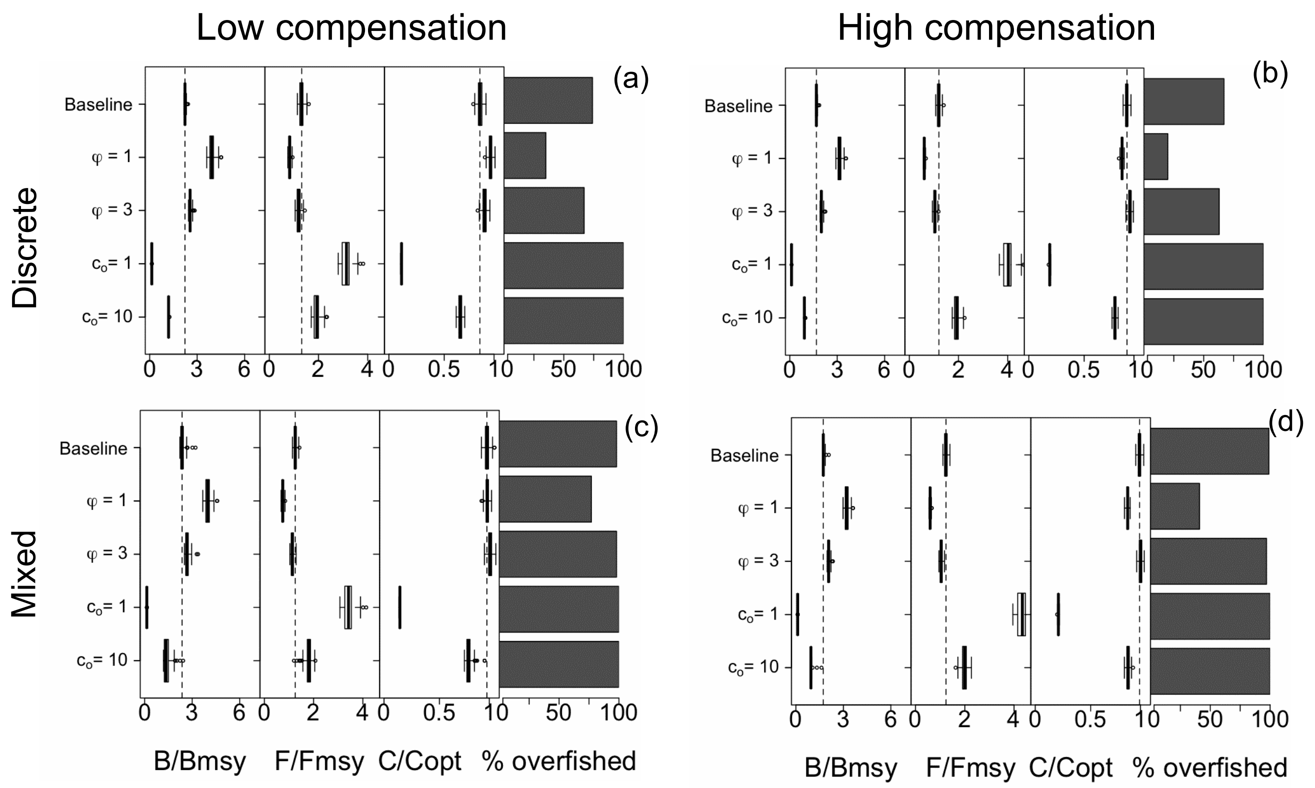

Figure 3.12: Same as Figure 3.11 except for simulations based on the 100-day fishing season scenario. The set of simulations explored for the 100-day fishery excludes MP options ($2.5F_{MSY}$, $\tau = 0.1$, , $\tau = 0.5$) because they did not affect performance in the 40-day fishing season scenario.

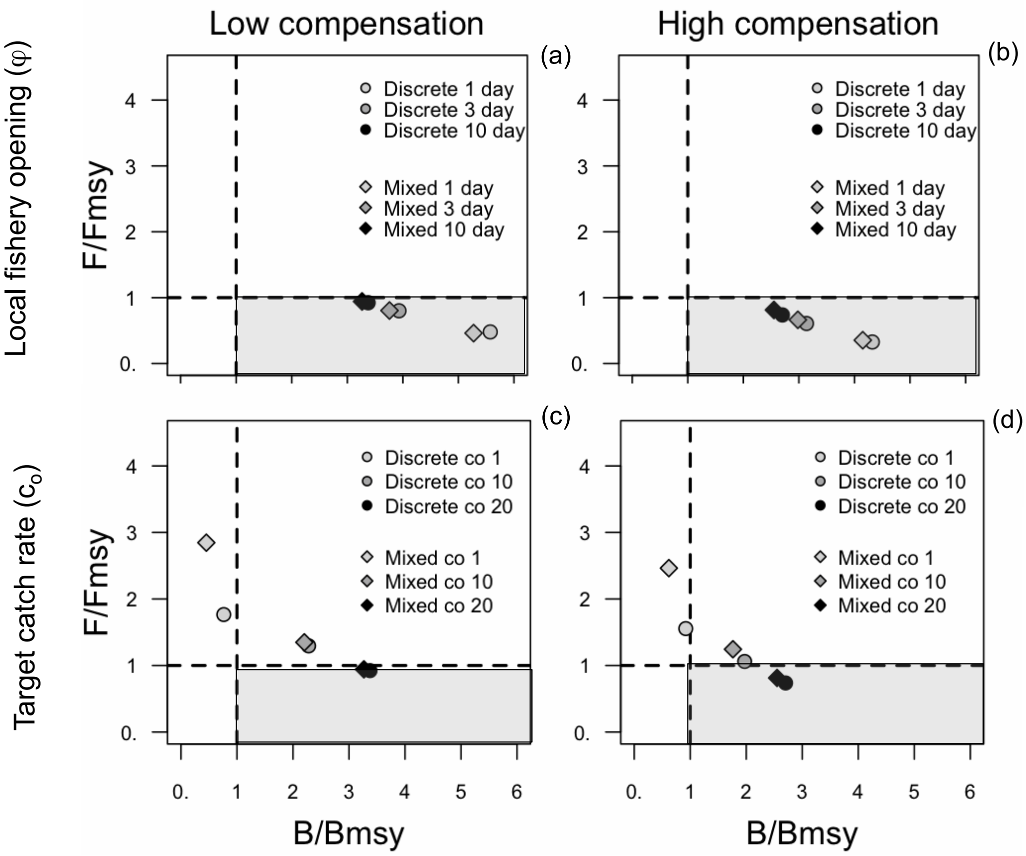

Figure 3.13: Relationship between the median (over simulations) average annual ratio of total B/B_{MSY} and total F/F_{MSY} for the 40-day fishery simulation scenarios. Panels are arranged in columns based low- and high-recruitment compensation (left- and right-hand panels, respectively), and by rows based on the options for alternative harvest dynamics (duration of local fishery opening (φ; 1, 3, 10 days) and IFD target catch rate (c_o; 1, 10, 20 tonnes/boat); top and bottom rows, respectively). Shaded area depicts the most desired region.

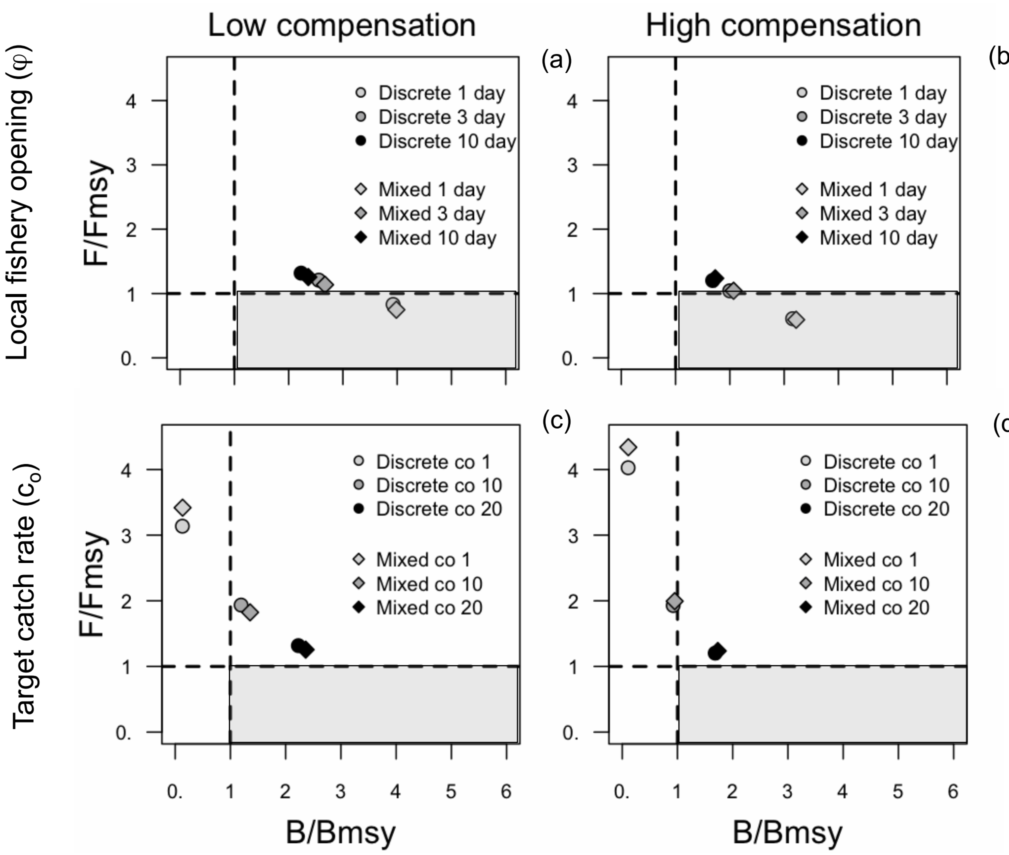

Figure 3.14: Same as Figure 3.13 except for simulations based on the 100-day fishing season scenario.

Chapter 4

The Impacts of National Policy Shifts on the Quality of Fisheries Science in Canada

4.1 Introduction

The traditional roles of science have been to discover, communicate, apply knowledge, and educate the next generation of scientists (Lubchenco 1998). Following from these roles, science has been expected to generate the best possible knowledge and to produce something that is of use to society. These expectations are based on a linear model of science that represents a flow of knowledge from basic research and then to applied research to its application for generating societal benefits (Bush 1945). This model guided decisions that promoted public funding of scientific research and development in the post-war era, and yielded massive innovations in science and engineering. However, society's needs are evolving as human population growth and technological innovation have enabled the exploitation and alteration of the natural environment in unprecedented ways (Lubchenco 1998). Research on topics such as global climate change, biodiversity loss, habitat alteration, and sustainable development has fostered a societal expectation that science will help to mitigate and manage human impacts on the natural world (Lubchenco 1998). Such environmental problems are highly complex, and can rarely be solved by traditional, 'Mode-1' science, which focuses on its own needs and which is concerned primarily with advancing scientific knowledge for its own sake (the so-called "pure" or "basic" research) (Nowotny et al. 2006). Solutions to complex environmental problems may be more likely to emerge from 'Mode-2' science, which produces knowledge in the context of application, is multidisciplinary, and responds explicitly to external needs (Gibbons 2002). The linear model of scientific innovation is insufficient for Mode-2 science because the ultimate goal is subject to multiple objectives, accountabilities, and interests (Bielak et al. 2008). Scientists in Mode-2 research act as knowledge brokers, and engage in a recursive dialogue with multiple users of information, including policy-makers (Bielak et al. 2008, Pielke 2007).

4.1.1 Mode-2 science

Recent changes in science policy in many countries align with the Mode-2 paradigm[1], which has generated significant interest among academics, policy makers, and funding agencies interested in promoting high-profile, trans-disciplinary research that is immediately relevant to society (Wald 2007). Researchers now commonly invoke the language of 'Mode-2', 'post-normal science', and 'triple helix' in reference to the shared benefits of research that is closely aligned with the requirements of policy-makers and industry (Gibbons 2002, Garcia and Charles 2008, Wald 2007). The anticipated benefits of Mode-2 science include enhanced economic growth, international competitiveness, and, for environmental problems, increased acceptance of policies that

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[1] Trend reported in OECD Conference Report on Science Funding in Transition - Changing Paradigms and First Experiences of Implementation. Available on-line at: http://www.OECD.org/dataoecd/23/10/2508930.pdf

promote conservation and sustainable use of natural resources (Beesley 2003, Garcia and Charles 2008). Some critics of Mode-2 science focus on the potential impacts of changes in the funding strategy from being purely publicly-funded to partnerships and privately-funded research. By focusing on 'priority' research questions and promoting a multiplicity of research efforts on essentially the same topic (e.g., isolation of a virus), the latter funding arrangements may lead to an an inefficient allocation of resources between basic and applied research (Dasgupta and David 1994). This inefficiency could ultimately cost society more in terms of lost long-term opportunities for innovation than 'no strings attached' publicly-funded science (Dasgupta and David 1994, Wald 2007).

Issues of funding Mode-2 science extend beyond the long-term impacts on innovation to a more immediate point of whether and how involving private interests in the generation of scientific knowledge affects the autonomy and objectivity of science. The individual experiences of researchers vary widely in this regard. For example, a survey of European academics who perform contract-based research found relatively little evidence of industry sponsors interfering in the analysis, interpretation, or dissemination of research (Gulbrandsen and Smeby 2005). Industry sponsorship has additionally facilitated important research that would otherwise not have been undertaken, including the first study of the effect of trans fatty acids on heart disease risk (Katan 2007). However, other researchers have documented a significant association between industry sponsorship and pro-industry conclusions in the bio-medical, pharmaceutical, food and nutrition, and oil/energy fields, a phenomenon referred to as the 'funding effect' (Krimsky 2006, Lesser et al. 2007, Moses et al. 2005). These results suggest that the source of funding can impact the degree to which scientists adhere to scientific norms (e.g., neutrality, disinterestedness, universalism, communalism, organized skepticism (Merton 1938)), which may degrade the quality and legitimacy of science.

4.1.2 Research objective

The overarching objective of this chapter is to explore the impact of changing funding arrangements on the quality of fisheries science in Canada. Within the context of this chapter, I use 'science' to represent the information generated by government, industry, or academic researchers for purposes of deciding on regulations or management actions. Mode-2 science is not necessarily of a lower quality than Mode-1 (Gibbons 2002), however, the arguments presented above suggest that the attributes of the Mode-2 knowledge production system (e.g., funding arrangements, institutional capacity, social arrangements) may influence the adherence of researchers to the standards of science (used here as a metric of the 'quality' of science). The key characteristics of the Mode-1 and Mode-2 paradigms are summarized in Table 4.1. Under the Mode-1 paradigm, 'basic' science is assumed to represent perfect application of the scientific method in which the principles of autonomy, objectivity, and universalism are consistently achieved, and scientists are free to dictate the direction of their research without consideration of how and when it will impact society. The Mode-2 paradigm represents applied, policy-relevant research, and is therefore directly relevant to fisheries science. In reality, the mode of science is likely to be at some dynamic, intermediate point along a continuum, rather than at either extreme. It is therefore important to understand the processes driving shifts in the mode of science in order to fully understand how these dynamics might impact the quality of research.

The funding effect does not appear to have been previously investigated in fisheries science. However, a study of the state of scientific objectivity in New Zealand deep-water fisheries management found that industry-funded scientists tend to promote the interests of their funders at science and management meetings (Cawthorn 2007). Such advocacy by scientists introduces non-scientific criteria into the scientific process and puts at risk the objectivity of the information used to make fisheries management decisions (Cawthorn 2007). An important feature of New Zealand fisheries management is its reliance on industry funding for management-related activities, including science (Cox 2003). This is referred to as cost-recovery, and it is a central aspect of the private-property rights form of governance used in New Zealand fisheries. This governance model is intended to foster strong management and conservation incentives in the industry, and to promote incentives to collaborate with scientists and managers (Cox 2003). It also aligns the New Zealand fisheries science model almost perfectly with Mode-2.

Science policy is the purposeful, politically-motivated activity of funding and regulating research that is conducted with public money (Edqvist 2003). Changes in this policy reflect different patterns in the perceptions of the purpose of research, the relationship between science and society, the ways of doing research, the funding systems, and the institutional arrangements that govern the above (Edqvist 2003). These collectively determine the mode of science, and current science policies tend to reflect previous policies in one or more of these aspects (Ruivo 1994). The legacy of previous policies can create problems for the organization and management of Mode-2 science, because the associated issues differ substantially from Mode-1 science. Examples include the commodification of knowledge, rights, and patents, accountability of research organizations, and threats to academic freedom (Jacob 2001). Appropriate institutions and governance are therefore particularly important for Mode-2 science.

In this chapter I examine the influence of changes in governance on fisheries science in Canada. I extend the definition of the 'quality' of science to include both the nature of science and the way in which scientific information is used in fisheries policy development and implementation. I begin by developing a set of information standards for fisheries management based on the current literature. My intent is not to conduct a point-by-point evaluation of the Canadian fisheries science system, rather, I aim to highlight the institutional pressures exerted on the science process and to explore the ways in which they may shape the quality of science and management outcomes in Canada. The standards developed and applied in this chapter reflect that intent. In the following sections I provide what others have stated as the theoretical justification for the institutional shift toward Mode-2 science and cost-recovery, and describe ongoing changes in Canadian fisheries governance.

4.2 The History of Fisheries Management in Canada

Prior to 1979 the Fisheries Research Board (FRB) was the principal research agency working on fisheries and aquatic science in Canada (Johnstone 1977). The FRB was an administrative anomaly that was not associated with any government department. This changed in 1979 with the creation of the Department of Fisheries and Oceans (DFO), which marked the full integration of fisheries science into a political, federal government agency (Hutchings et al. 1997). Within this agency the Minister of Fisheries and Oceans (an elected official) holds the authority and responsibility to regulate the conservation and preservation of fisheries resources, including specifying the type of fishing gear, catch limits, duration and timing of fisheries, and the size and species of fish that may be captured (Parsons 2010). In 1995, the Canadian government undertook a deficit reduction program known as "Program Review" that resulted in large cutbacks across all government departments. DFO's budget was cut by 1/3 and its staff was reduced by 40%, which greatly restricted the Department's ability to meet its management obligations and undertake research on the state of fish stocks (Scarth 2004). Similar to Environment Canada and Natural Resources Canada (NRCan), DFO was forced to rely on voluntary management measures after Program Review, and began moving toward self-regulation by the fishing industry (Scarth 2004). The organizational structure at DFO changed in 1997 with the creation of the Oceans Branch, which began to shift the priority and focus of DFO toward generalized 'oceans research' and split the already diminished Departmental funding between 'oceans' and 'fisheries' (Lane and Stephenson 2000). Recognition that Canada is not legally free to devolve complete rights and responsibilities for fisheries science and management to the resource user (the reason is explained in a later section on Canadian fisheries governance), led to the development of protocols for 'alternative service delivery' and increased emphasis on cost-recovery programs, which were implemented in parallel with partnership arrangements between the industry and government that began to move governance toward formal co-management of the fisheries (Lane and Stephenson 2000).

Science is a central element of the fisheries management process in Canada, and as such, it is an important vehicle for promoting the public good. However, its applied nature means that fisheries science also has elements of a private good (Schrank et al. 2003). These competing pressures have not been adequately articulated in Canadian fisheries science policy and this oversight has created tension between the fishing industry and DFO, which relies on industry funding to offset budget limitations and to support its core science and management activities. Prior to 2006, many industry-DFO partnerships were based on the sale of fish to pay for science. This activity was subsequently ruled illegal by the Federal Court of Appeal which held that the Minister of Fisheries and Oceans cannot fund its science programs by the sale of publicly-owned resources (the fisheries). That ruling has come to be known as the Larocque Decision, and is based on a technical interpretation of the Financial Administration Act[2]. However, the events leading to the Larocque Decision reflect a failure of DFO to articulate the elements of fisheries science that serve the public versus the private interest[3]. As I show below, failure to do so may have interfered with the ability of scientists to develop research programs that successfully navigate across the spectrum of public and private science, and as the reduced funding for surveys and test fishing

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[2] Decision of the Canadian Federal Court of Appeal, 2006 (Larocque v. Canada (Minister of Fisheries and Oceans), 2006 FCA 237).

[3] Public and private interests in fisheries science are described further in a later section.

suggests[4], this oversight may ultimately limit the ability of DFO to provide the 'best' information for management.

4.3 Information Standards for Fisheries Management

Scientific information is commonly viewed as an input to a properly functioning fisheries management system (de la Mare 1998). However, the interaction between the system of institutions and the system of scientific researchers and their funders (which together constitute the 'management system') involves three interacting processes that contribute to the relevance of knowledge and dictate its influence in the policy arena: *generation*, *transmission*, and *use* of knowledge (Ascher et al. 2010). Canada relies on scientific advice for decision making and analysis. As a result, the Federal Government outlined the SAGE (Scientific Advice for Government Effectiveness) Principles and Guidelines which provide direction for generation, transmission, and use of scientific information (Industry Canada 1999). The six core principles of SAGE are:

1. Early issue identification
2. Inclusiveness
3. Sound science and sound advice
4. Uncertainty and risk
5. Transparency and openness
6. Peer review

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[4] DFO Interim Policy Guidance on the continuation of Collaborative Agreements that previously relied on the use of fish. Online access: http://www.dfo-mpo.gc.ca/science/newpoli-polinouv/guidance-conseils-eng.htm

The SAGE Principles focus less on the standards of science for knowledge generation, and more on clarifying the ways in which science is communicated and used to inform policymakers. This is evident in the emphasis on the processes of communication, stakeholder engagement, and accountability of decision makers. Principle 3 is intended to maintain the quality and objectivity of science by requiring science advice that is 'unfiltered by policy considerations', promoting peer review, maintaining a high level of scientific training, and developing close ties between science and policy (Industry Canada 1999). Within this framework, scientific information is important because it can influence policy and society at large (i.e., Mode-2). However, the Principles emphasize that science is only one input to government decision making, and it is often weighed against cultural, ethical, and other considerations.

The advisory process for Canadian fisheries management aligns with the SAGE Principles, but place a stronger emphasis on inclusiveness, transparency and openness, and uncertainty[5]. These principles and guidelines are mainly aspirational but provide some insight into the intent of managing the science-management interface in Canada. However, they could benefit from closer consideration of phrases such as "sound science" and "sound advice". I elaborate on these concepts below.

Criteria for evaluation of fisheries science acknowledge its use in policy development and implementation of management decisions, and seek to maintain its objectivity and validity in order to ensure that policy decisions are based upon the 'best available' science (FAO 1995, Sullivan et al. 2006). These standards are based on the norms dictated by the scientific method, and typically include the elements outlined in Table 4.2. They describe the scientific principles applied in the knowledge-generation process. An emerging class of standard for the generation of fisheries information is the scope of research. Specifically, there is consensus that the scope of traditional, single-species fisheries research is too narrow and must be broadened in order to inform

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[5] DFO Science Advisory Process Framework. (SAPF) Online access: http://www.dfo-mpo.gc.ca/csas-sccs/process-processus/advice-avis-eng.htm#goals

an integrated, ecosystem-based approach to fisheries management (EBFM) (Botsford et al. 1997, FAO 1995, Pikitch et al. 2004). However, implementation of EBFM is constrained by uncertainty about the information that would be required to manage fisheries in an ecosystem context. The standards for EBFM are arguably higher than for single-species assessments because they require scientists to present the trade-offs among competing objectives based on their understanding of interactions between the components of the fishery system (Link et al. 2002). This understanding requires the information obtained by current resource surveys, augmented by the monitoring of environmental variables, habitat quality, non-target species, and predator-prey interactions(Link et al. 2002). These latter types of data are not consistently collected in fisheries surveys.

Fishery scientists tend to be well informed about the rationale and criteria underpinning the generation of good scientific information. In contrast, knowledge transmission and use receive relatively less attention by scientists, but these processes dominate the interests of society, policymakers, and politicians (Ascher et al. 2010). Fishery scientists can influence policy outcomes by engaging in the transmission and use functions, but the appropriateness of scientists engaging in the policy process to promote their preferred outcomes has been called into question (e.g. Lackey 2007, Rosenberg 2007). Others are unconcerned about the general phenomenon of scientists engaging with the political process; they argue that scientists are part of society, and have both a democratic right and an interest in doing so. Problems arise when political arguments are framed as, and equated with, scientific arguments (Pielke 2007). In cases where scientists do engage in the policy process, the onus is on them to be aware of the role they are playing, and to understand the limits of science in relation to policy and politics. To this end, Pielke (2007) outlined four idealized ways in which scientists can advise on policy:

1. Pure scientist - undertakes research without concern for its relevance, and has no direct contact with decision-makers,

2. Science arbiter - seeks to stay removed from policy and politics but provides answers to specific questions from decision-makers,
3. Issue advocate - takes a clear position and uses scientific knowledge and influence to argue for a specific policy action,
4. Honest broker - engages directly in the policy process and seeks to integrate scientific knowledge and stakeholder concerns to clarify alternative courses of action.

Pielke (2007) shows a clear preference for scientists acting as honest brokers of scientific information, and warns against scientists knowingly (or unknowingly) acting as 'stealth issue advocates' - claiming to focus only on science and staying above the political fray - while working to restrict the scope of choice available to decision-makers. Without appropriate oversight, science arbiters can easily fall into this role (Pielke 2004). Opinions on the appropriate role of fisheries scientists in policy tend to downplay the option of pure scientist and promote the science arbiter and honest broker (e.g., Lackey 2007). However, issue advocates are also recognized as profoundly influential in marine fisheries policy, in spite of a pervasive assumption that 'advocacy' research is of lesser quality than 'value-free' research (Rosenberg 2007).

Engaging in the policy arena requires that scientists carefully consider how best to communicate highly technical information (Peterman 2004). Fisheries scientists should also recognize that questions asked by managers are often framed in more simplistic ways than is appropriate, and that providing answers to such questions can be akin to manufacturing certainty where none exists (Kraak et al. 2010). The peer review element of the knowledge generation process is intended to address misleading or misguided science by ensuring that published research meets high standards for objectivity, data collection and analysis, and interpretation of results. However, Kraak et al. (2010) found a discrepancy between fishery scientists' statements about uncertainty in peer-reviewed publications and advisory reports on the same topic, which suggests that they use different standards for communication with different audiences. In addition, there is an obligation on the part of both the scientist and the decision maker to understand the limits of scientific knowledge in decision making. For example, extensions to the linear model of innovation imply that more science will reduce uncertainty, simplify policy decisions, and foster acceptance of the decisions among stakeholders (Sarewitz 2004). Experience suggests otherwise. The move toward inclusive, interdisciplinary science adds dimensions of uncertainty to the understanding of natural systems; the reduction of uncertainty about one process does not necessarily translate to others, and may reduce the coherence among competing scientific disciplines (Sarewitz 2004).

The standards of information for fisheries management outlined in Table 4.2 address the key processes of generation, transmission, and use of knowledge. The institutions required to promote an information system of this sort receive relatively little attention in the literature. However, indicators of successful governance of science for EBFM have been proposed. They include: (i) collection of long time series of data, (ii) conducting assessments and providing advice that is relevant to managers, and (iii) strategic research to improve (i) and (ii) (Sissenwine and Mace 2003). The authors note that institutions must be carefully structured to ensure appropriate levels of funding for all three complementary forms of knowledge. Specifically, they suggest implementing 'firewalls' between sources of funding so that pressure to invest in one activity (usually management-relevant stock assessments) does not jeopardize the other two.

The need to protect resources for 'other' science from being subsumed into the Mode-2 ('management-oriented') science is interesting in the context of fisheries management information because it implies that trade-offs are being made in the generation process that pit the resources for stock assessment against those for ecosystem-based management. Furthermore, when discussing the American approach, Sissenwine and Mace (2003) indicate that the trade-off tends to favor 'relevant' (i.e., stock assessment) information over information that may not be immediately useful for operational fisheries management, but which is widely acknowledged to be fundamental for adopting an ecosystem-based approach, namely, the maintenance of long-term data sets and a strategic approach to problem identification (Sissenwine and Mace 2003). A somewhat different scenario holds in Canada, where government budget allocations are weighted toward generalized 'oceans' research over fisheries science, and industry is expected to contribute the resources that will ensure the provision of management-relevant information (Lane and Stephenson 2000). The justification for this arrangement has been developed by fisheries economists seeking the appropriate allocation of private and public resources in fisheries management. This rationale is presented in the following section.

4.4 The Economic Justification for Cost-Recovery

"[Fisheries science] is a type of marine research that is specifically intended to benefit the fishing industry by providing estimates of current stock abundance and making it easier to evaluate the consequences of today's catches on the future availability of fish. This type of research is of limited utility to the public at large."

Arnason et al. (2003)

Fisheries economists have recently begun to consider the costs of fisheries management (including fisheries science) in their evaluations of present value of the net benefits flowing from the fishery. If the cost of management is negligible, omitting it from cost-benefit analysis will have no effect on the estimation of net benefits. However, if the cost is substantial relative to the value of the landings, then the omission will make the fishery appear to be more profitable than it is, and will mis-inform policies intended to maximize the economic benefit (Arnason et al. 2003). Recent work on the subject indicates that the costs of fisheries management can be significant, totaling \$2.24 billion in the OECD countries in 1997, and accounting for 36% of all government transfer payments to the fishery sector (Wallis and Flaaten 2003). The fact that these costs are predominantly borne by governments raises questions related to who should pay and why, whether the expenditures are too high and are being put to their best use, and whether taxpayers are receiving a suitable return on their investment (Andersen and Sutinen 2003, Arnason et al. 2003).

4.4.1 Government failure in fisheries management

Services included in 'fisheries management' include research, formulation and implementation of policies and rules, and enforcement in accordance with the law (Arnason et al. 2000). The funding for these services usually comes from general tax revenues (Wallis and Flaaten 2003). An important feature of fisheries management services is that they all have strong public good characteristics, that is, they are non-excludable (it is very difficult to exclude anyone from using the good) and non-rivalrous (one's use of the good does not diminish its availability to others). Pure public goods (or services) are relatively rare, but in general, the institutions, processes, and outputs of management systems appear to come close. However, fisheries management systems are more aptly characterized as club goods, which are a subset of public goods in that they are non-rivalrous but excludable, because those not belonging to the 'club' can be barred from their use (Varian 1992). Although it is relatively difficult to exclude anyone from benefiting from fisheries management, it is clear that the industry is the primary beneficiary, and it is difficult to imagine those not involved in the fishery making use of the services. In this sense the 'club' is restricted to those participating in a given fishery, and who stand to gain from the increased efficiency and rent generation that follows from management (Andersen and Sutinen 2003). However, the market benefits of fisheries management can extend beyond the industry to include consumers who benefit from a more stable supply of product and lower prices that can create consumers' surplus (Wallis and Flaaten 2003).

The non-exclusive nature of public goods means that it is difficult to collect payment for those goods, and public goods are therefore not adequately provided by the market. As a result, the provision of public goods generally falls to governments. However, just as market failures exist in the provision of public goods, non-market (government) failures also exist (Wolf 1993). The latter occur when there is a divergence between the decision makers' benefits and costs and society's benefits and costs that results in economic inefficiencies. This divergence derives from the fact that there are few incentives for government decision makers to allocate public funds efficiently (Andersen and Sutinen 2003). Moreover, individuals within a government bureaucracy stand to gain from increasing their budgets, which creates incentives to increase spending on unnecessary items (Wallis and Flaaten 2003). In addition to inappropriate incentives within the government structure, the collective action process can be unbalanced by the fact that the groups receiving government services are different from those paying for the services. This decoupling of benefits and costs leads to an inflated, inefficient demand for public sector services (Wolf 1993). Public choice theory dictates that the public sector works efficiently when voters pay in proportion to the services they receive. This occurs when the benefits and costs are concentrated in the same groups. However, government failures exist when the benefits are concentrated and the costs are diffuse, or when the benefits are diffuse and the costs are concentrated. Pure public goods exist where both costs and benefits are widely distributed (Andersen and Sutinen 2003). Asymmetry in the distribution of costs and benefits is problematic because it can stimulate rent seeking, which is a common cause of government failure. The incentives for rent seeking are particularly strong when the costs of public programs are not borne by the beneficiaries (Andersen and Sutinen 2003). For example, fishermen represent a relatively small portion of the tax base that receives services funded by the majority of taxpayers. Because the marginal cost of providing fisheries management services is virtually negligible for most taxpayers, there is unlikely to be resistance to increasing management services. However, the fishery special interest lobby is likely to dispute any reduction in those services, which creates incentives for governments to oversupply fisheries management (Schrank et al. 2003).

The discussion thus far indicates that public goods are not usually adequately supplied by the market system, however, it does not necessarily stand to reason that governments do a better job of providing them. According to many economists, the problem of inappropriate incentives within bureaucracies, combined with the decoupling of the costs and benefits of providing public services, renders governments poor candidates for providing fisheries services (Arnason 2003). Society gains little by replacing a market failure with a government failure. The next section outlines the economic solutions to this problem, which include cost-recovery.

4.4.2 Correcting government failure

Government failures occur because of inefficiencies in the allocation of goods or services that would not exist in the absence of government intervention. According to economists, supply-side inefficiencies arise because public employees do not have an incentive to provide fisheries management at the least cost; demand-side inefficiencies occur because demand for management services is inflated by lobbying by the fishing industry, and because of decoupling between the costs and benefits of management (Andersen and Sutinen 2003). In order to remedy the supply-side problem, the incentives within the bureaucratic structure must change. However, it is unlikely that supply-side inefficiencies will be eliminated under the government-based arrangement (Andersen and Sutinen 2003). Alternate methods of supplying fisheries services have the potential to improve the efficiency, but this depends greatly on the extent to which the consumers are willing to pay the cost. If the recipients of the services also paid for those services, it would have the added benefit of addressing the demand side failure by bringing the benefits received better into line with what the users pay (Wallis and

Flaaten 2003).

Alternatives to government provision of fisheries services are listed in Table 4.3 (Arnason 2003). The centralized arrangement (1) was discussed above. cost-recovery (2) is a direct result of the user-pay concept in the provision of government services (Cox 2003). This arrangement makes significant progress toward eliminating the asymmetric distribution of benefits and costs and aligns both the demand-side and supply-side incentives in an economically efficient manner (Arnason 2003). Additionally, because the industry is funding management, there is a strong push for cost effectiveness and efficiency in the provision of the service. However, the interest of the industry in the value of the service is unlikely to entirely eliminate inappropriate incentives within government. If anything, the fact that a government fisheries department would automatically receive funding from industry without having to compete within the larger bureaucracy might exacerbate the incentive problem (Arnason 2003). The third arrangement, contracting out, is essentially the opposite of cost-recovery, and eliminates supply-side incentives, but does nothing to address the asymmetry in costs and benefits. Self-management (4) goes a long way toward eliminating both the government incentive problem and the asymmetry problem. However, under this arrangement, management services are a club good to industry members. Consequently, the incentive problems are likely to resurface (Arnason 2003). Additionally, the present governance structure in most countries does not provide an institutional framework that would facilitate self-management of fisheries resources.

The subject of cost-recovery has received increased attention in recent years, and is a key feature in the ITQ systems in New Zealand, Australia, Iceland, Canada, and the United States (Cox 2003). Many of the fisheries managed under ITQs generate substantial economic rents, and the need for subsidies in any form (including management) has been called into question (Arnason et al. 2000). In some cases, cost-recovery programs have proved to be more palatable to fishermen than methods designed to capture resource rents that are based on the notion of a ‘return to the nation'. For example, when the Quota Management System was introduced to New Zealand in 1986, ITQ holders were required to pay annual resource rentals on their quota, regardless of whether the quota was actually caught (McClurg 2000). The intent was for the system to fund itself, however there was no link between the costs incurred by the government and the level of rent collected. In addition, the rents were directed toward general revenues rather than a fishery-specific fund. The formula for calculating the resource rent was based on a complicated trade-off between fishery profitability and the 'super-profits' accruing to the quota holders. The rent capture program was designed to recouping the industry profits gained by moving to an ITQ system, but because it drove the value of the ITQs to low levels, it effectively eliminated the efficiency and conservation incentives that ITQ systems are intended to promote (McClurg 2000). The New Zealand industry strongly supported a change to cost-recovery on the basis that it would lead to more efficient specification and delivery of fisheries management. Many economists now believe that shifting the cost of management to industry, combined with the freedom to utilize innovative harvesting and marketing techniques, creates incentives and means with which the industry can maximize the economic potential of the ITQ system (Cox 2003, McClurg 2000, Schrank et al. 2003).

The Canadian approach to funding fisheries management does not fall into a single category, but involves aspects of arrangements 1-4 (Table 4.3). The arrangements vary on a fishery-by-fishery basis, depending on the fishery's ability to pay, the extent to which co-management arrangements have been implemented, and the availability of in-house expertise at DFO. For example, staffing limitations at DFO mean that consultants now conduct a large number of the stock assessments on the Pacific coast. Some are paid directly by industry, while others are hired by DFO. The following section provides insight into how policy and funding changes at DFO influence this process.

4.5 Fisheries Governance in Canada

4.5.1 Managing fisheries for the public good

The federal Fisheries Act gives the Minister of Fisheries and Oceans exclusive authority over allocation and conservation decisions for fisheries resources that are held in trust for the public (Loucks et al. 2003). The Public Trust doctrine historically protected navigation, commerce and fishing. It has laid dormant in Canada since the nineteenth century, but recent judicial decisions reaffirm that the doctrine forms part of Canadian common law (Smallwood 1993). The "public trust" defies a simple and straightforward explanation, however, it is best considered as a fiduciary responsibility in which the state has a duty to protect public resources, including fisheries (Smallwood 1993). Furthermore, this responsibility is an active one, requiring that the state must act to *promote* as well as to *preserve* the resource held in trust for the public (Smallwood 1993). The doctrine has also been repeatedly confirmed and extended in American law since the 1970s, and is believed to be an important emerging legal concept in Canada, emphasizing that public rights to natural resources are special and deserving of careful management and protection (Smallwood 1993).

Government policy is influenced by the prevailing ideology of the governing party as well as existing laws and other institutions. The definition of 'public interest' can be greatly influenced by the world view of the government, and different governments can have vastly divergent perspectives on their role in promoting the public interest (Loucks et al. 2003). Fisheries management objectives tend to fall into three world views: economic efficiency, resource conservation, and social/community objectives (Charles 2001). The pattern of federal governance decisions over the past two decades suggests that the governments in power in Canada during this period have assumed that the public interest is best served if government promotes economic and conservation objectives, and that the best way to achieve these objectives is by trading access to fishery resources in favor of reduced government spending on science, monitoring, and enforcement (Loucks et al. 2003). As discussed in the subsequent sections, these policy changes are intended to shift power to the fishing industry, and to limit the involvement of government in fisheries science and management. A partial transfer of fisheries management costs to the industry has featured prominently in these policy shifts, and has been largely successful at reducing economic inefficiencies in Canadian fisheries that are managed using market-based tools (primarily individual transferable quotas (ITQs)) (Grafton et al. 2000). However, the impact of these governance changes on fisheries science has been largely overlooked in evaluations of changes in the Canadian fisheries system. In the following sections I will outline the evolving approach to fisheries governance in Canada, and describe the institutional pressures on science and scientists that are emerging in Canada from the shift in fisheries policy toward economic efficiency.

4.5.2 Co-management and cost-recovery

Modern patterns of governance in most sectors of the economy are characterized by devolution of power, deregulation, and privatization, as the state seeks to minimize its management role (Symes 1997). Fisheries are increasingly following suit, seeking an alternative to the expensive and often ineffectual command-and-control management that typifies fisheries worldwide (Christy 2000, Scott 2000). The main institutional options - bureaucracy-based, market-based and community-based regulation - differ in the form of rights granted to individuals in the system (Yandle and Dewees 2003). Property rights have been proposed as the solution to the threats to sustainability, efficient use, and resource allocation arising from the well-known 'tragedy of the commons' problem that plagues natural resource systems (Hardin 1968). In fisheries, the evolution of property rights has advanced in two directions: private property rights (mainly ITQs) and co-management (Charles 2000). Both refer to a partial transfer of responsibility and authority for certain limited decisions from government agencies to networks of public and private bodies (Rhodes 1996), the difference between the two is found in the emphasis placed on the right to use versus the right to manage (i.e., regulate use patterns), and the varying degree to which state involvement is tolerated (Honneland 1999).

Co-management emerged in Canadian fisheries in the early 1990s in response to community and industry interest in greater involvement and transparency in management (Wilson 2008). It is an innovative ‘bottom-up’ approach to governance that recognizes the importance of participation by both the state and stakeholder groups in management, and in which a high level of state involvement is tolerated (Jentoft 1989, Pinkerton 1989). In contrast to the private-property approach that emphasizes the generation of wealth and economic efficiency, co-management arrangements emphasize the social benefits of collective action (such as persistent and successful management institutions) which are believed to outweigh the associated problems and costs (such as the potential for free-riding) (Jentoft et al. 1998). While the principles of co-management are increasingly drawing the attention of budget-constrained governments worldwide, the applicability of co-management to large resources in developed nations has been called into question (e.g., Symes 1997). Potential problems include differing world views of user groups hindering consensus building, questions regarding the commitment of stakeholder organizations to regulating their members, the resilience of the co-management process to strong vested interests of industry, the lack of free access to state-owned resources such as data, and the high potential for the government to be ’captured’ by the most powerful stakeholder group (Pinkerton 1999, Eythorsson 2003). Co-management was initially anticipated to result in more equitable sharing of both the resource benefits and the burdens (costs) of management between the Canadian government and industry, First Nations, and community stakeholders. However, these social and community objectives have not been achieved because government policies tend to be strongly biased toward economic and conservation objectives for fisheries management. As a result, co-management arrangements in Canada tend to favor large-scale operators that can afford to pay for science and monitoring, an approach referred to as “corporate co-management” (Loucks et al. 2003).

Co-management requires the government to recover the 'private' costs of management from the resource user, however the separation of private and public costs and benefits is often unclear (Schrank et al. 2003). Australia and New Zealand stand apart from most other nations in the extent to which they have implemented policies for devolution, cost-recovery, and co-management. Both have enacted legislation to create efficiencies in the provision of fisheries management services, and to create incentives within the industry to take greater responsibility for fisheries management (Cox 2003, Stokes et al. 2006). cost-recovery is a critical aspect of both countries' fisheries management systems. Australia's Commonwealth fisheries are managed on a full cost-recovery basis. This means that the commercial fishing industry pays 100% of the costs attributable to its management, and the government pays for management activities that benefit the broader community (including industry) (Cox 2003). Australia uses a two-stage process to determine which management costs are attributable to the industry and which are borne by the government. In stage one, if the answer to the following question is 'yes', costs are assigned to the industry: Would the non-existence of this industry group eliminate the need for the management activity in question? An affirmative answer moves to stage 2, which addresses the question of cost-recovery. A determination of whether to recover the assigned costs is based on the extent of benefits realized by industry, extenuating socio-economic conditions, cost effectiveness of recovering the costs, and consistency with cost-recovery policies in other areas of the Commonwealth (Cox 2003). Stage two grants the government flexibility to address special considerations such as the preservation of a traditional way of life and the ability of different sectors of the fishing industry to pay for management (i.e., small-scale vs. large-scale operators). Assigning costs of management activities such as enforcement, rule making, and administration of fisheries management is clear under this approach, but the costs and benefits of research are complicated. As a result, the costs of research are shared by industry and government. Approximately 25% of research administered by the largest Australian fisheries research agency is paid for by a levy on the gross value of landings across all Australian fisheries; this amount is matched by the Commonwealth government, which also provides unmatched funding equal to 0.5% of the gross value of fisheries production(Cox 2003).

The New Zealand approach extends the principle of 'assignable cost' to include 'assignable risk' - that is, management costs are to be recovered from parties who benefit from, or generate risks in the marine environment (Stokes et al. 2006). The cost-recovery approach in New Zealand is highly complex, and rooted in dissatisfaction and distrust of publicly-funded research. Specific concerns surround the 'provider capture' of science, which is the tendency of (government) service providers to advance their own interests in identifying research projects and obtaining industry funding for projects that may not be relevant to fisheries management (Stokes et al. 2006). In the current cost-recovery scheme the provision of research is 'contestable' and the projects identified by the Ministry of Fisheries are adjudicated in annual planning meetings attended by industry and government scientists and fisheries managers. Research services procured by the Ministry and paid for by industry focus on environmental effects, catch monitoring, and (primarily) stock assessment (Stokes et al. 2006). Attribution of research costs is clear when it occurs on a stock-by-stock basis, but projects involving multiple stocks and the ecosystem effects of fishing are highly contentious because of the public-good aspects of such research (Stokes et al. 2006).

In contrast to Australia and New Zealand, the Canadian approach to cost-recovery is neither cohesive nor rational (Kaufmann and Geen 1997). The Canadian approach is best described as a 'fee system' in which the government charges 'access fees' that reflect the estimated value of the benefit of having access to a valuable natural resource - the value of the fee is not directly linked to the costs of management, and as such it is not a true cost-recovery system[6] The failure to link the fee to the realized costs

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[6] Treasury Board of Canada Departmental Performance Reports, Fisheries and Oceans

means that the charge will exceed the costs of management in some fisheries, and be significantly lower in others. This creates issues of fairness within both the fishing sector and society at large. For example, the Pacific salmon fishery was estimated to create a net loss to society of $55 million per year during 1988-1994, after accounting for the costs of management and provision of employment insurance to the industry (Schwindt et al. 2000). An additional issue of fairness arises because the fees are not tied to fishery revenues - this yields a significant imbalance in the ratio of access fees to the landed value by sector [7]. Profitable fisheries bear significantly lower relative costs (as low as 1% of landed value) than fisheries with declining value (as high as 10% of landed value). In implementing the fee system, DFO was reluctant to use the terms 'cost-recovery', 'resource rent' or 'tax' when developing the fee system (Kaufmann and Geen 1997). However, the intent of the policy was to shift the costs of fisheries management to the industry (cost-recovery), and to charge industry a fee that reflects the value of the privilege to exploit valuable public resources (by definition, a resource rental) (Kaufmann and Geen 1997). The phrase 'cost-recovery' has begun to be used in relation to activities DFO requires the industry to undertake as a condition of licencing (such as dockside monitoring) and in association with co-management agreements, but it is not used to describe the overarching system of fee collection in Canada.

Where it exists, the Canadian system of cost-recovery is applied to the most profitable components of the sector, which have been acting to 'fill the gap' in organisational budgets that arose in the 1990s after the federal deficit reduction program known as "Program Review" (Lane and Stephenson 2000). A key failing of the Canadian approach to cost-recovery is that there is no national policy specifying which

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Canada 2007-2008. Online access: http://www.tbs-sct.gc.ca/dpr-rmr/2007-2008/inst/dfo/st-ts02-eng.asp#dfo_Note_3

[7] Cumulative Impact of Federal User Fees on the Commercial Fish Harvesting Sector. Report prepared for Fisheries and Oceans Canada, 1999. Online access: http://www.dfo-mpo.gc.ca/reports-rapports/gp-ga_e.pdf

science activities to assign to industry and which should remain the responsibility of DFO (Wilson 2008). As such, the scope of industry-funded science across the country is broad, ranging from data gathering and participating in surveys, to conducting annual stock assessments and evaluating the likelihood of success of alternative management strategies (e.g., Cox and Kronlund 2008, Lane and Stephenson 2000). The lack of cohesion extends to the long-term contracts between DFO and industry that specify the roles and responsibilities of each party for providing and participating in management related activities (Wilson 2008). Under most of these "Joint Project Agreements" (JPAs), DFO agrees to provide management services which industry finances through direct payments, in-kind contributions (such as ship time) and other arrangements (Wilson 2008). The details of the JPAs are negotiated on a fishery-by-fishery basis and are not made public. However, the previously mentioned Larocque Decision in 2006 voided many of these arrangements and resulted in confusion and uncertainty regarding the roles and responsibilities of both DFO and industry for providing management-related services.

4.6 Potential Threats to the Quality of Fisheries Science in Canada

The division of the public and private aspects of fisheries science is not clearly articulated in Canadian policy, but the governance changes described in the previous sections nonetheless reflect an institutional view that stock assessment and other Mode-2 science is at least the partial responsibility of the fishing industry, and environmental and ecosystem-based research is the primary responsibility of government[8]. This perspective is reflected in the allocation of public funds for federal research and

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[8] 8The Blue Paper: The Future of Aquatic Science in Canada (Oceans Emphasis). Prepared by Jake Rice. Online access: http://www.dfo-mpo.gc.ca/science/publications/aquat-2020/bluebleu/pdf/paper-eng.pdf

in the five-year (2007-2012) research agenda of DFO[9], which emphasizes ecosystems, biodiversity, aquaculture, and climate change, with the expectation that research in these areas will inform stock assessments in the future. These research priorities are intended to support an integrated approach to management, and to continue to promote world-class public science in Canada. However, these priority topics may have been formulated without explicit consideration of the policy implications of a shifting funding emphasis.

4.6.1 Institutional pressures on Canadian fisheries science

Canadian fisheries science has been marked by a tension between political interests and scientific independence since its integration into DFO in 1979 (Hutchings et al. 1997). This arrangement has enabled political and bureaucratic objectives to exert significant influence on the generation, transmission, and use of scientific information (Hutchings et al. 1997). Involvement of non-scientists in the fisheries science process is therefore a well established phenomenon in Canada, however, the scope of this involvement is changing with the governance paradigm. The intensification of the Mode-2 paradigm in Canadian fisheries science is reflected in the shift toward industry-DFO funding arrangements that were described above. This shift has resulted in fisheries scientists within DFO facing new personal and professional pressures that are not formally acknowledged in the current science system. Without sufficient institutional barriers to deflect pressure from individuals, DFO opens its scientists to the compounded influence of bureaucratic directives and powerful stakeholder interests, which may promote biased interpretations of data and limit the scope of research that is required to generate information for fisheries management decisions.

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[9] Fisheries and Oceans Canada Five-Year Research Agenda (2007-2012). Online access: http://www.dfo-mpo.gc.ca/science/publications/fiveyear-quinquennal/index-eng.htm#a42

4.6.1.1 Pressures internal to DFO

DFO science is increasingly managed by multiple layers of bureaucracy, and many managers of science programs have little or no research experience (Shelton 2007a). This bureaucratization has changed the standards used to evaluate scientists and has begun to de-emphasize scientific credentials and knowledge in favor of management-specific criteria that reflect the degree to which scientific endeavors promote the aspirations of senior officials (Trecarten, 2001 cited in Shelton 2007a). The commercialization of science has paralleled this change, and has entrenched the focus of managers on the needs of science 'clients' (primarily the industry). While it has promoted close working relationships between industry and scientists, DFO management has shown little interest in the details of the Joint Project Agreements, the degree to which the resultant research promotes (or does not promote) the public good, or the potential damage these types of arrangements invoked on public perceptions of federal science (Shelton 2007a)[10]. The latter point is an important one, perceptions that public science is of low quality have weakened the role of science in Atlantic groundfish management decision making (Shelton 2007b), and scientists are increasingly facing the scrutiny of the public. A prominent example is the Cohen Commission of Inquiry into the Decline of Sockeye Salmon in the Fraser River that is currently underway in British Columbia. Negative perceptions of fisheries science are compounded by repeated, non-scientific criticisms and selective use of scientific results by senior bureaucrats at DFO (Hutchings et al. 1997). These issues are strong disincentives for scientists to engage in the policy process, but they are not unique to Canada. Scientists in the United States have pondered the high personal, and sometimes professional, cost to scientists of engaging in the fishery management process, and suggest that there is a 'looming crisis' in fisheries science that may result in government scientists avoiding applied issues altogether (Sissenwine and Mace 2003).

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[10] Similar concerns are raised by the Professional Institute of the Public Service of Canada, http://www.hyper-media.ca/pipsc/#

4.6.1.2 Pressures external to DFO

The working relationship between the fishing industry and DFO scientists is determined on a fishery-by-fishery basis, following the details of the JPAs (Wilson 2008). In some cases the industry has extensive access to individual scientists, without the oversight of managers or higher-level bureaucrats. This can create a confusing situation for both the industry officials, who may feel that DFO works 'for' them (an understandable assumption, given the repeated reference by DFO to the industry as the 'client' of DFO), as well as the scientist who is aware of broader public-good aspects of research, and who may, and perhaps should be, engaging with other stakeholders in science meetings (e.g., First Nations and environmental groups). There may be strong personal and professional resistance to the institutional positioning of the scientist as an advocate for one particular industry group. Conversely, the lack of formal guidelines clearly specifying the role of a government scientist in 'partnership' arrangements exposes DFO to the agency capture phenomenon, in which an entire agency (or subset thereof) comes to identify so closely with those they are charged with regulating that they lose motivation for preserving the public interest and the independence of the management agency (Singleton 2000). The potential for such pressures to influence the nature and scope of Canadian fisheries science is unknown at present.

4.6.2 Diminishing resources for fisheries science

The Larocque Decision has created a substantial gap in the provision of high-quality, Mode-2 fisheries science. As previously discussed, the governance changes in Canadian fisheries management were intended to devolve responsibility for fisheries science and management to the resource users, and to free up Departmental resources for generalized research on oceans and ecosystems. These parallel objectives were partially achieved by entering into collaborative agreements with the fishing industry which filled the budget shortfalls for stock assessment and management that arose after Program Review. Post-Larocque, the challenges facing Mode-2 science are significant. Foremost is the erosion of the quality of data used to generate science-based advice for management. The most obvious and well-known effect of the decision has been to threaten the funding available for scientific fishery-independent surveys of fish stocks, which are the primary avenue for industry collaboration in Canada. Such research surveys provide basic information for conservation and management decisions, and require careful planning and evaluation in order to ensure that they accurately represent the fish population of interest (see Chapter 2 of this Dissertation for an extensive discussion on this point). In recognition of their importance, DFO has enacted an Interim Adjustment Strategy to replace funding for surveys in fisheries impacted by the Larocque Decision (footnote 3). However, the level of funding is significantly lower than that available in previous years: DFO estimates that prior to 2006 it had access to between \$10 and \$15 million per year as a result of industry co-management agreements on the Pacific coast. This amount dropped to \$3.5 million in 2007.[11] Interim funding is available until 2013 for fisheries impacted by the Larocque Decision, with no guarantee of new funding after that date.[12]The impact on individual fisheries is perhaps more informative. For example, the B.C. herring fishery provided \$4 million per year for in-season test fishing and stock assessment in years prior to the Larocque Decision. In 2007 the industry contribution ended and DFO provided only \$900,000 for both activities. The reduction in funding restricted the spatial coverage and the number of samples collected during the annual herring survey (Schweigert et al. 2009). The degree to which these changes affect the quality of advice provided

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[11] Cost estimates in this section were obtained from testimony presented to the Standing Committee on Fisheries and Oceans by Christina Burridge, Executive Director, B.C. Seafood Alliance on May 3, 2007.

[12] Lisa Mijacika, DFO. Presentation at the Integrated Herring Harvest Planning Committee, October 14, 2010, Vancouver BC. Meeting minutes available online at: http://www.pac.dfo-mpo.gc.ca/consultation/fisheries-peche/pelag/her-har/ihhpc/docs/min-pv/2010-10-14.pdf).

to managers has not been evaluated.

Another significant impact of the Larocque Decision is that it has created an immediate requirement for strategic Mode-2 research, which, with the exception of a select few examples (e.g., Cox and Kronlund 2008), neither DFO nor the fishing industry appear to be addressing. Such strategic research would involve evaluating the management implications of reduced data quality, and developing alternative management strategies that can meet fishery objectives given a lower (i.e. cheaper) level of data collection (Kelly and Codling 2006). The absence of such evaluations implies a degradation in the standards of information used to generate scientific advice for management. In particular, it calls into question the adherence of Canadian fisheries science to the standards of experimental design and data analysis outlined in Table 4.2.

4.6.3 'Consensus' on the drivers of change in fish stock abundance

The 'Thompson-Burkenroad Debate' is well-documented in the fisheries literature, broadly referring to a long-running debate between fisheries oceanographers and fisheries ecologists about the relative importance of environmental change and overfishing as causes of declines in fish abundance. Decades of research and scores of correlative studies have not yielded an unequivocal result as to which process has dominated fisheries collapses and/or failure to recover from collapse, because the effects of fishing and environmental variation tend to be confounded in fisheries data (Walters and Martell 2004). As such, there is a high degree of uncertainty about the mechanisms governing variations in abundance of marine fish species. The inability of science to differentiate between environmental and fishery-induced causes of population decline presents an interesting problem when allocating public funding for research. The process of science involves evaluating multiple hypotheses by examining data, which may alter the degree of support for the initial hypotheses. Deliberately biasing an interpretation of data to illegitimately favor on hypothesis may therefore degrade the quality of information provided to managers. Hutchings et al. (1997) documented such a bias in the 1995 groundfish Stock Status Reports (SSR, renamed Science Advisory Reports (SAR) in 2005) on the east coast of Canada. SARs are the primary tool used to communicate the current state of scientific knowledge on fish stocks managed by DFO, and they reflect the Department's 'scientific consensus' on fish stock status. Hutchings et al. found the groundfish SSR favored environmental explanations of the northern cod collapse and suppressed (internal) research that found fishery effects. Hutchings et al. concluded that the Canadian public was poorly served by the failure of DFO to present all hypotheses and to convey the full uncertainty about cod population dynamics. Recent explanations of dramatic declines in Pacific herring stocks in British Columbia suggest that this may be a persistent problem; they dismiss fishing as a potential cause of decline on the basis that herring are managed using a 20% harvest rate and a biological reference point (Schweigert et al. 2010). However, this management approach failed to consistently meet conservation objectives in simulation tests of robustness (Cleary et al. 2010). The decline in herring spawning biomass is instead presented as an environment-driven, ecosystem-based process that is associated with an apparent increase in natural mortality (Schweigert et al. 2010). By discounting fishery impacts, Schweigert et al. fail to convey the complexities and interactions between climate and fishing effects; they make no mention of the potential for fishing to increase sensitivity of herring populations to climate change by removing older individuals, reducing population complexity, altering life history traits, and changing the spatial distribution of the populations(Perry et al. 2010).

The degree to which environmental explanations for declines in fish populations are favored by scientists, and their potential relationship to funding priorities at DFO has not been established. However, the northern cod and Pacific herring examples indicate that unbalanced representations of the drivers of fish population dynamics can occur at DFO. Without clear articulation of the assumptions and uncertainties inherent in the analysis, such explanations may be perceived by managers and the public as reflecting scientific consensus. Furthermore, they can position government scientists as 'stealth advocates', unknowingly promoting policies that under-represent the risks associated with harvesting marine fish populations. This role does not align with the standards for transmission of scientific information, which would require scientific advice to be neutral, transparent, open about uncertainty and risk, and honest about the complexity of the management problem (Table 4.2). In addition, this role restricts the ability of managers to understand the biases and limitations of scientific advice, and to modify their decisions accordingly.

4.7 Summary and Recommendations

The objective of this chapter was to evaluate the impact of a shift of Canadian fisheries science policy toward cost-recovery and co-management. However, closer evaluation of the trends in science policy indicates that there are significant legal and institutional constraints on Canadian fisheries governance that have restricted a complete shift in this direction. These restrictions stem from a disjointed governance system at DFO, in which the authority for planning and executing management decisions rests with the Minister of Fisheries and Oceans, but where institutional reforms have been made in favor of devolving some responsibilities for fisheries science and management to stakeholders. The Larocque Decision challenged aspects of these reforms on the basis that they violated a fundamental legal principle. Specifically, the decision holds that the Minister can not appropriate publicly owned resources in order to increase its operational budgets. By overturning many management agreements between industry and DFO, the decision has significantly limited budgets for fisheries science and management. The Government of Canada has provided funding to support fisheries management activities as an interim measure while seeking to formalize a new governance model that would permit co-management arrangements with stakeholders, thereby “modernizing” the Fisheries Act[13]. However, the proposed modifications to the Act have met with significant public concerns about whether moving away from a patriarchal management structure would create a power imbalance and yield control of important management decisions to industry organizations14.[14]. There appears to be little recognition of the implications of governance shifts on the quality of fisheries science in Canada. In this regard, it is not clear that the current bureaucratic structure of DFO promotes the generation, transmission, and use of high-quality science in fisheries management. By preferentially funding oceans and ecosystems research over stock assessment, this structure appears to have entrenched a scientific perspective that favors environmental explanations of variability in fish population abundance, and may reduce the investigation of competing hypotheses that point to the potential negative impacts of fishing. Furthermore, involving industry directly in the planning and execution of fisheries science may promote the funding effect, which could further deflect attention from research into fishery effects on populations. These are non-trivial issues that have not received adequate attention from Canadian fisheries scientists, industry groups, and policy makers. However, the Larocque Decision has created an opportunity to involve all parties in the development of institutions that will support ongoing funding of fisheries science and which will enable scientists to appropriately engage in the policy process.

A key requirement of any new governance arrangement will be a clear and transparent rationale for partitioning fisheries management costs among public and private agencies. The current ’fee-based’ approach in Canada is not transparent and is not fairly applied across all sectors of the fishing industry. These problems arise from

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[13] The details of the proposed changes are in Bill C-32: An Act respecting the sustainable development of Canada’s seacoast and inland fisheries

[14] Modernizing the Fisheries Act, Presentation Deck. http://www.dfo-mpo.gc.ca/far-rlp/c32-presentation-eng.htm

the failure of the Canadian government to openly institute a policy of cost-recovery (Kaufmann and Geen 1997). Such a system would explicitly link the fees charged to the industry to the costs of providing management-related activities (including fisheries science), and would promote the development of effective management institutions. Kaufmann and Geen (1997) suggest that fishery management agencies lack the interest and ability to design cost-recovery policies, and point to the importance of engaging finance and treasury departments in their design and implementation. This moves the required governance changes beyond DFO, and potentially beyond the current proposed changes to the Fisheries Act. DFO should further consider restructuring the management of science as the shift to Mode-2 continues. In particular, it is important to develop a national approach to science that seeks to develop a set of standards for the evaluation of management-related information. The standards should include national conservation objectives such as the national harvest policy under the Sustainable Fisheries Framework (Shelton and Sinclair 2008), but should be broadened to ensure that the transmission and use of management-related information are not biased. This would provide assurance to all parties of the continued provision of high quality science in a strong Mode-2 management arrangement.

Table 4.1: Broad characteristics of Mode-1 and Mode-2 science (Dasgupta and David 1994, Gibbons 2002, Wald 2007).

	Mode-1	Mode-2
Funding arrangements		
	public, guaranteed	private partnerships, performance-based
Principles governing the production of knowledge		
	objectivity autonomy universalism disciplinary peer-review	incorporation of values collaboration specific application inter-disciplinary peer-review, social & economic criteria

Table 4.2: Standards of Information for Canadian Fisheries Management

Criterion	Key Sources
Generation	
(1) Clear statement of objectives	Botsford et al. (1997)
(2) Conceptual model of the process/system	Sullivan et al. (2006) (2)-(6)
(3) Good experimental design and standard method of data collection	
(4) Statistically rigorous analysis	
(5) Clear documentation of methods, results, and conclusions	
(6) Peer review	
(7) Ecosystem-based approach	Pikitch et al. (2004)
(8) Interdisciplinary - social and economic research	Stephenson and Lane (1995)
Transmission	
(1) Role: science arbiter	Lackey (2007)
(2) Openness	SAGE Guidelines[15]
(3) Transparency	Kraak et al (2010)
(4) Communicate uncertainties and risk	FAO (1995)
(5) Accurate representation of complexity in an understandable manner for the audience	Peterman (2004)

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[15] See section 4.3

(Table 4.2 continued)

Criterion	Key Sources
Use	
(1) Strategic use of knowledge to identify emerging needs	DFO SAPF[16]
(2) Inclusiveness - seek out diverse opinions	SAGE Guidelines
(3) Develop an awareness of biases in different sources of information	DFO SAPF
(4) Evaluate impact of knowledge on previous decisions (learning)	SAGE Guidelines

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[16] See footnote 5

Table 4.3: Potential arrangements for the provision of fisheries management services (Arnason 2003).

	Arrangement	Provider	Payee
1	Centralized	Government	Government
2	Cost-recovery	Government	Fishing Industry
3	Contracting out	Private Sector	Government
4	Self-management	Fishing Industry	Fishing Industry

Summary and General Conclusions

This thesis focused on the concept of biodiversity for fisheries management. While biodiversity is widely accepted as a desirable feature of marine ecosystems, it has not been adequately integrated into operational objectives, indicators, and methods of evaluation. This limitation has restricted its utility for traditional fisheries management, which focuses on single-species, single-population "stocks". The single-species approach has been widely criticized as narrow in scope, far removed from the realities of data collection and fishery dynamics, and perhaps irrelevant for holistic management of ecosystems. However, the theories upon which single-species fisheries science developed recognize diversity in time and across space within populations. I recommend revisiting this approach in order to develop or improve existing protocols for data collection, stock assessment, and decision making in order to begin to operationalize biodiversity-based management (Chapter 1). Debate over the definition of a fish population has been central to fisheries science and management for over 100 years, and it is directly relevant for biodiversity research. Most recent definitions admit that fish populations are not single, homogeneous aggregations of fish, and that many 'stocks' defined for management can include groups of fish that exhibit significant diversity in their behaviour, phenotype, and genotype (Chapter 1). Such diversity is manifest over space and time as groups of fish differ in their use of available habitat. This spatial diversity is often ignored in fisheries management because of uncertainty related to the mechanisms governing population structure (e.g., dispersal rates, local population density, and life history variation), and because existing data collection programs are insufficient for conducting finescale stock assessments (Chapter 2). However, spatial diversity within populations 136 contributes to their ability to adapt and persist over a broad range of environmental conditions. Any loss of spatial diversity - signalled by genetic, phenotypic, or behavioural homogenization - may therefore reflect reduced resilience to anthropogenic and environmental change. Pacific herring exhibit marked variation in both the timing and distribution of spawning, a feature that may be important for the persistence of the populations. These dynamics are not formally included in the management structure for herring, which focuses on maintaining the total biomass within a management (stock) area. Disaggregating the Strait of Georgia (SOG) herring spawning data to the finest scale reveals a striking pattern of concentrating spawning in time and space since the 1970s (Chapter 2). However, the interpretation and significance of this pattern is unclear because of changes in the survey protocol over time. Specifically, the spatial coverage of the survey may have changed such that it is difficult to confirm the absence of spawning herring in some sites over time. This feature of the data has created uncertainty regarding both the quality of the SOG herring survey, and the impact of ignoring the spatial diversity in stock assessment and management. Simulation studies have shown that failing to account for spatial diversity in management can lead to a loss of sub-populations, and erosion of complexity within populations. These results imply that management must operate on the appropriate biological scale (i.e. treat population components separately) in order to be biologically conservative. I developed a closed-loop simulation model to test the hypothesis that ignoring spatial complexity will lead to management failures for SOG herring (Chapter 3). Previous simulation studies on other fisheries made simplifying assumptions about the spatial dynamics of fish populations, fisheries, or both, which makes it difficult to draw broad conclusions about the need for fine-scale fisheries management. I relaxed those restrictive assumptions by taking these processes explicitly into account and evaluating their impact on management outcomes. My analysis indicates that the combined spatial dynamics of fish populations and fisheries have powerful implications for fisheries management. In particular, the aspects of spatial structure that appear to be most important for management (i.e. the processes governing the overlap between populations and fisheries) may not be reflected in standard measures of stock structure such as genetic diversity. I recommend using in-season management to control the local impact of fisheries on spatially diverse populations. My research suggests that fisheries science can begin to accomodate biodiversitytype objectives and indicators by recognizing spatial dynamics and complexity within fish populations. However, the integration of biodiversity (and ultimately, ecosystems) into tactical fisheries management may be restricted by changes in science policy in Canada (Chapter 4). Large reductions in federal science budgets have promoted a distinction between ecosystem and fishery science, which appear to be viewed as public- and private-goods, respectively. Current research plans at DFO reflect this distinction; ecosystem-level research and monitoring are the priority areas for DFO science, and the fishing industry is expected to contribute to stock assessment and management. I explore the potential for changes in available funding and the adoption of the user-pay approach to providing management services (including science) to impact the quality of advice provided to decision-makers. This research reveals several challenges for fisheries science in Canada, including, but not limited to:

1. political and economic pressures on the scope and interpretation of fisheries science;
2. uncertain and variable funding for fisheries surveys and stock assessments;
3. preferential funding of research that addresses environmental and ecosystem drivers of variation in fish populations over fishery effects;

4. a shortage of stock assessment expertise.

The final chapter of this dissertation provides clues about the origins of the patterns that are observed and discussed in the previous chapters. For example, the changing survey design in the B.C. herring data example (Chapter 2) is at least partly related to an increased reliance on industry cooperation and funding for the annual surveys of spawning biomass. The herring fishery is managed using quotas, and this approach requires good data in order to obtain stable estimates of total biomass. The definition of 'good data' that is applied to herring is to sample the largest spawning aggregations in order to minimize the variance in survey estimates of abundance. Ensuring adequate and consistent spatial coverage of both large and small spawning components of the population does not appear to serve the private interests of the herring industry. Similarly, developing stock assessment methods to account for biodiversity has not featured on the industry science agenda. However, the industry did fund research into stock structure in order to address concerns about overfishing of distinct stocks within each of the management areas. This research identified genetically distinct stocks, but did not affect the scale at which DFO manages herring populations (Chapter 3). These results indicate that although biodiversity features prominently on the political agenda in Canada, explicit and active management of fisheries science from a public-good perspective is required in order to operationalize biodiversity for management.

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