EVALUATION OF MANAGEMENT SYSTEMS FOR KS^{n} FISHERIES AND POTENTIAL APPLICATION TO BRITISH COLUMBIA'S INSHORE ROCKFISH FISHERY

by

Michael Hamilton
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APPROVAL

Name:	**Michael Hamilton**
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Examining Committee:

Chair: Garret Cooper
Master's Student, Fisheries Research Group, School of Resource and Environmental Management, SFU

Dr. Sean Cox
Senior Supervisor
Assistant Professor, School of Resource and Environmental Management, SFU

Dr. Andy Cooper
Supervisor
Associate Professor, School of Resource and Environmental Management, SFU

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ABSTRACT

Some fisheries target marine organisms that are benthic, late to mature, long-lived, sedentary, and those structured as geographically-isolated sub-populations which together form a larger metapopulation. In this study, such fisheries were defined as KS^n fisheries. The life-history and spatial characteristics of species targeted in KS^n fisheries leave them particularly susceptible to management and stock assessment challenges, resulting in a higher risk of overfishing and localized depletion. The inshore rockfish fishery in British Columbia, Canada, is a specific example of a KS^n fishery. 13 KS^n fisheries were examined in the context of a conceptual framework to determine methods of fishery sustainability/success, and through this management advice was provided for the inshore rockfish fishery. It was determined that the implementation of a territorial use rights fishery (TURF), an exclusive form of resource access, worked to increase fisher incentive for resource sustainability, the major obstacle to success in the inshore rockfish fishery.

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1. INTRODUCTION

Failure to manage the impacts of commercial and recreational fishing at spatial scales that match the biological scale over which marine populations function may have undesirable long-term consequences for fisheries resources (Ludwig et al. 1993; Carvalho and Hauser 1994; Botsford et al. 1997; Orensanz and Jamieson 1998; Hilborn et al. 2003a/b). Conservation of spatial population structure is especially important where organisms exhibit diverse life-history characteristics such as growth and mortality, maturation, and dispersal rates over space (i.e., biocomplexity: emergent properties of components of a system working together in a complex manner). Biocomplexity buffers an ecosystem to change due to nonlinearity in ecosystem dynamics (Levin 1998), and biocomplexity in the form of spatial population structure is important for maintaining the resilience of populations in highly variable environments (Holling and Meffe 1996; Hilborn et al. 2003b; Berkeley et al. 2004).

Most conventional fisheries stock assessment and management approaches assume that fisheries harvest single, panmictic (i.e., unstructured in terms of mating populations) stocks that exhibit stationary average life-history characteristics (Gulland 1969). For some fisheries, harvest information is detailed enough and fish population structure is coarse enough (e.g., via migration and dispersal) to account for whatever stock structure appears to exist. For instance, although stock structure exists for Pacific Ocean perch (*Sebastes alutus*) (Gharrett et al. 2006), their relatively high movement and mixing has allowed most fisheries to continue without fully depleting the stock. Notwithstanding the success of some Pacific Ocean perch fisheries, most assessment and management systems do not collect information at the small scales necessary to account for the spatial heterogeneity of harvested marine organisms (Ludwig et al. 1993; Carvalho and Hauser 1994; Botsford et al. 1997; Orensanz and Jamieson 1998; Prince 2005). For example, sockeye salmon (*Onchorhychus nerka*) in the Bristol Bay region of Alaska exhibit a wide range of spawning location and substrate choices (e.g., small creeks, lake beaches, and large rivers) even within a single lake system or ‘stock’ (Hilborn et al. 2003b; Hilborn 2006). Although the management agency enumerates these spawning behaviour types individually in the spawning areas, they cannot be identified and individually targeted in mixed-stock ocean and river fisheries. Such indiscriminate harvesting may therefore reduce the long-term capacity of these sockeye salmon populations to respond to changes to their spawning grounds such as changes to gravel substrate in a stream (Hilborn et al. 2003b; Hilborn 2006).

Sedentary populations are populations that do not readily inter-breed with other populations due to spatial separation, and include species with relatively little migratory stock movement that typically inhabit the same region throughout their lifecycle (Christy 1982). Unlike salmon populations, which undergo long migrations and often disperse among local populations (Hilborn et al. 2003b), sedentary species typically exist within very narrow environmental conditions for their entire lifetime. Local adaptation to environmental conditions may result in sedentary populations due to the lack of migration, and population resilience is sensitive to changes in connectivity among sedentary sub-populations (Hilborn et al. 2003b; Prince 2005).

The rate of re-population following depletion in sedentary populations is dependent upon hydrologic connectivity, the water-mediated transport of matter, energy and organisms (Freeman et al. 2007). For example, to sustain stock productivity sub-populations may rely on re-population from other sub-populations through transport of larval propagules.

Sedentary populations may exhibit unique life-history and/or spatial characteristics that vary depending on factors such as latitude, depth, and substrate (Hilborn and Walters 1992; Orensanz and Jamieson 1998; Walters 2000; Berkeley et al. 2004). For instance, size-at-age in some rockfish (*Sebastes spp*.) varies by latitude, and fishing pressure and evolutionary differences in reproductive strategy are thought to be factors driving the variation (Boehlert and Kappenman 1980, Boehlert and Kappenman 1980; Gunderson et al. 1980; Pearson and Hightower 1991;

Yamanaka and Lacko 2001). Effective management of sedentary species often depends on an adequate match between the spatial scale of management and the spatial scale of sedentary populations (Prince 2005). Prince (2005) defined the spatial scale of sedentary stocks, or the actual scale of component units of stock, as self-recruiting units of stock, where population exchange between units of stock occurs at a low rate.

A 'tragedy of the commons' (Hardin 1968), where overexploitation results from individuals maximizing their own welfare, may result from unsustainable fishing pressure on fish populations when resource access is not regulated (Hilborn et al. 2005). Effects of the tragedy of the commons are of particular concern for species with slow body growth, longevity, late age of sexual maturity, low rates of population increase, and relatively large body size. These species are referred to as 'K-selected' species (MacArthur and Wilson 1967) and they have low sustainable yields in relation to virgin biomass (Hilborn 2003a). Rebuilding populations of K-selected species following overfishing therefore occurs at a relatively low rate.

A tyranny of scale is the mismatch between the scale of management and the scale of component units of stock (Prince 2005). A tragedy of the commons may increase a tyranny of scale because fishers have little incentive to cease fishing pressure on a sedentary population (Prince 2005) because continual fishing on a single sedentary population requires lower effort and is more cost-effective than searching for new fishing grounds.

Sedentary populations are susceptible to localized depletion (Hilbon 2003a; Prince 2005). Prince (2005) explained localized depletion as the depletion of a single component unit of stock. A tragedy of the commons may increase localized depletion of sedentary populations, particularly when a tyranny of scale exists, because fishers have little financial incentive to conserve individual units of stock. When sedentary populations, localized depletion, tragedy of the commons, and a tyranny of scale are present in a fishery, unique management and stock assessment challenges result. These challenges are compounded in fisheries for K-selected species because they have relatively low sustainable yields and rebuild at a relatively low rate.

1.1. KS^n fisheries

Parma et al. (2001) referred to small-scale fisheries for spatially-structured, sedentary stocks as ‘S-Fisheries’ because this particular category of fishery can be described by several ‘S’ words. Specifically, Parma et al. (2001) identified fisheries that: (i) are ‘small-scale’ in terms of the size of fishing vessels (small boats normally less than 10m. long), (ii) have ‘sedentary stocks’ for which adult movement is low and thus much of the life cycle is completed over a small region in space, and (iii) have ‘spatially-structured stocks’ or metapopulations that are interconnected through larval or juvenile dispersal and that exhibit persistent sedentary populations.

S-Fisheries are not by definition K-selected, and as such some S-Fisheries may be less susceptible to overfishing, may sustain higher rates of exploitation, and may rebuild from depletion at a relatively high rate (Adams 1980; Myers et al. 1997; Musick 1999; Hutchings and Reynolds 2004; Jennings et al. 2004). For instance, Jennings et al. (2004) found that large, late-maturing fish species were less able to sustain a given rate of fishing mortality, relative to smaller fish that mature at an earlier age. Hutchings and Reynolds (2004) focused upon the collapse of Atlantic Cod (*Gadus morhua*) and found that persistence and recovery of an exploited fish stock is limited by life-history and especially by delayed maturity.

Management strategies that are effective in some S-Fisheries may not be sustainable in S-Fisheries for K-selected species because the population dynamics of K-selected species operate over longer timescales that are typically not conducive to sustainable fishing under traditional stock assessment and management practices (Hilborn 2003b). Also, the examination of sustainability of S-Fisheries has focused mainly on benthic invertebrates in artisanal fisheries (Parma et al. 2001; Orensanz et al. 2004), which makes extrapolation to fish species difficult.

Building on the category of the S-Fishery, the term ‘KS^n fishery’ may be used to describe S-Fisheries for K-selected marine species where many ‘S’ words may be included for these species, including slow growth, sessile (e.g., many marine invertebrates), spasmodic successful recruitment, and susceptibility to both a tyranny of scale and serial (i.e., large-scale local) depletion. There has been limited attention paid to fisheries for species that are also susceptible to both a tyranny of scale and serial depletion.

1.2. Biology of KS^n species

Regarding life-history, r-selection and K-selection relate to the selection of organism traits that permit success in particular environments (MacArthur and Wilson 1967). The intrinsic rate of population increase, ‘r’, is the sum of the change in the amount of harvestable stock estimated by recruitment and growth minus natural mortality. ‘r’ is used as a measure of how much a population can increase over time. Species with high ‘r’ values, or ‘r-selected’ species, typically have the ability to recover population size relatively quickly because the stock productivity (e.g., maximum number of recruits per spawner) is relatively high.

The carrying capacity of the local environment, or ‘K’, is the supportable population of an organism within that environment (MacArthur and Wilson 1967). Compared to r-selected species, the stock productivity of ‘K-selected’ species is low, and K-selected species are long-lived, which may be an evolutionary adaptation to promote iteroparity (Myers et al. 1997; Musick 1999; Hutchings and Reynolds 2004). K-selected species have high fecundity and a long reproductive period during their life-cycle. The longevity of K-selected species may be a reproductive adaptation to highly variable environments in which average juvenile survival is typically low but may occasionally be very high (Longhurst 2002). For this reason, some researchers argue that maintenance of a broad age structure is critical to the existence of K-selected marine populations because overfishing a population of K-selected organisms may truncate the age and size structure, thus reducing the number of older, mature, and highly fecund adults in the population (e.g., Berkeley et al. 2004). Such a change in population composition may decrease stock productivity and resilience during periods of low recruitment (Berkeley and Markle 1999; Berkeley et al. 2004).

Most species of rockfishes are K-selected because they are long-lived (20-140 years; Archibald et al. 1981; Leaman and Beamish 1984; Love et al. 1990), relatively unproductive (Adams 1980; Musick 1999), and reach sexual maturity at relatively old ages, ranging from 11 to 20 years depending on species (Yamanaka and Richards 1993; Kronlund and Yamanaka 2001). Furthermore, rockfish have a relatively large body size, and although some species produce up to 417,000 eggs per female (boccaccio – *S. paucispinis*; Haldorson and Love 1991), they tend to have low larval survivorship (Musick 1999). Some rockfish also have spasmodic recruitment success, which may be affected by oceanic conditions (Gunderson 1977; Leaman and Beamish 1984; Botsford et al. 1994).

KS^n fisheries may exhibit biocomplexity because they are spatially structured. Biocomplexity in KS^n fisheries has typically been apparent in differences in size-at-age and size-at-maturity over space. For instance, Kronlund and Yamanaka (2001) and Yamanaka and Lacko (2001) found differences in size-at-maturity over space for rockfish (*Sebastes spp.*). Prince (1989, 2005) found that abalone (*Haliotid spp*.) populations had highly variable sizes-at-maturity over space resulting from self-recruiting microstocks that composed the larger abalone population.

1.3. Management of KS^n fisheries

Localized depletion is a primary management concern in KS^n fisheries because it has the potential to reduce the biocomplexity, and thus the resiliency, of the stock (Holling and Meffe 1996; Levin 1998; Hilborn et al. 2003b). The targeting of sedentary (i.e., rarely moving, but not constrained to a single place) or sessile (i.e., attached to the substratum) marine organisms that have high site-fidelity around geomorphological features is increasingly effective with recent advances in fishing gear and navigational equipment (Walters and Martell 2003).

Along with advances in fishing gear, localized depletion may result from failure to adequately designate management areas that account for stock structure, thus inhibiting the ability to promptly recognize localized depletion of sub-populations of marine organisms (Prince 1989, 2005, for abalone). Localized depletion is documented in several fisheries. For instance, localized depletion of global sea urchin (*Strongylocentrous spp.*) resources has occurred at a range of scales, as has the over-fishing of the larger, older individuals (Botsford et al. 2004). The principal cause of sea urchin serial depletion and age-structure truncation has been the movement of fishers to select areas of high sea urchin density to maximise profit (Botsford et al. 2004). A management regime was not in place to detect and halt the localized depletion. Localized depletion due to fishing pressure in marine finfish was reported in Hanselman et al. (2007) for north Pacific rockfish. Again, a tyranny of scale is thought to be the mechanism that drove the localized depletion.

The effect of localized depletion is likely more severe in KS^n fisheries than in fisheries for species that are not K-selected. For instance, it may take decades for some species of rockfish to re-colonize and rebuild their population age-structure, due in part to the low productivity of rockfish stocks and irregular recruitment success from populated ('source') areas (Gunderson 1977; Leaman 1991; Ianelli and Ito 1992). Initial rebuilding of age-structure and repopulation of depleted areas for Pacific Ocean perch (*Sebastes alutus*) and cowcod (*Sebastes levis*) have been estimated to be approximately 23 and 40 years, respectively (Archibald et al. 1983; Jacobson and Cadrin 2002). Because of their inherent low productivity, K-selected species cannot recover quickly from fishing pressure, which leads fishers to move on to new, unexploited sub-populations to maintain their fishing profits. This results in a path of depleted sub-populations (i.e., serial depletion), undermining biocomplexity and decreasing the long-term resilience of the stock (Levin 1998).

Locally depleted sub-populations may eventually be repopulated through the process of larval dispersal (dispersal of planktonic larval propagules) from 'source' areas (e.g., Shanks et al. 2003). In this case, the concern for fishery sustainability would rest solely on overfishing the relatively panmictic stock if genetic variation (differences in physiology or behaviour) were relatively low within a metapopulation (Prince 2003).

1.3.1. Fishery management

One of the main roles of fishery management is to specify the total allowable catch (TAC) that may be taken from a management area or stock if the latter is known with any certainty. Methods of calculating TAC involve: (i) data, (ii) assessment of stock status, and (iii) regulation (de la Mare 1998). The following sections describe the main components involved in this assessment process and how each is especially problematic in KS^n fisheries.

Data

KS^n fisheries *typically* rely upon fishery-dependent catch and effort data to index stock condition under the tenuous assumption that catch-per-unit-effort (CPUE) is proportional to stock abundance or biomass. However, the use of fishery CPUE in stock assessment has many well-documented deficiencies. In this section, I describe a few particular difficulties associated with KS^n fisheries and the use of CPUE data. First, CPUE data collected at a course spatial scale (e.g., 100 km^2) cannot address the spatial complexity of KS^n fisheries (Richards 1994 for *Sebastes spp.*) because sub-populations in KS^n fisheries often occupy environments of less than 100 m^2 (Prince 1989, 2005). Second, CPUE is only expected to be proportional to abundance when handling times are small and the search for fish is random (Hilborn and Walters 1992); such conditions are rarely, if ever met in fisheries. Finally, for marine organisms that aggregate at specific habitat features, the potential for hyperstability exists (Breen 1992). Hyperstability is exhibited when CPUE is relatively constant over a wide range of abundances. Two mechanisms may result in such hyperstability (Hilborn and Walters 1992): (*i*) CPUE is maintained at high levels as fishers move between previously unexploited sub-populations (i.e., local and serial depletion) and/or (*ii*) fish re-aggregation around structural habitat features following short-term depletion from fishing.

Assessment of Stock Status

The calculation of TAC for KS^n fisheries is necessary to determine the level of acceptable fishing pressure on sub-populations to avoid localized depletion. Assessing KS^n fisheries is challenging not only due to hyperstability in CPUE data. There is also generally a deficiency of life-history data for long-lived species (Love 2002), and the susceptibility of some KS^n species to barotrauma complicates the use of in-situ stock monitoring (i.e., direct observation) since tagging techniques that require the fish be brought to the surface cannot be used. The expense of monitoring isolated populations and the difficulty in delineating stock boundaries further complicates assessment of KS^n fisheries. Also, KS^n fisheries target species with highly-variable recruitment success thereby complicating methods of stock assessment which are unable to promptly recognize levels of recruitment (e.g., limiting the usefulness of catch-curve analysis) (Yamanaka and Lacko 2001 for *Sebastes spp.*).

Regulation

Conventional fisheries management typically includes a combination of input and output controls in an attempt to limit total fishing mortality (Walters and Martell 2003). Fisheries managed using input controls seek to limit the intensity of fishing or the fishing effort. Typical input controls include a limit on areas and times fished (e.g., spatial or temporal closures), restrictions on fishing effort, restrictions on number of licenses, limits on type or size of vessels, and restrictions on fishing gear. Conversely, output controls place limits on the size or sex of species (e.g., size limits, bag limits) or on the total weight of the catch. Obvious examples of output controls include quota or TAC management systems.

Both input and output control of fisheries present unique challenges and risks that are particular to KS^n fisheries. Input controls on fishing effort have the potential to merely shift the spatial or temporal distribution of fishing effort rather than decreasing total effort. Because the total amount of fish caught is not under direct control, input controls are of limited use where bycatch, discarding, and high-grading are common. Finally, input controls rely on the assumption that the catchability coefficient is stable, and thus that fishing mortality rate is proportional to fishing effort (Walters and Martell 2003). Fishing mortality rate is most likely not proportional to fishing effort in KS^n fisheries, as discussed above regarding CPUE data. In contrast, output control management of fisheries is an effective means of limiting the total harvest from a stock if there is adequate monitoring and accounting for fishery catch, including catches that are discarded. However, output control systems place the challenge of conservation planning on the stock assessment science system by requiring estimates of stock size or biomass. Precise and unbiased estimation of stock biomass is extremely difficult even for intensively monitored fisheries. Finally, where catch quotas are not formally shared among harvesters (e.g., individual transferable quotas), output control systems may result in 'derby fishing' as individuals race to catch as much of the quota as possible before the fishery is closed (Walters and Martell 2003). Output control systems have specific risks in spatially structured stocks, namely: (i) the quota may be taken unevenly over the fishing grounds, (ii) reliance upon unbiased estimates of biomass, (iii) fishing effort may adjust to take the entire quota even when fish stocks are very low, which may lead to depensatory fishing mortality (Walters and Martel 2003) and thus a high risk of overfishing and possibly even stock collapse. Due to difficulties in estimating stock size, excessive concentration on applying data-heavy harvest control rules to specify fishing intensity or TAC in KS^n fisheries may not be practicable (Parma et al. 2001). Harvest control rules may not be practical since "it is generally true that data-poverty is symptomatic of more fundamental problems that need to be addressed [in S-Fisheries]", and "excessive focus on data shortage may obscure these more fundamental limitations [such as fisher incentive]" (Parma et al. 2001).

To manage fisheries where there exists a lack of basic biological data, it has been suggested that stocks be protected from overfishing caused by data limitations by combining conventional input/output approaches with other non-conventional fishery management approaches (e.g., Lauck et al. 1998). Non-conventional fishery management approaches include the use of closed areas, which simulation studies suggest may be beneficial for the management of sessile benthic species, when used in combination with minimum landing sizes and effort control (e.g., Quesne et al. 2007). Non-conventional fishery approaches may also place an emphasis on co-operative management through user-rights and fisher incentive for resource sustainability, address the importance of biodiversity, are less data-intensive, and are often designed to address realistic spatial complexity within a particular fishery (e.g., Castilla and Fernandez 1998; Caddy 1999; Perry et al. 1999; Botsford et al. 2003). For instance, Botsford et al. (2003) discuss the benefits of modelling marine reserves and suggest methods to account for uncertainty in fishery data. Caddy (1999) emphasize the benefits of stock indicators such as reference points to effectively manage data-poor fisheries. Castilla and Fernandez (1998) found that non-conventional management in the form of conferring quasi-property rights to fishers led to better management in artisanal Chilean inshore benthic fisheries.

The tragedy of the commons results in overexploitation of common property resources because exclusive user-rights (e.g., Hilborn et al. 2003a) do not exist and there is no individual benefit to resource conservation (Hardin 1968). Common property resources are "those to which access is both free and open to a set of users or potential users" (Christy 1982), and are characterized by lack of user exclusivity and by subtractability, where the use of the resource by one user-group/individual adversely affects its use by another user-group/individual. In an open-access fishery it is assumed that stakeholders act rationally (i.e., in their own interest), and therefore fish an area until it becomes uneconomical to continue, or participate in a 'race-to-fish' due to the limited timeframe during which fishing is permitted (Hilborn and Walters 1992). Resource waste, economic waste, and conflict among users are all products of the use of a common property resource (Christy 1982).

In KS^n fisheries, the presence of multiple user-groups without clear resource access rights may result in localized and serial depletion and a higher risk of overfishing due to a tragedy of the commons (Hardin 1968). For example, competition between artisanal and industrial fleets in the Chilean inshore benthic fisheries resulted in fishing ground conflicts and overfishing. To remedy this, exclusive fishing rights to specific areas within the fishing zone were given to each of the two groups to overcome the problems associated with conventional management (Castilla and Fernandez 1998).

Effective regulation depends on management goals of the fishery. In light of the above, effective regulation to maintain biocomplexity, and thus system resilience, in KS^n fisheries would be accomplished through a combination of non-conventional and conventional management, drawing on management methods used in relatively sustainable KS^n fisheries.

1.4. Research objectives

The fishery for inshore rockfish off the west coast of British Columbia (BC), Canada, (hereafter, 'the IR fishery') targets a complex of K-selected benthic rockfish species in recreational, aboriginal, and commercial fisheries. Species in the IR fishery are typically sedentary around geomorphological features on the ocean floor. The size of present management areas in the fishery may not account for the possible biocomplexity of the stock and IR are currently exhibiting localized depletion due to a tyranny of scale. More importantly, although non-conventional management approaches such as protected areas have recently been instituted in the fishery, the status of the stock is uncertain and there is no method in place to monitor changes in stock abundance (Yamanaka et al. 2004). The IR fishery is susceptible to the entire suite of management challenges just described, resulting in a high risk of overfishing.

The goal of this study is to identify a fishery management system that may potentially provide a sustainable IR fishery despite the major challenges and risks associated with KS^n fisheries. To accomplish this goal, I addressed the following objectives: (i) identify a set of sustainability criteria that adequately describe a "sustainable" fishery management system; (ii) use selected KS^n fisheries from around the world as model management systems and evaluate them in terms of the sustainability criteria; (iii) identify which management systems tend to promote sustainability of KS^n fisheries; and (iv) attempt to apply the management system(s) identified in (iii) to the IR fishery.

2. EVALUATION OF MODEL MANAGEMENT SYSTEMS FOR KS^N FISHERIES

2.1. Introduction

To accomplish the research objectives, the management systems of other KS^n fisheries were considered models from which to assess and improve management of the IR fishery. A model is a structurally descriptive representation of a system or process to aid in theoretical or empirical understanding (Barber et al. 2004). Models vary from qualitative or interpretive to mathematical or statistical (i.e., quantitative). Qualitative models are often conceptual and can be used to highlight important connections in real world systems or processes, while quantitative models can be used to solve relevant equations of a system or may characterize a system (Seidewitz 2003). Quantitative models often go beyond qualitative approaches to make quantitative predictions about the reactions of fish populations to different management decisions (Hilborn and Walters 1992). Actual fishery management systems may be considered as models used to aid in the design of fishery management systems in similar contexts. For example, the fishery for rock lobster (*Jasus edwardsii*) in New Zealand, discussed in detail below, employs quantitative stock assessments and reference points to specify management actions in response to changes in stock abundance. Quantitative computer simulations are used to test different management regimes and reference points in the fishery. The New Zealand rock lobster fishery is itself a model composed of qualitative and quantitative sub-models. Qualitative models used in the fishery are based on stakeholder involvement in decision making and resource allocation, which aid in understanding of the importance of specific institutional criteria or specific management systems regarding fishery management performance.

For this study, a selection of KS^n fisheries were chosen as candidate 'model fisheries'. Through a review and synthesis of specific elements of the candidate models, particular model components were assessed regarding their effect on the sustainability of each fishery. Sustainability criteria used to assess each fishery were derived from the fisheries literature. General features of the sustainability criteria were related to the spatial scale of management, resource access and allocation, resource monitoring, the decision-making structure of the fishery, and assessment of managed stocks. Model fisheries incorporating most of the sustainability criteria were relatively sustainable/successful. Several practical recommendations were provided for the IR fishery and an emphasis was placed on resource access and allocation.

2.2. Model fishery selection

Model fisheries that were chosen and their locations are presented in Table 1. The KS^n fisheries classification described earlier was used as a guide to identify candidate model fisheries. Specific organism characteristics that are common to these fisheries include:

(i) benthic following juvenile stage;

(ii) broadcast spawners (pelagic larval propagules must remain suspended in the water column for a period of at least five days);

(iii) have a maximum reported age greater than 30 years;

(iv) susceptible to barotrauma;

(v) form sub-populations that are presumed to compose a metapopulation;

(vi) inhabit depths less than approximately 300 meters;

(vii) relatively sedentary, which I defined as an average adult home-range less than approximately $500m^2$; and

(viii) not fished using trawling gear.

Clearly, the model fisheries I considered in this study were not chosen randomly, and therefore the results of this study may be biased toward fisheries in relatively developed nations with a significant amount of published information.

Because they share a significant similarity with the IR fishery, some fisheries were included in this study even though they did not meet all life-history and spatial characteristics selection criteria. For example, neither invertebrate fisheries nor the fishery for pink snapper (*Pagrus auratus*) in inner Shark Bay, Western Australia, suffer barotrauma, and it is unknown if goliath grouper (*Epinephelus itajara*) in the Gulf of Mexico form geographically-isolated populations (Jackson et al. 2005; SEDAR 2004).

2.3. Conceptual framework methodology

Bardach (2004) coined the term "conceptual framework" to describe methods similar to those used here. Conceptual frameworks are useful when a more appropriate methodology is not available and when a strictly quantitative comparison between groups is impractical (Bardach 2004). I developed seven criteria for judging fisheries management sustainability based on existing literature. KS^n fisheries that are potentially sustainable over the long-term should have similar properties. Analyzing the model fisheries using the seven criteria was determined to be the most applicable methodology available to account for uncertainty and avoid the assumption of replication of conditions between model fisheries, as suggested in Bardach et al. (2004). For instance, it would be impractical to assume that the stakeholder access and oceanic conditions were consistent between all of the reviewed KS^n fisheries. As such, management measures employed in Australia that benefit from the oceanic conditions in that area may not be suitable in British Columbia.

2.3.1. Sustainability criteria

Hilborn et al. (2003a, 2005) hypothesize that fisheries management success depends on three criteria:

(i) the spatial scale of management must match the spatial scale of the biology and population dynamics of the resource;

(ii) the resource access and allocation methods must create incentives for sustainability; and

(iii) the decision-making structure of the institutions must be transparent and management must devolve from a single, central controlling body.

Clearly, the first criterion requires detailed knowledge of the biology of the targeted fish species, while the latter two favour the use of conservation incentives and open institutional structure.

To improve stakeholder communication and management of fisheries, de la Mare (1998) suggests that modern fisheries management is in need of a new approach that emphasizes the management rather than the biological problems of fisheries. His management oriented paradigm (MOP) is based on four criteria (with numbering continued from above):

(iv) a set of measurable and operational management objectives;

(v) a management procedure based on decision rules (criteria for making decisions and the complete set of decisions that can be made);

(vi) assessments based on specified data and methods, such that a scientific judgement on the state of a resource may be determined; and

(vii) a prospective evaluation of the management procedure using performance measures (measures of management success or failure), such that management procedures may be examined through simulation before they are put into place.

The criteria described by de la Mare (1998) require more biological information than the first three criteria, and are based more upon computer simulation than resource access and decision-making structure. However, developing measurable management objectives (the fourth criterion) inherently requires involvement by decision-makers. For instance, fishery stakeholders are required to work together to form management objectives which are necessary for the fourth criterion.

In general, Hilborn et al. (2003a, 2005) focuses upon relatively broad social and institutional structure while de la Mare (1998) focuses primarily on operational structure. As a result, the theory presented in Hilborn et al. (2003a, 2005) and de la Mare (1998) complement each other, rather than overlap.

The following sections describe how each of the management criteria promotes fishery sustainability, particularly in the case of KS^n fisheries.

Criterion 1: Spatial scale of management

As described above, effective management of KS^n species may depend on an adequate match between the spatial scale of management and the spatial scale of sedentary populations (i.e., the scale of relatively self-recruiting units of stock) (Prince 2005).

Hilborn et al. (2005) define the spatial scale of management as the spatial scale at which regulations are set, data is collected, and science is conducted. If a tyranny of scale exists in a KS^n fishery, a locally or serially depleted stock may be replenished by either very limited adult or juvenile movement or by the dispersion of larval propagules, since based on propagule duration, initial development, and hydrodynamic processes at the time and place of spawning, stock-recruitment dynamics potentially occur at scales larger than the scale of relatively sedentary adults (Alverez et al. 2001). The dispersal of larval propagules affects populations of marine organisms by both sustaining populations with new recruits and maintaining genetic continuity or gene flow between sub-populations (Shanks et al. 2003), and the understanding of sub-population connectivity through larval propagule dispersion is a conservation concern in KS^n fisheries because reproductive adults are generally sedentary (Buonaccorsi et al. 2005). Therefore, the dispersion of larval propagules was examined in each model fishery to determine if the spatial scale of management was sufficient.

Criterion 2: Resource access and allocation

Fisher incentive for resource sustainability requires two crucial elements. First, fishers must be provided with harvesting or territorial rights to fish, with particular emphasis on long-term and secure rights (Hannesson 2004). Second, access rights must be enforced to protect the value of the assets and encourage a sustainable flow of benefits from a fishery (Grafton et al. 2006). A tragedy of the commons may be avoided if fishers are granted a level of ownership or exclusive access to a resource because fishers may be motivated to pursue sustainable management of the resource (e.g., Hilborn et al. 2003a, 2005; Maguire 2003; Grafton et al. 2006).

Fishing rights may take several forms based on the level of resource exclusion and the size or composition of the bodies holding the rights (Grafton et al. 2006). Access rights range from open access (the least exclusive) to exclusive access (e.g., Hilborn et al. 2005). In open access fisheries, all individuals who desire to do so may go fishing. In limited entry fisheries, access is slightly more constrained, and license availability typically limits access. The next level of exclusivity is a form of individual or group quota (e.g., catch or effort quotas), where each fishery participant is assigned a percentage of the total quota. Individual vessel quotas (IVQs) are a form of individual quota (IQ), typically expressed as an individual share of an aggregate quota or TAC. The most exclusive access exists in fisheries with territorial fishing areas, where individuals or organizations are essentially assigned a level of ownership of the resource (Hilborn et al. 2005).

Many model fisheries had multiple user-groups with varying user-rights competing for the same resource. For instance, the fishery for pink snapper in Inner Shark Bay, Western

Australia, has substantial recreational and commercial fishing sectors, and both user-groups are regulated with a TAC (Jackson et al. 2005). In comparison, the sea urchin fishery in British Columbia, Canada, is harvested almost exclusively by the commercial sector (Campbell et al. 1999; DFO 2006a). Therefore, it is intuitive that the fishery (whether recreational or commercial) with the assigned rights must be the dominant user-group for a form of exclusive access rights to be effective. As a general guideline, a user group was considered dominant if it accounts for approximately 90% or more of annual landings. For this reason, it was essential to consider the level of recreational catch when reviewing the presence of exclusive user-rights in each fishery. Some fisheries had recreational sectors that accounted for more than 10% of annual landings. If user-rights were specified only for a single user-group in a multiple user-group fishery, the effectiveness of assigning of user-rights would obviously be limited.

Criterion 3: Decision-making structure

Co-operative management (co-management) is the sharing of power and responsibility between the state and resource user-groups when managing natural resources (Pinkerton 1989). Such an arrangement results in the devolution of management because local resource users, stakeholders, external agents, and management authorities work together in the decision-making process (Pinkerton 1989, 1992, 1994; Pomeroy and Rivera-Guieb 2005). Therefore, the level of co-management in each fishery was used as an indicator for the decision-making structure criterion.

For KS^n fisheries it is assumed that fishery sustainability increases with the devolution of management because it increases both fisher participation in decision making and transparency in the decision making structure. Devolution of management may be particularly effective in small-scale fisheries, where top-down, centralized systems of management have been regarded as ineffective (Hilborn et al. 2005). Complex and non-transparent fishery management systems often cause ineffective fishery management and may result in overfishing (Healey and Hennessey 1998).

Criterion 4: Measurable management objectives

The importance of both clearly stated fishery management objectives (e.g., Shepard 1991; World Bank et al. 1992; Pido 1995; Berkes et al. 2001) and *measurable* fishery management objectives have been emphasized in the literature (e.g., Barber and Taylor 1990; Francis and Shotton 1996; Murawski 2000; Sainsbury 2000). When fishers have common objectives they become an accountable partner and are actively involved in decision-making regarding allowable harvest levels and allocation of fish among users. Also, fishers gain a sufficient understanding of how fishery management decisions are made, and make difficult decisions regarding fishery management (Christy 1982; Lane and Stephenson 2000; Berkes et al. 2001; Hilborn et al. 2003a, 2005; Grafton et al. 2006).

Management objectives operationally support management goals and as such should be measurable and verifiable statements (Barber and Taylor 1990). There are typically two groupings of fishery management objectives. The first are biological objectives (Clark 1985), such as a particular spawning stock size, regarding the biological sustainability of the resource. The second are economic objectives, which relate to efficiency (optimization of economic returns to the fishery) and equity (the distribution of economic benefits). Both groupings of management objectives may be further categorized as either aspirational or operational. Aspirational management objectives, or ‘goals’ (Barber and Taylor 1990) are akin to mission statements; for example ‘to maximize economic benefits from a fishery’, ‘to stabilize stock levels’, or ‘to provide employment’. Conversely, operational management objectives include calculable elements such as the probability that a stock will rebuild to a specified size within an agreed-upon period of time (de la Mare 1998).

The clarity and measurability of management objectives were reviewed for each fishery to gauge the presence of measurable management objectives. Clarity of management objectives was assessed based on how the management objectives were determined and whether the management objectives were specific to each managed stock. Measurability of management objectives was assessed based on whether the objectives were aspirational or measurable.

Criterion 5: Management procedure based on decision rules

According to de la Mare (1998), fisheries management policy requires the advance specification of all management actions that should be taken in all circumstances. A management procedure avoids potential gaps in policy by specifying a feedback control system consisting of a set of decision rules to set, remove, or vary management regulations in response to changes in stock status (Butterworth et al. 1997; Cochrane et al. 1998; de la Mare 1998; Butterworth and Punt 1999; McAllister et al. 1999). Management procedures further specify what data will be collected, how the data will be collected and processed, what estimates will be made from the data, and how the estimates will determine harvest controls (Bentley et al. 2005). Also, a management procedure involves weighting multiple management criteria and requires the specification of measurable management objectives (de la Mare 1998).

Harvest control rules are often used in fisheries management to specify catch quotas or fishing intensity in terms of some other variable regarding the status of a stock, such an index of spawning biomass (Restrepo and Powers 1999). Such rules are typically designed to be precautionary (Rosenberg et al. 1994) and thus represent an essential component of a management procedure (de la Mare 1998). Stock threshold levels, or reference points, include lower limit reference points (LRPs) to set "boundaries which are intended to constrain harvesting within safe biological limits within which the stocks can produce maximum sustainable yield" and target reference points which "are intended to meet long-term management objectives" (UN 1995). As such, a LRP may be considered a form of decision rule to dictate management action based on an index of stock size or health. The specification of measurable and operationally-unambiguous definitions of LRPs is used as a precautionary measure in fishery management to avoid overfishing (Rosenberg et al. 1994; Caddy 2004) and to meet specific fishery goals (Hilden 1993; Leaman 1993; Rivard and Maguire 1993).

Criterion 6: Assessment based on specific data and methods

In a MOP, assessments require the collection of specific data and parameter estimates by agreed-upon methods (de la Mare 1998). An assessment is a scientific judgement to advise a management authority of the state of a resource (de la Mare 1998). Methods and frequency of assessments vary, as do data requirements for assessment methods (Hilborn and Walters 1992).

To review each fishery with regards to Criterion 6, both the presence and frequency of assessments were considered. If a fishery did not have an assessment, I assumed that a managing authority could not adequately be advised because the state of the resource could not be determined (de la Mare 1998). Frequency of assessment was used as a surrogate measure of specificity of data and parameter estimates by agreed-upon methods because no adequate method was found to review whether each fishery used assessments based on specific data and parameter estimates and agreed-upon methods. Because an assessment determines the state of a resource, it is assumed that more frequent assessments (e.g., annually rather than every four years) provide more readily available assessment results, which would in turn more accurately index stock status, leading to a lower risk of overfishing in KS^n fisheries.

Criterion 7: A prospective evaluation of the management procedure using performance measures

Prospective evaluation (i.e., review prior to implementation) of a management procedure is achieved through a simulated feedback system, an iterative process involving computer simulation consisting of evaluation of the proposed decision rules (i.e., control rules). The results of the evaluation can then be presented to fishery stakeholders and decision-makers, the control rules can be revised based on this discussion, and the control rules can be re-evaluated

(McAllister et al. 1999). A prospective evaluation is similar to a management strategy evaluation (Hilborn 1979), a decision analysis framework, or a harvest strategy evaluation. A prospective evaluation includes both a prospective component and performance measures, derived from measurable management objectives, to present outcomes and demonstrate the likelihood that the management system will meet its objectives (de la Mare 1998).

Specific measures of sustainability/success, both biological and economic, are difficult to determine for a fishery (Hilborn 2003b). Determining the biological and economic sustainability of KS^n fisheries is particularly challenging due to a paucity of data and often unreliable, or non-existent, estimates of stock size. Hilborn et al. (2003b) measured the biological health of several fisheries using current biomass in relation to biomass at maximum sustainable yield (MSY), and economic health as current catch in terms of the long-term maximum. Mace (2004) used magnitudes of increase or decrease in fishing mortality and biomass to assess biological status of several marine organisms. However, the efficacy of this study was not limited by the absence of a specific measure of success because understanding the sustainability of each fishery was accomplished using a conceptual framework that is centred upon the presence of the criteria rather than strict quantitative analysis of each fishery (Bardach 2004).

2.4. Results

Tables 2-6 summarize fishery status, management, and stock assessment of the model fisheries. Each table is explained in detail in the following section in the context of the seven criteria. The pre-closure management regime in the Gulf of Mexico Goliath Grouper (GMGG) and California Bay Cowcod (CBC) fisheries were evaluated in this study; the fisheries closed in 1990 and 2001, respectively. The remaining 11 fisheries remain open and their most recent management regime was evaluated.

Table 2 summarizes the historic and present biological status of each fishery. Documented historical overfishing occurred in eight fisheries and overfishing is uncertain, but presumed, in two more fisheries. The only fisheries without documented overfishing were the Demersal Shelf Rockfish (DSR), British Columbia Sea Urchin (BCSU), and British Columbia Geoduck (BCGD) fisheries.

Seven of the 13 fisheries lacked stock assessment data and therefore lacked quantitative estimates of stock status. In place of quantitative estimates of stock status, the qualitative status of each fishery was recorded (Table 2). The majority of fisheries with unknown quantitative estimates of stock status were documented as overfished or had evidence of excessive fishing mortality. For example, a quantitative estimate of stock status was unavailable for the Western Australian Dhufish (WAD) fishery, but there was evidence that current fishing mortality was greater than natural mortality, a level of fishing pressure that was determined to be 'unacceptable' and which 'may not be sustainable' (Hesp et al. 2002; St. John and King 2004). In general, there has been a trend of increasing stock sizes and 'recovering' of stocks over time in most of the fisheries.

Table 3 summarizes the major regulatory strategies employed in each fishery. Specifically, Table 3 lists the harvest strategy and method of implementation, reference points, and quota allocation scheme/access structure of the dominant user group in each fishery. Harvest strategies varied from input control of fishing intensity to output control in the form of constant catch levels. Eight fisheries employed constant fishing mortality to specify a form of TAC. The Western Australian Rock Lobster (WARL) fishery was unique in that total effort was the limiting factor in the fishery, from which individual fisher effort levels were specified. Some fisheries employed multiple regulatory strategies. For example, the New Zealand Rock Lobster (NZRL) fishery used area-specific (spatial) harvest control rules to specify TAC, along with reference points in each management area.

Table 3 also lists whether each fishery employed reference points. Nine fisheries instituted measures such as precautionary reference points to control stock exploitation at times of low stock level or uncertain stock status. Only the GMGG, CBC, IR, and Chilean Loco (CL) fisheries lacked any form of reference point. Reference points are related to decision rules and as such are covered in detail under Criterion 5, below.

Finally, Table 3 lists the quota allocation scheme/access structure of the dominant user-group of each fishery, covered in detail under Criterion 2, below. Five fisheries employed limited entry, four employed individual transferrable quotas, two employed individual quotas, and only the Gulf of Mexico Goliath Grouper (GMGG) and CBC fisheries had no quota allocation scheme for the dominant user-group.

The following section describes in detail the degree to which each of the seven criteria are incorporated into each fishery, summarized in Tables 4, 5, and 6. Table 7 is interpreted in the Discussion section below, and considers whether each criterion is applicable in the IR fishery and whether meeting the criteria has resulted in relative sustainability of each fishery.

Criterion 1: Spatial scale of management

Most KS^n fisheries are data-poor, and as a result the degree of biocomplexity and stock structure are relatively poorly understood and defined (Table 4). Because of a paucity of data in KS^n fisheries, the ability to gauge each fishery regarding Criteria 1 was limited. It is presumed that the CL and VAB fisheries are managed at the scale of the sub-population because most harvested bed areas are managed individually. Because of either a lack of data or management areas larger than the biological scale over which the managed population is thought to function, it was unknown if any of the remaining fisheries had a sufficient spatial scale of management.

Information regarding the distance travelled by marine larval propagules is rare (Shanks et al. 2003); however, Shanks et al. (2003) found a significant positive correlation between duration of the dispersal stage and dispersal distance for a range of benthic marine organisms. Along with a general lack of data, the movement of larval propagules in each fishery was uncertain. For this reason, it was practical to assume that some model fisheries were not managed at the scale of the sub-population. For instance, larvae may travel up to 1000 km. if suspended in the water column for a period of approximately one month (Shanks et al. 2003). However, the distance travelled is highly dependant on oceanographic conditions which are for the most part uncertain and difficult to predict (Yamanaka et al. 2004 for inshore rockfish). Therefore, avoidance of overfishing may have resulted from general features of each fishery such as relatively high rates of movement of some species, the presence of data regarding migration and movement patters, low fishing intensity, etc. Each of these features is discussed in detail in the Discussion section, below. Because most KS^n fisheries were data-poor and target species with limited mobility, it is intuitive that smaller management areas would be more advantageous.

Criterion 2: Resource access and allocation

With the exception of the NZRL fishery, sustainable fisheries were regulated by catch or effort quotas held by a single dominant user-group, or employed a form of catch quota for each user-group in the fishery (Table 4). The majority of sustainable fisheries met criterion two (Table 6). The NZRL fishery was the only sustainable fishery that had an open-access recreational sector (Bentley et al. 2005).

The WARL, BCSU, BCGD, Victorian Abalone (VAB), and CL fisheries are regulated by catch or effort quotas held by a single dominant user-group (Table 4). The Shark Bay Pink Snapper (SBPS) and the Victorian Rock Lobster (VRL) fisheries had multiple user-groups and employed a form of catch quota for each user-group in the fishery. An exception is the commercial DSR fishery which begins only when, and if, the proceeding annual recreational fishery catch has been accounted for (GOA FMP 2005). Both the IR and NZRL fisheries employed a form of catch quota; however, because of relatively large recreational sectors, neither fishery had a single dominant user-group. The remaining model fisheries did not have a dominant user-group or quotas regulating catch or effort for each fishing sector.

Interestingly, some fisheries were targeted by a dominant user-group in the absence of a management directive to limit users of the resource. It was assumed that some fisheries had limited multi-user (e.g., recreational along with commercial) potential due solely to issues of feasibility or desirability. For example, a fishery which targets a species in deep water and far from a metropolitan area would have a lower potential for recreational access.

Criterion 3: Decision-making structure

The majority of sustainable fisheries met criterion three (Table 6). The VRL, BCSU, VAB, and CL fisheries had explicit documentation of co-management arrangements, and were relatively sustainable (Table 4). Other sustainable fisheries, such as the NZRL, included important elements of co-management without explicit documentation of co-management.

The level of co-management employed in each fishery ranged from the relatively informal presence of particular elements of co-management (e.g., the IR fishery) to the formal, explicit documentation of co-management arrangements (e.g., the VRL and CL fisheries). For example, co-management is not required under Canadian federal or provincial law. Conversely, the 1995 Fisheries Act in Victoria, Australia (Anon 2003) and the 1991 Fisheries and Aquaculture Law in Chile (Gelcich et al. 2007) both require co-management as an essential component of fisheries management. Only the GMGG and CBC lacked references to co-management.

The NZRL fishery has an element of co-management despite a legal mandate to do so. For example, management objectives for the fishery were determined by the New Zealand Rock Lobster Management Group, representing all stakeholders in the fishery.

Some model fisheries employed co-management only recently, which limited inferences about how co-management structure influences fishery sustainability. For example, co-management of the WAD fishery, which began in 2003, involved a Management Planning Panel and a Commercial Access Panel that were both subject to a four-month public comment period. The Commercial Access Panel was tasked with determining user access and level of allocation to the state's wetline fishery stocks, such as the equitable allocation of the total quota, while the Management Planning Panel developed management arrangements for each stock in the region

(St. John et al. 2004). The co-management process for the VRL fishery, which was not formalized until 2003, was developed through consultation with license holders, fishery managers, and fishery officers (Anon 2003).

Criterion 4: Measurable management objectives

The majority of sustainable fisheries met criterion four (Table 6). The SBPS, NZRL, and WARL fisheries had clear and quantifiable management objectives and were relatively sustainable (Table 5). The VRL, BCGD, VAB, and CL fisheries had clearly stated and measurable means with which to work toward non-measurable management objectives, and were also relatively sustainable.

The management objective of the WARL fishery was that “management arrangements adopted would ensure that the abundance of breeding lobsters is maintained at or above the levels in the late 1970s/early 1980s (i.e., about 20-25 per cent of the unfished parental biomass)” (RLIAC 1999). The estimated 1980 level of biomass (approx. 22% of unfished parental biomass) was assumed sustainable and was therefore chosen as a lower limit that breeding stocks are to remain above under future management (de Lestang and Melville-Smith 2004). The management objective was not specific to any of the three stocks in the fishery and, similar to the SBPS fishery, the management objective did not specify a timeframe for rebuilding. At present, all three stocks were close to maximum sustainable yield and breeding stock was at or above target levels in each of the three management areas (de Lestang and Melville-Smith 2004).

A total of six management objectives were employed in the NZRL fishery. These management objectives were used to develop candidate harvest control rules for each of ten management areas (NRLMG 2005), suggesting that spatial scale of management may be adequate.

Jackson et al. (2005) conducted a study of some of the most popular recreational marine fisheries in Australia and found that 'clear and quantifiable' management objectives were only present in the SBPS fishery.

The VRL, BCGD, VAB, and CL fisheries lacked measurable management objectives, but employed relatively clearly stated and measurable means with which to work toward management objectives. For this reason, the clarity and measurability of the means to work toward the management objectives were assessed in these fisheries. For example, one of the management objectives the VRL fishery was the 'sustainability of the rock lobster resource' (Anon 2003). To reach this goal, strategies such as the implementation of a lower reference point based on spawning level were employed.

The remaining fisheries employed clearly stated, yet non-measurable objectives. For example, in the BCSU fishery, the 'collection of biological information' is listed as one of the primary management objectives, to better understand growth and recruitment parameters of the resource (DFO 2006c). The only exception was the WAD fishery, for which no management objectives were stated.

Criterion 5: Management procedure based on decision rules

The majority of sustainable fisheries met criterion five (Table 6). Fisheries that incorporated a specific management procedure based on decision rules were relatively sustainable, although all fisheries incorporated some form of decision rule (Table 5). Fisheries with relatively dynamic decision rules, where the TAC was updated annually for example, were relatively sustainable, and fisheries that did not employ reference points were considered relatively unsustainable.

The NZRL fishery employed a management procedure, decision rules, and limit reference points most closely resembling those required in a MOP (Bentley et al. 2005). For instance, the NZRL fishery used area-specific decision rules to either specify TAC based on simulation trials that quantified the probability of rebuilding biomass to a pre-specified level, or to mandate an assessment of the stock based on CPUE data. Decision rules were applied within the context of a management procedure and were tested within a simulated feedback system (Starr et al. 1997; Bentley et al. 2003; Breen et al. 2003). In the NZRL fishery, measurable performance indicators were associated with each of six management objectives. For instance, both mean and median annual catch and the probability of falling below the current TAC were used as performance indicators to achieve the management objective of maximizing catch.

Reference points used in each fishery are summarized in Table 3. Only the GMGG, CBC, IR, and CL fisheries lacked reference points. Of these four fisheries, only the CL fishery was sustainable. The most common reference point was a limit reference point based on spawning biomass, below which fishing was prohibited.

The majority of fisheries employed both decision rules and limit reference points in the absence of a fully-specified management procedure (i.e., a level of fishing intensity or other management action is pre-determined at all estimated stock levels). Decision rules to determine TAC or total allowable effort in the SBPS, WAD, GMGG, CBC, IR, WARL, VRL, and BCSU fisheries were relatively static (e.g., TAC was set at intervals longer than annually). For example, the decision rule used in the WAD fishery was a proposed target catch range based on the average catch from 1990/1991 to 1999/2000 (St. John et al. 2004), while the decision rule used in the BCSU fishery was to calculate TAC every two years as a function of a conservative estimate of natural mortality and the current estimated total biomass of sea urchins (DFO 2006a).

Conversely, decision rules to determine TAC or total allowable effort in the DSR, NZRL, VAB, and BCGD fisheries were relatively dynamic (i.e., TAC was updated annually). For example, TAC in the DSR fishery was calculated annually as the product of a function of natural mortality and current estimate of adult spawning biomass.

Annually updating the TAC was not necessarily required for fishery success. For example, although the stock assessment model in the VAB fishery was updated annually, and thus allowed for annual variation in TAC, the actual TAC has been constant since quotas were first introduced in 1988. At present, the fishery is assessed as stable and fully-fished, and is therefore considered relatively sustainable because it is not overfished (DNR 2002).

Criterion 6: Assessment based on specific data and methods

Fisheries which conducted annual assessments were relatively sustainable, although conducting annual assessments was not a requirement for fishery sustainability (Tables 2 and 5). For example, the BCSU fishery conducted assessments every two years and was relatively sustainable. Fisheries that conducted assessments at irregular intervals or that did not conduct assessments were for the most part unsustainable. Fisheries with relatively complex stock assessments were sustainable. The CL fishery was the only fishery which required individual fishers to hire consultants to assess their fishing grounds, a measure which seems to work quite well. The majority of sustainable fisheries met criterion six (Table 6).

With the exception of the GMGG and CBC, all fisheries conducted periodic stock assessments. The SBPS, DSR, NZRL, WARL, VRL, BCGD, VAB, and CL fisheries conducted annual assessments, while the WAD and IR fisheries conducted assessment at irregular intervals. The BCSU fishery conducted assessments every two years. There were no assessments of the GMGG fishery before closure in 1990 (SEDAR 2004), and the first assessment of CBC was conducted one year before ‘no-retention’ management was initiated (Butler et al. 1999).

Methods used to conduct assessments varied among model fisheries. For example, the NZRL fishery conducted relatively complex and data-intensive Bayesian length-based stock assessments to simultaneously estimate recruitment, mortality, growth, maturity, selectivity, and seasonal vulnerability parameters (Starr et al. 2003; Bentley et al. 2005). To address the spatial structure of the managed stocks, assessment and assessment precision in the NZRL fishery varied by management area and a ‘rebuild trajectory’ (permitted CPUE over time based on stock size) specified rates of rebuilding in each area within a required timeframe (NRLMG 2005).

The daily egg production method was used for annual assessment of the size of snapper spawning biomass in the SBPS fishery, and due to large variances around estimates of egg production, DEPM estimates of snapper spawning biomass have often been imprecise (Jackson et al. 2005). Age-structured models were used in 2003 to refine the management strategy and to provide a more formal assessment of stock status (Jackson et al. 2005). The management objective in Shark Bay was to rebuild all three pink snapper stocks to $B_{40\%}$ (i.e., 40% of estimated unfished biomass) (Jackson et al. 2005). As of 2005, the last year published, two of the three stocks were above the $B_{40\%}$ objective (Jackson et al. 2005).

Stock assessment in the WARL fishery utilizes a range of fishery-dependent and fishery-independent data (de Lestang and Melville-Smith 2004). For example, commercial catch records from logbooks and fishery-independent monitoring of larvae settlement and breeding stock levels were all used in stock assessment (de Lestang and Melville-Smith 2004).

Assessments were conducted in the BCSU fishery to determine total current biomass based on bed area (DFO 2006a). A modified surplus production model was used to estimate MSY for the fishery based on surveys that estimate urchin density and changes to the commercial bed area (DFO 2006a).

The first and only assessment of the WAD fishery was conducted in 2002, and commercial catch rates and recreational creel surveys were the only data from which estimates of relative abundance could be determined (Hesp et al. 2002; St. John et al. 2004). In fact, the status of the resource is uncertain, and the dhufish stock may not be able to sustain current catch levels (Hesp et al. 2002).

A novel approach to annual assessments in the CL fishery requires that artisanal fishers finance studies of their fishing grounds, work to establish area-specific management plans, and contract external consultants to annually assess resources to determine annual changes in TAC. The annual assessments are presented to a central authority for review (Gelcich et al. 2007).

Criterion 7: A prospective evaluation of the management procedure using performance measures

The NZRL fishery was relatively sustainable and was the only fishery that employed a prospective evaluation of a management procedure using performance measures, employing extensive simulation trials testing various management procedures, harvest control rules, and performance measures to identify procedures that would rebuild biomass to a target level in a specified period of time; an approach which has proven successful (Bentley et al. 2005).

However, six other sustainable fisheries employed a form of prospective evaluation of their management strategy in the absence of a prospective evaluation of a management procedure using performance measures, as specified by de la Mare (1998). The majority of the remaining fisheries, which did not employ any form of prospective evaluation of their management strategy, were considered less sustainable (Table 6).

In the absence of a management procedure, the presence of a prospective evaluation, to either assess management strategies or to revise stock parameters, was reviewed for each fishery. Along with the NZRL fishery, the SBPS, WARL, VRL, BCGD, VAB, and CL fisheries employed a prospective evaluation of their management strategies. In the SBPS fishery, age-structured models were used to explore likely trajectories of mature biomass for a range of future catches (Jackson et al. 2005). In the WARL fishery, prospective modeling was used to assess stock sustainability and to forecast future catch levels (de Lestang and Melville-Smith 2004). In the BCGD fishery, age-structured projection modeling was used to assess the impact of various harvest rates on the fishery (Zhang and Hand 2006). In the VRL fishery, a prospective evaluation was used to examine the impact of alternate harvest strategies and various stock parameters on both spawning and available biomass (Anon 2003). In the CL fishery, models regarding fisher decision-making have recently been employed to determine harvesting decisions, and changes in TAC are assessed annually (Gelcich et al. 2007). The remaining six model fisheries (WAD, DSR, GMGG, CBC, IR, and BCSU) did not employ any form of prospective evaluation of their management strategy.

2.5. Discussion

The goal of this study was to identify a fishery management system that may potentially provide a sustainable IR fishery in British Columbia, despite the major challenges and risks associated with KS^n fisheries. A collection of model KS^n fisheries were identified as relatively sustainable/successful (Table 3), and the degree to which each fishery met seven criteria of fishery management success was analyzed. Several fisheries were identified as relatively sustainable and incorporated each of the seven criteria to higher degrees than the less sustainable fisheries (Table 6). With the exception of the first criterion, the majority of fisheries that met the criteria were relatively successful (Table 6). Accordingly, the management systems of these fisheries tended to promote sustainability and should be examined to determine if they could provide a model for a sustainable IR fishery.

The qualitative and quantitative stock status of each fishery (Table 2) was used as a proxy for fishery sustainability due to the lack of measurable management objectives in all but three fisheries. Fisheries closed due to overfishing were considered unsustainable. If neither the qualitative nor quantitative status of a fishery were known, it was assumed the fishery was relatively unsustainable due to high uncertainty in stock status. The WARL fishery has been certified under the Marine Stewardship Council (MSC 2008), and is therefore considered sustainable. Sustainable fisheries included the SBPS, DSR, NZRL, WARL, VRL, BCSU, BCGD, VAB, and CL fisheries. Conversely, fisheries that were currently closed or were otherwise considered relatively unsustainable were the WAD, GMGG, CBC, and IR fisheries. Some fisheries, such as the VRL fishery, were rebuilding under the current management regime from previous overfishing, and were considered relatively sustainable because the status of stocks were being tracked closely and the stocks were actively recovering (Hobday et al. 2005).

Correlations between the seven criteria and sustainable model fisheries are described below and are summarized in Table 6.

The importance of managing the impacts of fishing at spatial scales that match the biological scale over which populations function is emphasized by Ludwig et al. (1993), Carvalho and Hauser (1994), Botsford et al. (1997), Orensanz and Jamieson (1998), and Hilborn et al. (2003a/b); however, it was challenging to determine scale matching the majority of the fisheries because of varying oceanographic conditions and uncertain life-history characteristics (with the exception of the VAB and CL fisheries). Also, similar to S-fisheries (Parma 2001), the KS^n fisheries considered in this study were in general data-poor and as a result, the degree of biocomplexity and stock boundaries were poorly understood and defined. For this reason it was assumed that local and potentially serial depletion may result in the fisheries if the spatial scale of management is not set correctly, potentially reducing biocomplexity and thus the resiliency of the stock (Holling and Meffe 1996; Hilborn et al. 2003b). Because most KS^n fisheries were data-poor, it is intuitive that smaller management areas would be more advantageous.

The first criterion was significantly different from the other criterion in that the majority of fisheries were relatively successful yet apparently did not meet the criterion. Therefore, avoidance of overfishing may have resulted from the following general features of each fishery:

(i) adequate specification of stock boundaries, permitting assessment;

(ii) relatively high rates of movement of some species (e.g., rock lobster) to recolonize depleted areas, thus lessening the consequence of localized depletion;

(iii) the presence of data regarding migration and movement patterns;

(iv) no mortality of the targeted species due to barotrauma, thus permitting in-situ monitoring;

(v) the presence of a dominant user-group to increase fisher incentive for sustainability; and

(vi) low fishing intensity.

This would explain why seven relatively sustainable fisheries did not meet the first criterion (Table 6).

The presence of catch quotas for each major user-group was related to sustainability in KS^n fisheries, and fisheries with higher levels of resource exclusion were more sustainable (Table 6). Assigning catch quotas to each user-group is a form of exclusive access which provides motivation for sustainable resource use (Hilborn et al. 2003a, 2005; Maguire 2003; Grafton et al. 2006) and works to decrease a tragedy of the commons (Hardin 1968; Hilborn et al. 2005). The NZRL fishery was the only sustainable fishery that did not have a designated quota for all fishing sectors; sustainability in the NZRL fishery results from the presence of other criteria, discussed below.

Fisheries with devolved and less complex decision-making structures were relatively sustainable (Table 6). A notable exception was the NZRL fishery which employed relatively complex stock assessment techniques together with important elements of co-management. The presence of co-management was used to determine the level of decision-making in each fishery since stakeholders work together in decision-making processes (Pinkerton 1989, 1992, 1994; Pomeroy and Rivera-Guieb 2005). It was challenging to determine the level of co-management present in each fishery. For instance, when conducting the literature review it was often unclear whether stakeholders were actively involved in management decisions (i.e., ‘true’ co-management), or if stakeholder input was used solely to help guide management decisions. For this reason a fishery was presumed to meet the third criterion if the literature implied that stakeholders were actively involved in decision making.

Clear and quantifiable management objectives were only somewhat correlated with sustainability in the fisheries (Table 6). However, the three model fisheries with clear and quantifiable management objectives were relatively sustainable. Many fisheries employed management objectives that were clearly stated yet non-measurable, and were relatively sustainable. The only fishery without any documented management objective was the WAD fishery, which was considered relatively unsustainable.

Due to a general paucity of data in all other model fisheries, only the NZRL fishery employed a management procedure based on decision rules as specified in a MOP (de la Mare, 1998). However, all fisheries incorporated some form of decision rule (a crucial element of a MOP) to set, remove, or vary management regulations (as specified in Butterworth et al. 1997; Cochrane et al. 1998; de la Mare 1998; Butterworth and Punt 1999; McAllister et al. 1999). The three fisheries that did not employ a lower reference point were considered relatively unsustainable (Table 6).

Fisheries with frequent, annual assessments were more sustainable that fisheries with infrequent or non-existent assessments (Table 6), and fisheries with relatively complex assessments were more sustainable than fisheries with basic assessments. For example, age-structured models were used in the relatively sustainable SBPS fishery to determine stock status and to work towards management objectives in each management area, and the NZRL fishery employs complex Bayesian methods for assessments in each of several management areas. Conversely, the relatively unsustainable WAD fishery has only recently began conducting assessments and uses only CPUE data in a single, large management area.

A prospective evaluation of a management procedure using performance measures as specified by de la Mare (1998) was present only in the NZRL fishery. However, several other model fisheries employed computer simulation to prospectively evaluate their harvest strategies, which was related to sustainability in these fisheries (Table 6).

Several challenges were evident in this study, most notably the presence of confounding (i.e., non-independence) of criteria. For example, if a fishery did not meet the specifications of the fourth criterion, that is clear and quantifiable management objectives, the fishery could not have a management procedure as specified in de la Mare (1998), which was by definition composed of clear and quantifiable management objectives. Therefore, the presence of some successive criteria are inherently dependant upon some prior criteria. This is evident in the methods section above, as several surrogate measures were developed to address this challenge.

The DSR fishery was unlike the other fisheries in several ways because it incorporated the seven criteria to a relatively low extent, yet was considered relatively sustainable. This is most likely because of the unique access structure of the DSR fishery. Specifically, the commercial DSR fishery only occurs following the full accounting of all DSR catch in all other fisheries that target or incidentally catch DSR. The commercial DSR fishery was closed in 2005 due to high catch in the recreational sector. As well, an estimate of absolute abundance was available in the DSR fishery, calculated using fishery-independent data collected from submersible surveys of DSR abundance and habitat. Methods of stock assessment and management in the DSR fishery may not be feasible in KS^n fisheries due to a general lack of data and the inability to exclude a user-group.

Some characteristics of invertebrate fisheries inherently resulted in fewer stock assessment and management challenges, compared to finfish. For example, the species captured in invertebrate fisheries were not susceptible to barotrauma, most were targeted in single-species fisheries, and none were taken as bycatch. Furthermore, for abalone and geoduck fisheries, stock assessment via direct observation is relatively straightforward because these species are sessile and prefer relatively shallow depths. With the exception of the DSR fishery, all of the model fisheries that targeted species susceptible to barotrauma were considered relatively unsustainable. Even in the presence of multiple user-groups, discard mortality would be lower if the targeted species were not susceptible to barotrauma.

Due to low stock productivity typically associated with KS^n fisheries, it was reasonable to assume that there is a time-lag between implementation of management methods and a corresponding change in the status of a fishery. In other words, a fishery may be rebuilding under methods of management that did not result in the overfished state. For example, catch rates declined in the VRL fishery from the 1950s until the early to mid-1990s, and have since remained stable or have increased (Anon 2003). The recent improvement in catch rates was concurrent with implementation of relatively sophisticated methods of stock assessment and management, and reviewing the fishery in the early to mid-1990s, using the seven criteria, would reveal a management system that was not in place during overfishing.

The definition of a sustainable/successful fishery is challenging and in part depends on the management goal(s) of a particular fishery. There are a lack of conservation metrics in the model fisheries (few quantitative assessments), and as such it was difficult to compare stock status between fisheries and to compare fisheries regarding their assessments. Also, there is an obvious bias toward ‘developed’ countries and ‘developed fisheries’ because they have the majority of internet-based resources and the most documented fishery management practices.

Because only 13 model fisheries were reviewed in this study, and because the fisheries were not randomly selected, it may not be possible to extrapolate the results outside the context of this study. A more in-depth investigation using similar methods is required, perhaps by increasing the number of model fisheries considered, or by selecting fisheries with more specific life-history and spatial characteristics. However, using the conceptual framework of this study it was possible to extrapolate from best practice (Bardach 2004) to provide management recommendations for the inshore rockfish fishery.

Using the conceptual framework, the management systems of many KS^n fisheries appeared to promote sustainability. Ranking each fishery using the sustainability criteria appears to be effective because most of the sustainable fisheries incorporated each of the criteria to a higher degree than the relatively unsustainable fisheries. Accordingly, to identify a fishery management system that may potentially provide a sustainable IR fishery in British Columbia, it was necessary to determine if similar successful management systems would be appropriate in the IR fishery.

3. CASE-STUDY: INSHORE ROCKFISH IN BRITISH COLUMBIA

3.1. Introduction

Inshore rockfish are a complex of six rockfish species targeted in a KS^n fishery, and as such exhibit a suite of life-history and spatial characteristics that leave them particularly susceptible to overfishing. Inshore rockfish are targeted using hook and line gear in commercial, recreational, and aboriginal fisheries in British Columbia, and are caught incidentally in all other hook and line fisheries. Limitations on harvest of inshore rockfish began in 1986 and have become increasingly restrictive (Yamanaka et al. 2004).

Inshore rockfish populations are currently depleted due to overfishing and the lack of an effective management regime (Yamanaka and Lacko 2001). Based on the criteria established in this study, the IR fishery was relatively unsustainable (Table 3) and the management system met few of the seven criteria outlived above. The following section considers the biology of inshore rockfish and the current management system and challenges. Also, possible alternate management strategies for the inshore rockfish fishery are assessed based on sustainable components of KS^n fisheries, in terms of the seven criteria.

3.2. Biology of inshore rockfish (*Sebastes spp.*)

Inshore rockfish are K-selected (Archibald et al. 1981; Leaman and Beamish 1984; Love et al. 1990), have low productivity (Adams 1980; Musick 1999), and are late to reach sexual maturity. In fact, 50% of inshore rockfish are sexually mature from the ages of 11 to 20, depending on species (Yamanaka and Richards 1993; Kronlund and Yamanaka 2001). Members of the genus *Sebastes* also have a relatively large body size and have low larval survivorship

(Musick 1999), although they are highly fecund (Haldorson and Love 1991). Size-at-age and age-at-maturity of yelloweye rockfish (*Sebastes rubberimus*) has been shown to vary with latitude, presumably resulting from lasting effects of differential fishing pressure (Kronlund and Yamanaka 2001; Yamanaka and Lacko 2001). Species of inshore rockfish are assumed to undertake only limited migration after recruitment (Gunderson 1997), and therefore planktonic larval propagules are assumed to be the principal means of repopulating depleted areas.

3.3. Current management system and challenges

As a KS^n fishery, the use of traditional approaches of fisheries stock assessment and management leave inshore rockfish particularly susceptible to overfishing. Factors limiting the efficacy of stock assessment methods for inshore rockfish include:

(i) the use of fishery catch-per-unit-effort (CPUE) to index population trends and to detect disproportionate depletion within management areas;

(ii) variation in management strategies throughout the CPUE time-series (e.g., the implementation of limited entry in 1992 (Yamanaka and Lacko 2001; Yamanaka et al. 2004);

(iii) the lack of basic biological data (Love et al. 1990; Kronlund 1997; Parker et al. 2000);

(iv) the lack of a reliable abundance index or abundance estimate (Yamanaka and Lacko 2001);

(v) the inability to fully account for catch in all fishing sectors (inshore rockfish are incidentally caught in commercial, recreational, and Aboriginal fisheries along with all hook and line and trawl fisheries off the coast of BC) (Kronlund 1997; Yamanaka and Lacko 2001); and

(vi) an estimated 100% mortality of discarded fish due to barotrauma, thus complicating the use of traditional tag-recovery analysis and some survey methods.

These challenges, coupled with the use of large management areas (several large zones off the coast of BC) relative to the biology of inshore rockfish, may inhibit the ability to detect localized depletion (e.g., Yamanaka and Kronlund 1996; Yamanaka and Lacko 2001).

3.4. Possible alternative management systems

The “ideal” KS^n fishery management strategy should involve sustainable management methods, based on the seven criteria, from each of the model fisheries considered in this study (specified in Chapter 2). Due to the fishery, ‘site’, and species-specific characteristics, there does not appear to be a simple prescription to promote success when managing KS^n fisheries. Therefore, I focused on a collection of sustainable components from each fishery that were identified in Chapter 2. For example, the New Zealand rock lobster (NZRL) fishery is relatively sustainable but is highly data-dependent and employs complex computer simulation to assess management procedures in a prospective evaluation framework. Catch and biological data is relatively easy to collect in the NZRL fishery in comparison to the IR fishery. Similar management would be ideal in the IR fishery, but at present lack of data make such management impractical. In contrast, the Chilean loco (CL) fishery has been relatively sustainable since the inception of a particular co-management system known as a Territorial User-Rights Fishery, or a ‘TURF’ (Gelcich et al. 2007). Co-management is typically less data-intensive than traditional approaches to fisheries stock assessment and management, and as such is ideal for the IR fishery for the reasons specified in the section above. The small-scale of the CL fishery, relative lack of data, and limited mobility of loco are similar to the IR fishery and thus many elements of the TURF approach may be employed in the IR fishery. A TURF has been used in the CL fishery for over a decade and has decreased or eliminated the tyranny of scale by managing at a relatively small scale. Assignment of user-rights has also eliminated the tragedy of the commons effect. In general, a TURF works to address the sustainability criteria. For this reason, the following section describes the form of TURF employed in the CL fishery and then determines if the IR fishery may be structured as a TURF.

3.4.1. Co-operative management

'Command and control' management assumes that resource management problems are well-bounded, clearly defined, relatively simple, and often linear with regard to cause and effect (Holling and Meffe 1996). Co-management is in direct opposition to command and control management, which often results in undesired consequences in terms of sustainability (Holling and Meffe 1996). Co-management involves projects at a local level and stewardship with a high degree of community involvement that actively involves resource users in decision-making, and fosters communication between stakeholders, empowers users in resource management, and incorporates knowledge from more sources than conventional management (Pinkerton 1989, 1994; Berkes et al. 2001; Defeo and Castilla 2005). Co-management implies local fishing access rights and as such increases fisher incentive for resource sustainability, working in opposition to command and control management (Pinkerton 1989; Ostrom et al. 1994; Baland and Platteau 1996; Lane and Stephenson 2000; Dietz et al. 2003; Pomeroy and Rivera-Guieb 2005).

3.4.2. Territorial use rights fisheries (TURFs)

Fishing access rights are the central element of a TURF (Christy 1982), which is a form of community-based co-management where access rights to engage in fishing in a particular geographical location are assigned to stakeholders (e.g., individuals, groups, governments). As noted in Christy (1982), TURFs increase user-rights and limit capital and labour to the point where greatest net benefits are produced, and therefore avoid a tragedy of the commons by conferring a level of resource ownership to the user. TURFs are most effective where specificity of ownership is clear and decision making is relatively simple. (Christy 1982; Hilborn et al. 2003, 2005; Gelcich et al. 2007). Enforcement of TURFs is typically performed by a nearby community since traditional TURFs are community-based (Defeo and Castilla 2005). Historically, TURFs have been used in small-scale artisanal fisheries in coastal waters where small groups of fishers operate from small boats (Defeo and Castilla 2005).

Low mobility fish and invertebrate species, such as those targeted in KS^n fisheries, are most suitable for management under a TURF (Christy 1982) because both co-management and fishery monitoring occur at relatively small scales (Christy 1982; Gelcich et al. 2007).

The Chilean fishery for loco was re-structured as a TURF in 1991 as an essential component of the Chilean Fisheries and Aquaculture Law. TURFs were originally employed to counter overfishing of loco by commercial and artisanal fishers. Catches in the fishery were lowest in approximately 1982, stabilized in 1993, and have increased since (Castilla and Defeo 2001; Gelcich et al. 2007). Under the co-management system, syndicates of fishers apply for fishing rights for specific areas of the seabed. Sustainable applicants pay for a baseline study of the area, from which catch quotas are determined and management plans are established. The syndicates are responsible for hiring external consultants who conduct annual assessments of fishing grounds and suggest annual changes in TAC if necessary (Gelcich et al. 2007). Benthic resources and TAC proposals within each TURF are co-managed by the central authority (i.e., the Undersecretary of Chilean Fisheries) and the syndicates (Castilla and Defeo 2001; Parma et al. 2001; Gelcich et al. 2007). The CL fishery, structured as a TURF, appears have contributed to its growing success.

3.4.3. The inshore rockfish fishery under a TURF system

Some basic conditions already exist in the IR fishery that may allow restructuring the fishery as a TURF. This section elaborates on seven components that were identified to affect the creation and maintenance of sustainable TURFs (Christy 1982); Management areas in the IR fishery should (1) be large enough so that harvesting outside of the area does not significantly diminish the value of use within the area. If the entire IR fishery were structured as a TURF, spawning biomass should be kept above a permitted level (a lower reference point) in all areas, thus allowing recruitment from source areas that may be outside of the managed area(s). Because of the uncertainty of IR movement and population structure, delineating TURF areas in the IR fishery may be challenging. However, similar to the selection of existing Rockfish Conservation Areas (RCAs), delineation of areas could be accomplished through the combined traditional ecological knowledge of recreational, commercial, and aboriginal harvesters, and with existing fishery-dependent and independent data.

Regulations in each management area in the IR fishery should be (2) monitored closely and protected by overarching federal laws. This is possible using existing electronic monitoring technology as required by Fisheries and Oceans Canada (DFO). Areas of the present IR fishery are in remote locations, and the fishery takes place in a large geographical area compared to traditional TURFs. As such, fishers may not live near their fishing grounds, which may limit the use of methods of enforcement and monitoring used in traditional TURFs. Electronic monitoring is already used most Canadian groundfish fisheries and the applicability of using electronic monitoring to manage IR fishery as a TURF should be considered (Ames et al. 2007).

Areas of the IR fishery should also (3) be clearly demarcated and identifiable. This would also be possible using modern navigational and charting equipment and mapping software provided by electronic monitoring, as is sustainable with delineating areas of the IR fishery which are currently closed to fishing.

IR are relatively sedentary, and as such (4) possess the requisite biological characteristics to be managed as a TURF. However, because of the presence of both a recreational and First Nations fishery, it is uncertain whether (5) cultural conditions that permit acquisition of exclusive user rights are present in the fishery. Cultural factors may be taken into account by allowing specific areas of the TURF to be used exclusively by particular fishing sectors, as occurs in the CL fishery.

To address the crucial issue of resource access and allocation, (6) profits from an IR TURF need to be distributed equitably and (7) by a government authority. A lottery or auction system could be employed if fishing location preferences coincide. The use of auction and lottery systems for U.S. fisheries has been promoted by Macinko and Bromley (2002). Lottery systems for quota allocation are employed and apparently sustainable in the SBPS fishery (Jackson et al. 2005), the commercial fishery for geoduck in Washington (Orensanz et al. 2004), fish stocks in the Falkland Islands (Barton 2002), and for fish corrals, oyster culture beds, and milkfish fry in the Philippines (Smith and Panayotou 1984). However, both aboriginal and recreational sectors also use the IR resource, and the equitable distribution of fishing rights to these sectors is more challenging. To address this, areas of the TURF could be allocated to these fishing sectors for their exclusive use. DFO would continue to enforce the distribution of fishing rights and would limit or exclude access to each managed area.

A TURF approach could also improve management and stock assessment. For example, similar to methods successful in the CL fishery, fishers could work individually or in syndicates to determine the amount and type of use within each managed area. Also successful in the CL fishery is the requirement that syndicates hire external consultants to conduct annual stock assessments of each management area, and present the findings to a central authority. Also similar to the NZRL and CL fisheries, management and stock assessment in the IR fishery may vary by management area, permitting the development of area-specific stock rebuilding targets such as a specified level of CPUE or stock abundance.

Under DFO’s guidance, each fisher or syndicate would maintain the rights to extract benefits from each of their territories, as specified in a fishing license. It is also necessary to ensure future returns from the fishery, which may be low because of the low productivity of the species.

One of the primary management challenges in the IR fishery is the absence of a reliable index of stock size, which arises mainly from the suite of stock assessment challenges (Yamanaka and Lacko 2001). Structuring the IR fishery as a TURF would address more fundamental issues regarding common property resources and user-rights, as explained above. A TURF may eventually allow more data-intensive and sustainable management methods such as those employed in the NZRL and CL fisheries. For instance, the use of fishery-dependent data as a primary source of indexing stock status was sustainable in some KS^n fisheries, in particular, the relatively data-rich NZRL fishery where harvest control rules and reference points were based on CPUE data from the fishery.

4. CONCLUSION

Despite substantial challenges in KS^n fisheries, relative sustainability/success has been achieved in several situations. Partly due to fishery, site, and species-specific characteristics, each of the model fisheries implemented the seven sustainability criteria to varying degrees. Fisheries that implemented the criteria to a higher degree were in general more sustainable.

The IR fishery requires a unique method of management to address each of the criteria. The single most challenging aspect of applying the criteria of each candidate model fishery to the IR fishery was the lack of fisher incentive for resource sustainability, stemming from the absence of sufficient user-rights. For this reason, it was determined that a system of governance based on user-rights, specifically a TURF, could potentially improve management of the IR fishery. The CL fishery shares many similarities with the IR fishery, and was structured as a TURF to recover from overfishing. Since structuring as a TURF, the CL fishery has improved considerably as fundamental concerns regarding common property resources and user-rights were addressed. Structuring the IR fishery as a TURF may benefit from the existing IVQ system, an existing network of RCAs, and a comprehensive monitoring system which could improve data collection to permit more specific stock assessment and a prospective evaluation of harvest and management strategies, which have proven quite sustainable in the NZRL fishery.

Several limitations of this study were provided. Most notably was the confounding between the seven criteria, effecting the interpretation of whether sustainability of each fishery was related to the level of incorporation of each criterion. Also, due to access of information, the selection of model fisheries was limited in number and to fisheries in relatively developed nations. A worthwhile extension of this study should include more model fisheries and should concentrate on species with a narrower scope of spatial and life-history characteristics.

This study developed a valuable framework to compare sustainability between fisheries and was an effective tool to address the objective of this study, which was to potentially provide a sustainable IR fishery in British Columbia despite the major challenges and risks associated with KS^n fisheries.

REFERENCE LIST

Adams, P.B. 1980. Life-history patterns in marine fishes and their consequences for fisheries management. Fishery Bulletin 78 (1): 1-12.

Alvarez, P., L. Motos, A. Uriarte, and J. Egana. 2001. Spatial and temporal distribution of European hake (*Merluccius merluccius*), eggs and larvae in relation to hydrographical conditions in the Bay of Biscay. Fish Res 50: 111–128.

Ames, R.T., Leaman, B.M., and Ames, K.L. 2007. Evaluation of video technology for monitoring of multispecies longline catches. North American Journal of Fisheries Management. 27(3): 955-964.

Anon. 2003. Rock Lobster Fishery Management Plan, 2003, Compiled by the Rock Lobster and Giant Crab Fishery Management Steering Committee, Fisheries Victoria Management Report Series 1: 52 p.

Archibald, C.P., D. Fournier and B.M. Leaman. 1983. Reconstruction of stock history and development of rehabilitation strategies for Pacific Ocean perch in Queen Charlotte Sound, Canada. North American Fisheries Management 3: 283-294.

Archibald, C.P., W. Shaw and B.M. Leaman. 1981. Growth and mortality estimates of rockfishes (Scorpaenidae) from B.C. coastal waters 1977–1979. Can. Tech. Rep. Fish. Aquat. Sci. 1048.

Baland, J.M. and J.P. Platteau. 1996. Halting Degradation of Natural Resources: Is there a Role for Rural Communities? New York: Oxford University Press.

Barber, K., editor. 2004. The Canadian Oxford Dictionary, second edition. Toronto: Oxford University Press.

Barber, W.E. and J.N. Taylor. 1990. The Importance of Goals, Objectives, and Values in the Fisheries Management Process and Organization: A Review. North American Journal of Fisheries Management 10: 365–373.

Bardach, 2004. Presidential Address-The Extrapolation Problem: How Can We Learn from the Experience of Others? Journal of Policy Analysis and Management 23: No.2: 205-220.

Barton, J. 2002. Fisheries and Fishery Management in Falkland Islands Conservation Zones. Aquatic Conservation-Marine and Freshwater Ecosystems 12(1): 127-135.

Bentley, N., P.A. Breen and P.J. Starr. 2003. Design and evaluation of a revised management decision rule for red rock lobster fisheries (*Jasus edwardsii*) in CRA 7 and CRA 8. New Zealand Fisheries Assessment Report 2003/30. 44 p.

Bentley, N., P.A. Breen, S.W. Kim and P.J. Starr. 2005. Can additional abundance indices improve harvest control rules for New Zealand rock lobster (*Jasus edwardsii*) fisheries? New Zealand Journal of Marine and Freshwater Research 39.

Berkeley, S.A. and D. Markle. 1999. Effects of fishing-induced age truncation on recruitment and reproductive potential in black rockfish (*Sebastes melanops*). Final report to Oregon Sea Grant, R/OPF-46, Corvallis.

Berkeley, S.A., M.A. Hixon, R.J. Larson and M.S. Love. 2004. Fisheries sustainability via protection of age structure and spatial distribution of fish populations. Fisheries 29 (8): 23-32.

Berkes, F., R. Mahon, P. McConney, R. Pollnac, and R. Pomeroy. 2001. Managing Small-scale Fisheries: Alternative Directions and Methods. Pub: International Development Research Centre.

Boehlert, G.W., and R.F. Kappenman. 1980. Variation of Growth with Latitude in Two Species of Rockfish (Sebastes pinniger and S. diploproa) from the Northeast Pacific Ocean. Marine Ecology Progress Series. Vol. 3: 1-10.

Botsford, L.W., C.L. Moloney, A. Hastings, J.L. Largier, T.M. Powell, K. Higgins and J.F. Quinn. 1994. The influence of spatially and temporally varying oceanographic conditions on meroplanktonic metapopulations. Deep-Sea Res. Part II: Top. Stud. Oceanogr. 41: 107–145.

Botsford, L.W., F. Micheli and A. Hastings. 2003. Principles for the design of marine reserves. Ecological Applications 13(1): Supplement, 2003, S25–S31.

Botsford, L.W., J.C. Castilla and C.H. Peterson. 1997. The management of fisheries and marine ecosystems. Science 277 (5325): 509-515.

Botsford, LW, Campbell A, Miller R. 2004. Biological reference points in the management of North American sea urchin fisheries. Can. J. Fish. Aquat. Sci. 61: 1325–1337.

Breen, P.A. 1992. A review of models used for stock assessment in abalone fisheries. In 'Abalone of the World: Biology Fisheries and Culture'. (*Eds*: Shepherd, SA, Tegner MJ, Guzmhn SA). Blackwell: Oxford. 253-75.

Breen, P.A., S.W. Kim, N. Bentley and P.J. Starr. 2003. Preliminary evaluation of maintenance management procedures for New Zealand rock lobster (*Jasus edwardsii*) fisheries. New Zealand Fisheries Assessment Report 2003/20. 65 p.

Buonaccorsi, V.P., C.A. Kimbrell, E.A. Lynn and RD Vetter. 2005. Limited realized dispersal and introgressive hybridization influence genetic structure and conservation strategies for brown rockfish, *Sebastes auriculatus*. Conservation Genetics 6: 697-713.

Butler, J.L., J.D. Jacobson, J.T. Barnes, H.G. Moser and R. Collins. 1999. Stock assessment of cowcod. *In* Pacific Fisheries Management Council. Status of the Pacific Coast Groundfish Fishery through 1998 and recommended biological catches for 1999: Stock Assessment and Fishery Evaluation.

Butler, J.L., J.D. Jocobson, T.J. Barnes and H.G. Moser. 2003. Biology and population dynamics of cowcod (*Sebastes levis*) in the southern California Bight. Fish. Bull. 101: 260–280.

Butterworth, D.S. and A.E. Punt. 1999. Experiences in the evaluation and implementation of management procedures. ICES Journal of Marine Science 56 (6): 985-998.

Butterworth, D.S., K.L. Cochrane and J.A.A. De Oliveira. 1997. Management procedures: a better way to management fisheries? The South African experience. *In:* Pikitch EK, Huppert DD, Sissenwine MP. ed. Global trends: fisheries management. American Fisheries Society Symposium 20: 83-90.

Caddy, J.F. 1999. Fisheries management in the twenty-first century: will new paradigms apply? Review in Fish Biology and Fisheries 9 (1).

Caddy, J.F. 2004. Current usage of fisheries indicators and reference points, and their potential application to management of fisheries for marine invertebrates. Can. J. Fish. Aquat. Sci. 61: 1307–1324.

Campbell, A., D. Tzotzos, W.C. Hajas and L.L. Barton A. 2001 Quota options for the red sea urchin fishery in British Columbia for fishing season 2002/2003. Canadian Stock Assessment Secretariat Research Document 2001/141.

Campbell, A., J. Boutillier and J. Rogers. 1999. Discussion on a Precautionary approach for management of the red sea urchin fishery in British Columbia. Canadian Stock Assessment Secretariat Research Document 1999/94.

Carvalho, G.R. and L. Hauser. 1994. Molecular genetics and the stock concept in fisheries. Rev. Fish. Biol. Fish. 4: 326-350.

Castilla, J.C. and M. Fernandez. 1998. Small-Scale Benthic Fisheries in Chile: On Co-Management and Sustainable Use of Benthic Invertebrates. Ecological Applications Vol. 8, No. 1, Supplement: Ecosystem Management for Sustainable Marine Fisheries (Feb.1998), S124-S132.

Castilla, J.C. and O. Defeo. 2001. Latin American benthic shellfisheries: emphasis on co-management and experimental practices. Reviews in Fish Biology and Fisheries 11 (1).

Christy, F.T. 1982. Territorial use rights in marine fisheries: definitions and conditions. FAO Fish. Tech. Pap. 227: 10 p.

Clark, C.W. 1985 Bioeconomic Modelling and Fisheries Management. Wiley-Interscience, New York.

Cochrane, K.L., D.S. Butterworth, J.A.A. de Oliveira and B.A. Roel. 1998. Management procedures in a fishery based on highly variable stocks and with conflicting objectives: Experiences in the South African pelagic fishery. Reviews in Fish Biology and Fisheries 8 (2): 177-214.

de la Mare, W.K. 1998. Tidier Fisheries Management Requires a New MOP (Management Oriented Paradigm). Reviews in Fish Biology and Fisheries 8(3): 349-356.

de Lestang, S. and R. Melville-Smith. 2004. West Coast Bioregion: Commercial Fisheries, West Coast Rock Lobster Managed Fishery Status Report. Department of Fisheries State of the Fisheries Report 2004/05. http://www.fish.wa.gov.au/docs/sof/2004/sof20042005-0203-01.pdf (Dec. 2006).

Defeo, O. and J.C. Castilla. 2005. More than one bag for the world fishery crisis and keys for co-management successes in selected artisanal Latin American shellfisheries. Reviews in Fish Biology and Fisheries 15: 265-283.

Department of Environment and Heritage. 2003. Victorian Abalone Fishery: Environmental assessment under the Environment Protection and Biodiversity Conservation Act 1999, Department of Environment and Heritage, Canberra, Australia.

Department of Natural Resources (DNR). 2002. The State of Victoria, Department of Natural Resources and Environment, April 2002. Victorian Abalone Fishery Management Plan. Fisheries Victoria ISBN 171060052.

DFO. 2006a. Pacific Region, Integrated Fisheries Management Plan, Red Sea Urchin by Dive. Fisheries and Oceans Canada. August 1st to July 31st, 2007. http://www-ops2.pac.dfo-mpo.gc.ca/xnet/content/mplans/plans06/RSU06pl.pdf (Feb. 2007).

DFO. 2006b. Pacific Region, Integrated Fisheries Management Plan, Geoduck and Horse Clam. Fisheries and Oceans Canada. January 1st to September 31st, 2006. http://www-ops2.pac.dfo-mpo.gc.ca/xnet/content/mplans/plans06/RSU06pl.pdf (Feb. 2007).

DFO. 2006c. Pacific Region, Integrated Fisheries Management Plan, Groundfish. Fisheries and Oceans Canada. April 1st 2006 to March 1st 2007. http://www-ops2.pac.dfo-mpo.gc.ca/xnet/content/mplans/plans06/Groundfish06-07/Fnl%2006_07%20Groundfish%20IFMP.pdf (Feb 2007).

Dietz, T., E. Ostrom and P.C. Stern. 2003. The Struggle to Govern the Commons. SCIENCE (302) 5652: 1907-1912.

Francis, R.I.C.C. and Shotton. 1996. "Risk" in fisheries management: a review. Can. J. Fish. Aquat. Sci. 54: 1699–1715.

Freeman, M.C., C.M. Pringle, and C.R. Jackson. 2007. Hydrologic connectivity and the contribution of stream headwaters to ecological integrity at regional scales. Journal of American Water Resources Association. Vol. 43(1): 5-14.

Gelcich, S., Edward-Jones, G., and Kaiser, M.J. 2007. Heterogeneity in fishers' harvesting decisions under a marine territorial user rights policy. Ecological Economics 61 (2-3): 246-254.

Gharrett, A.J., Palof, K., Kamin, Londzela, C. M.), and Heifetz, J. 2006. A Preliminary Look at Geographic and Cohort Variation in Genetic Structure of Alaskan Pacific Ocean Perch. Journal of Fish Biology 69: 263-263.

Grafton, R.Q., R. Arnason, T. Bjørndal, D. Campbell, H.F. Campbell, C.W. Clark, R. Connor, D.P. Dupont, R. Hannesson, R. Hilborn, J.E. Kirkley, T. Kompas, D.E. Lane, G.R. Munro, S. Pascoe, D. Squires, S.I. Steinshamn, T.B. Turris and Q. Weninger. 2006. Incentive-based approaches to sustainable fisheries. Canadian Journal of Fisheries and Aquatic Sciences 63: 699-710.

Gulland, J.A. 1969. Manual of methods for fish stock assessment. Part 1. Fish population analysis. FAO Manual of Fisheries Science 5: 1-54.

Gunderson, D.R. 1977. Population biology of pacific ocean perch (*sebastes-alutus*), stocks in Washington-Queen Charlotte Sound region, and their response to fishing. Fish Bull. 75 (2): 369-403.

Gunderson, D.R. 1997. Spatial patterns in the dynamics of slope rockfish stocks and their implications for management. Fish. Bull. 95: 219-230.

Gunderson, D.R., P. Callahan, and B. Goiney. 1980. Maturity and fecundity of four species of *Sebastes*. Mar. Fish. Rev. 42 (3-4): 74–79.

Haldorson, L. and M. Love. 1991. Maturity and fecundity in the rockfishes (*Sebastes spp.*) a review. Mar. Fish. Rev. 53 (2): 25-31.

Hannesson. 2004. Rights based fishing: Use rights versus property rights to fish. Reviews in Fish Biology and Fisheries. 15(3): 231-241.

Hanselman, D., P. Spencer and K. Shotwell. 2007. Localized depletion of three rockfish species. Biology, Assessment, and Management of Pacific Rockfishes. Alaska Sea Grant College Program. AK-SG-07-01, 2007.

Hardin, G. 1968. Tragedy of commons. Science 162 (3859): 1243-1248.

Healey, M.C. and T. Hennessey. 1998. The paradox of fairness: the impact of escalating complexity on fishery management. Mar. Policy 22: 109-118.

Hesp, S.A., I.C. Potter and N.G. Hall. 2002. Age and size compositions, growth rate, reproductive biology and habitats of the West Australia Dhufish (*Glaucosoma hebraicum*) and their relevance to the management of this species. Fish Bull. 100: 214-227.

Hilborn, R. 1979. Comparison of fisheries control systems that utilize catch and effort data. Journal of the Fisheries Research Board of Canada 36: 1477–1489.

Hilborn, R. 2006. Fisheries Success and Failure: the Case of the Bristol Bay Salmon Fishery. Bulletin of Marine Science 78: 487-498.

Hilborn, R. 2003c. The state of the art in stock assessment: where we are and where we are going . Scientia Marina 67: 15-20.

Hilborn, R. and C.J. Walters. 1992. Quantitative fisheries stock assessment: choice, dynamics and uncertainty. Chapman and Hall. New York.

Hilborn, R., J.M. (Lobo) Orensanz and A.M. Parma. 2005. Institutions, incentives and the future of fisheries. Phil. Trans. R. Soc. B (2005) 360: 47-57.

Hilborn, R., T.A. Branch, B. Ernst, A. Magnusson, C.V. Minte-Vera, M.D. Scheuerell and J. Valero. 2003a. State of the World's Fisheries. A. Rev. Environ. Resources 28: 359-399.

Hilborn, R., T.P. Quinn, D.E. Schindler and D.E. Rogers. 2003b. Biocomplexity and fisheries sustainability. Proceedings of the National Academy of Sciences 100: 6564-6568.

Hilden, M. 1993. Reference points for fisheries management: the ICES experience. In Risk evaluation and biological reference points for fisheries management. Edited by S.J. Smith, J.J. Hunt, and D. Rivard. Can. Spec. Publ. Fish. Aquat. Sci. No. 120. p. 59-65.

Hobday, D., A.E. Punt and D.C. Smith. 2005. Modelling the effects of Marine Protected Areas (MPAs) on the southern rock lobster (*Jasus edwardsii*) fishery of Victoria, Australia. New Zealand Journal of Marine and Freshwater Research, 2005, 39: 675–686.

Holling, C.S. and G.K Meffe. 1996. Command and control and the pathology of natural resource management. Conservation Biology 10: 328-337.

Hutchings, J.A. and J.D. Reynolds. 2004. Marine fish population collapses: consequences for recovery and extinction. Biol. Rev. 54: 297–309.

Ianelli, J.N. and D.H. Ito. 1992. 1992 Pacific Ocean perch *In* Stock assessment and fishery evaluation report for the Groundfish resources of the Bering Sea/Aleutian Islands region as projected for 1993 (November 1992), 36 p. North Pacific Fishery Management Council, P.O. Box 103 136, Anchorage, AK 99510.

Jackson, G., N. Sumner, A. Cribb and J. Norris. 2005. 'Comparing conventional social-based and alternative output-based management models for recreational fisheries using Shark Bay pink Snapper as case study.' Final Report to FRDC for Project no. 2003/066. Fisheries Research and Development Corporation, Canberra, Australia. 80 p.

Jacobson, L.D. and S.X. Cadrin. 2002. Stock-rebuilding time isopleths and constant-*F* stock-rebuilding plans for overfished stocks. Fish. Bull. 100: 519–536.

Jennings, S., J.D. Reynolds and S.C Mills. 2004. Life history correlates of responses to fisheries exploitation. Proc. R. Soc. Lond. B (1998) 265: 333-339.

Kronlund, A.R. 1997. A discussion paper on reconciling assessment and management of inshore rockfish. Canadian Stock Assessment Secretariat Research Document - 1997/137.

Kronlund, A.R. and K.L. Yamanaka. 2001. Yelloweye rockfish (*Sebastes ruberrimus*) life history parameters and assessed from areas with contrasting fishing histories. Spatial Processes and Management of Marine Populations. Alaska Sea Grand College Program. AK-SG-01-02. 257-280.

Lane, D.E. and R.L. Stephenson. 2000. Institutional arrangements for fisheries: alternate structures and impediments to change. Marine Policy 24: 385-393.

Lauck, T., C.W. Clark, M. Mangel and G.R. Munro. 1998. Implementing the precautionary principle in fisheries management through marine reserves. Ecological Applicaitons 8 (1): S72-S78.

Leaman ,B.M. and R.J. Beamish. 1984. Ecological and management implications of longevity in some Northeast Pacific groundfishes. Int. North Pac. Fish. Comm. Bull. 42: 85–97.

Leaman, B.M. 1991. Reproductive styles and life history variables relative to exploitation and management of *Sebastes* stocks. Environ. Biol. Fish. 30: 253–271.

Leaman, B.M. 1993. Reference points for fisheries management: the western Canadian experience. In Risk evaluation and biological reference points for fisheries management. Edited by S.J. Smith, J.J. Hunt, and D. Rivard. Can. Spec. Publ. Fish. Aquat. Sci. No. 120. p. 15-30.

Levin, S.A. 1998. Ecosystems and the Biosphere as Complex Adaptive Systems. Ecosystems 1:431-436.

Longhurst, A. 2002. Murphy's law revisited: longevity as a factor in recruitment to fish populations. Fisheries Research 56 (2): 125-131.

Love, M.S. 2002. The rockfishes of the northeast Pacific. University of California Press.

Love, M.S., P. Morris, M. McCrae and R. Collins. 1990. Life history aspects of 19 rockfish species (Scorpaenidae: *Sebastes*) from the southern California Bight. U.S.Dep. Commer., NOAA Tech. Rep. NMFS 87: 38 p.

Ludwig, D., R. Hilborn and C. Walters. 1993. Uncertainty, resource exploitation, and conservation - lessons from history. Science 260 (5104): 17 p.

MacArthur, R.H. and E.O. Wilson. 1967. The theory of island biogeography. Princeton University Press, Princeton, New Jersey, USA.

Mace, P. 2004. In Defence of Fisheries Scientists, Single-species Models and other Scapegoats: Confronting the Real Problems. Marine Ecology Progress Series (274): 285-291.

Macinko, S. and S.W. Bromley. 2002. Who Owns America's Fisheries? Washington, DC: Island Press.

Maguire, J.J. 2003. Large volume demersal fishery in the North Atlantic . *In* Report and documentation of the International Workshop on the Implementation of International Fisheries Instruments and Factors of Unsustainability and Overexploitation in Fisheries, Mauritius, 3-7 February 2003. *Ed.* J. Swan and D. Greboval. FAP, Rome, Italy.

MSC, 2008. Marine Steward Council Online, List of Certified Fisheries. http://www.msc.org/html/content_484.htm (April 2008).

McAllister, M.K., P.J. Starr, V.R. Restrepo and G.P. Kirkwood. 1999. Formulating quantitative methods to evaluate fishery-management systems: what fishery processes should be modelled and what tradeoffs should be made? ICES Journal of Marine Science 56: 900–916.

Murawski, S.A. 2000. Definitions of overfishing from an ecosystem perspective. ICES Journal of Marine Science 57: 649–658.

Musick, J.A. 1999. Criteria to define extinction risk in marine fishes - The American Fisheries Society initiative. FISHERIES 24 (12): 6-14.

Myers, R.A., G. Mertz and P.S. Fowlow. 1997. Maximum population growth rates and recovery times for Atlantic cod Gadus morhua. Fish. Bull. 95: 762–772.

NPFMC (GOA FMP). 2005. Fishery Management Plan for Groundfish of the Gulf of Alaska. North Pacific Fishery Management Council. Anchorage, Alaska. http://www.fakr.noaa.gov/npfmc/fmp/goa/GOA.pdf (Dec. 2006).

NRLMG. 2005. National rock lobster management group: 2005 Annual Report to the Minister of Fisheries. http://www.seafood.co.nz/doclibrary/industryorgs/rlic/NRLMG_ANNUAL_REPORT_FINAL_2005.pdf (Dec. 2006).

O'Connell, V., C. Brylinsky and D. Carlile. 2006. Gulf of Alaska Demersal Shelf Rockfish. Chapter 13: Assessment of the Demersal Shelf Rockfish Stock for 2006 in the Southeast Outside District of the Gulf of Alaska. http://www.afsc.noaa.gov/REFM/docs/2005/GOAdemersalrock.pdf (Dec. 2006).

Orensanz, J.M.L. and G.S. Jamieson. 1998. The assessment and management of spatially structured stocks: an overview of the North Pacific symposium on invertebrate stock assessment and management. In: Jamieson GS, Campbell A, eds. Proceedings of the North Pacific Symposium on Invertebrate Stock Assessment and Management. Canadian Special Publication on Fisheries and Aquatic Sciences 125: 441–59.

Orensanz, J.M.L., C.M. Hand, A.M. Parma, J. Valero and R. Hilborn. 2004. Precaution in the harvest of Methuselah's clams - the difficulty of getting timely feedback from slow-paced dynamics. Canadian Journal of Fisheries and Aquatic Sciences 61 (8): 1355-1372.

Ostrom, E.R., Gardner and J. Walker. 1994. Rules, games, and common-pool resources. Ann Arbor: University of Michigan Press.

Parker SJ, Berkeley SA, Golden JT, Gunderson DR, Heifetz J, Hixon MA, Larson R, Leaman BM, Love MS, Musick JA et al. 2000. Management of Pacific Rockfish. Fisheries 25: 22–29.

Parma AM, Orensanz JM (Lobo), Elías I, Jerez G. 2001. 2001 Australian Society for Fish Biology Workshop Proceedings. Diving for shellfish and data: incentives for the participation of fishers in the monitoring and management of artisanal fisheries around southern South America.

Pearson DE, Hightower JE. 1991. Spatial and temporal variability in growth of widow rockfish (*Sebastes entomelas*). NOAA Tech. Mem. NOAA-TM-NMFS-SWFSC-167: 43 p.

Perry RI, Walters CJ, Boutillier JA. 1999.. A framework for providing scientific advice for the management of new and developing invertebrate fisheries 9 (2).

Pido MD. 1995. Toward improved linkage between research and management in marine fisheries. Naga 18: 15-18.

Piner K. 2005. 2005 Stock Status of Cowcod in the Southern California Bight and Future Prospects. Southwest Fisheries Science Center. http://www.pcouncil.org/groundfish/gfstocks/Cowcod_Assessment_May25th.pdf (Dec. 2006).

Pinkerton E. 1989. *In* E. Pinkerton (ed.), Co-operative management of local fisheries: New direction in improved management and community development. University of British Columbia Press, Vancouver, British Columbia.

Pinkerton E. 1992. Translating Legal Rights into Management Practice: Overcoming Barriers to the Exercise of Co-Management. Human Organization 51 (4): 330-341.

Pinkerton E. 1994. Local Fisheries Co-Management: A Review of International Experiences and Their Implications for British Columbia Salmon Management. Canadian Journal of Fisheries and Aquatic Sciences 51 (10): 2363-2378.

Pomeroy RS, Rivera-Guieb R. 2005. Fisheries Co-management: A Practical Handbook. CABI/IDRC, 2005, ISBN 1-55250-184-1, 288 p.

Prince, J.D. 2005. Combating the tyranny of scale for haliotids: Micro-management for microstocks. Bulletin of Marine Science 76 (2): 557-577.

Prince, J.D. 1989. The fisheries biology of the Tasmanian stocks of *Haliotis rubra*. PhD Thesis, University of Tasmania, Tasmania, 174 p.

Prince, J.D. 2003. The barefoot ecologist goes fishing. Fish and Fisheries 4: 359-371.

Quesne, W.J.F., Hawkins, S.J., and Shepherd, J.G. 2007. A comparison of no-take zones and traditional fishery management tools for managing site-attached species with a mixed larval pool. Fish and Fisheries 8(3): 181-195.

Restrepo, V.R. and J.E. Powers. 1999. Precautionary control rules in US fisheries management: specification and performance. – ICES Journal of Marine Science 56: 846–852.

Richards, W.J. 1994. Symposium on recent advances in reef fish recruitment research – Introduction. Bulletin of Marine Science 54 (1): 203-205.

Rivard, D. and J.J. Maguire. 1993. Reference points for fisheries management: the eastern Canadian experience. In Risk evaluation and biological reference points for fisheries management. Edited by S.J. Smith, J.J. Hunt, and D. Rivard. Can. Spec. Publ. Fish. Aquat. Sci. No. 120. p. 31-57.

Rock Lobster Industry Advisory Committee (RLIAC). 1999. Operational and Work Plans for the West Coast Rock Lobster Managed Fishery. http://www.fish.wa.gov.au/docs/macs/rliac/rliac99/RLIACplans1999.pdf (Dec. 2006).

Rosenberg, A., P. Mace, G. Thompson, G. Darcy, W. Clark, J. Collie, W. Gabriel, A CacCall, R. Method, J. Powers, V. Restrepo, T. Wainwright, L. Botsford, J. Hoenig and K. Stokes. 1994. Scientific review of definitions of overfishing in US Fishery Management Plans. National Oceanic and Atmospheric administration (US) Technical Memorandum. NMFS-F/SPO-17. 205 p.

SAFMC. 1983. Fishery Management Plan for the Snapper Grouper Fishery of the South Atlantic Region. South Atlantic Fishery Management Council, Charleston, SC. http://www.safmc.net/Portals/6/Library/FMP/SnapGroup/SnapGroupFMP.pdf (Dec. 2006).

Sainsbury, K.J., A.E. Punt and A.D.M. Smith. 2000. Design of operational management strategies for achieving fishery ecosystem objectives ICES Journal of Marine Science 57 (3): 731-741.

Seidewitz, E. 2003. What Models Mean. IEEE Software. 20(5): 26-32.

Shanks, A.L., B.A. Grantham and M.H. Carr. 2003. Propagule dispersal distance and the size and spacing of marine reserves. Ecol. Appl. 13: S159–S169.

Shepherd, S.A. and L.D. Brown. 1993. What is an abalone stock: implications for the role of refugia in conservation. Can. J. Fish. Aquat. Sci. 50: 2001-2009.

Smith I.R. and T. Panayotou. 1984. Territorial use rights and economic efficiency: the case of the Phillippine fishing concessions. FAO Fish. Tech. Pap. 245: 17 p.

Southeast Data, Assessment, and Review. (SEDAR) 2004. SEDAR 6 Stock Assessment Report 1, Goliath Grouper. SEDAR Report SEDAR6-SAR1, Charleston, SC. http://sero.nmfs.noaa.gov/sf/safereports/docs/SAFE%20CD/SEDAR_assessments/Sedar_6_Goliath_Hogfish/SEDAR6_SAR1_Goliathall.pdf (Dec. 2006).

St John J., J. King and K. Saville. 2004. West Coast Bioregion: West Coast Demersal Scalefish Fishery Status Report. http://www.fish.wa.gov.au/docs/sof/2004/sof20042005-0203-11.pdf (Dec. 2006).

Starr, P.J., N. Bentley, P.A. Breen and S.W. Kim. 2003. Assessment of red rock lobsters (Jasus edwardsii) in CRA 1 and CRA 2 in 2002. *New* Zealand Fisheries Assessment Report 2003/41. 119 p.

Starr, P.J., P.A. Breen, R. Hilborn and T.H. Kendrick. 1997. Evaluation of a management decision rule for a New Zealand rock lobster substock. Marine and Freshwater Research 48: 1093–1101.

UN. 1995. Agreement for the implementation of the provisions of United Nations Convention on the Law of the Sea of 10 December 1982 relating to the conservation and management of straddling fish stocks and highly fish stocks. United Nations General Assembly Document A/CONF.164/37 (8 September 1995), New York.

Walters, C.J. 2000. Natural selection for predation avoidance tactics: implications for marine population and community dynamics. Mar. Ecol. Prog. Ser. 208: 299–313.

Walters, C.J. and S. Martell. 2003. Harvest Management for Aquatic Ecosystems. Princeton University Press, Princeton.

World Bank, United Nations Development Programme, Commission of Environmental Corporation, and Food and Agriculture Organizations (World Bank). 1992. A study of international fisheries research. The World Bank, Washington, D.C., USA.

Yamanaka, K.L. and A.R. Kronlund. 1996. Inshore Rockfish Stock Assessment for 1996 and Recommended Yield Options for 1997. Pacific Stock Assessment Review Committee: Working Paper G96-11. Fisheries and Oceans Canada.

Yamanaka, K.L. and L.C. Lacko. 2001. Inshore Rockfish (*Sebastes ruberrimus, S. maliger, S. Caurinus, S. melanops, S. nigrocinctus, and S. nebulosus*): Stock Assessment for the West Coast of Canada and Recommendations for Management. Canadian Science Advisory Secretariat Research Document 2001/139.

Yamanaka, K.L. and L.J. Richards. 1992. Inshore Rockfish. p. 221-266 *In* Leaman BM (Ed) Groundfish stock assessments for the west coast of Canada in 1991 and recommended yield options for 1992. Can. Tech. Rep. Fish. Aquat. Sci. 1866: 304 p.

Yamanaka, K.L. and L.J. Richards. 1993. 1992 research catch and effort on nearshore reef-fishes in British Columbia Statistical Area 12. Can. Man. Rep. Fish. Aquat. Sci. 2184: 77p.

Yamanaka, K.L., L.C. Lacko, J. Lochead, J.M. Martin, R. Haigh, C. Grandin, K. West. 2004. Stock assessment framework for Inshore Rockfish. Canadian Science Advisory Secretariat Research Document - 2004/068.

Zhang, Z. and C. Hand. 2006. Determination of Geoduck Harvest Rates Using Age-structured Projection Modelling. Journal of Shellfish Research 25 (2): 445-453.

Table 1: 'Model fisheries' reviewed in this study and their abbreviations.

Fishery	Abbreviation
Pink snapper (*Pagrus auratus*) in inner Shark Bay, Western Australia	SBPS
Western Australian Dhufish (*Glaucosoma hebraicum*) in Western Australia	WAD
Demersal Shelf Rockfish (*Sebastes spp.*) in Alaska, US	DSR
Goliath grouper (*Epinephelus itajara*) in the Gulf of Mexico, US (pre-1990 closure)	GMGG
Cowcod (*Sebastes levis*) in the Southern California Bight, US (pre-2001 no-retention management)	CBC
Inshore Rockfish (*Sebastes spp.*) in BC, Canada	IR
Red rock lobster (*Jasus edwardsii)* and packhorse lobster (*Sagmariasus verreauxi*) in New Zealand	NZRL
Western rock lobster (*Panulirus cygnus*) in Western Australia	WARL
Southern rock lobster (*Jasus edwardsii*) in Victoria, Australia	VRL
Red sea urchin (*Strongylocentrotus franciscanus*) in BC, Canada	BCSU
Geoduck (*Panopea abrupta*) in BC, Canada	BCGD
Blacklip (*Haliotis rubra*) and greenlip (*Haliotis laevigata*) abalone in Victoria, Australia	VAB
Loco shellfish (*Oncholepas concholepas*) in Chile	CL

Table 2: Biological status of each model fishery.

Fishery	Historical fishery status	Current fishery status		Source
	Previously documented overfishing?	Qualitative fishery status	Quantitative fishery status	
SBPS	Yes	Stock(s) 'rebuilt' in 2 of 3 management areas	Mature biomass above target reference point of 40% of unfished mature biomass in 2 of 3 management areas	Jackson et al. 2005
WAD	Unknown	Currently fishing mortality is greater than natural mortality, thus 'unacceptable', as current fishing mortality 'may not be sustainable'	Unknown	Hesp et al. 2002; St. John et al. 2004
DSR	No	Stock(s) not currently overfished	Unknown	GOA FMP 2005
GMGG	Yes	Stock(s) are 'overfished', with inadequate information to determine spawner-per-recruit	Unknown	SEDAR 2004
CBC	Yes	Stock(s) 'overfished', although 'overfishing' isn't occurring	Below 14-21% of pre-fishing biomass	Piner 2005
IR	Unknown	Evidence of 'excessive fishing mortality'	Unknown	Yamanaka and Lacko 2001
NZRL	Yes	Stock(s) near target biomass in most management areas	Unknown	NRLMG 2005
WARL	Yes	Stock(s) 'close to maximum sustainable yield and breeding stock at or above target levels (MSC certification in 1999/2000)	Above trigger reference point (22% of unfished mature biomass) in all 3 management areas	de Lestang and Melville-Smith 2004
VRL	Yes	Stock(s) 'rebuilding'	Spawning biomass 25% and 17% of available biomass in 1951 (B_{1951}) in Western and Eastern Zones, respectively	Anon 2003; Hobday et al. 2005
BCSU	No	Stock(s) 'stable', and 'no indication of concern'	Unknown	Campbell et al. 2001; DFO 2006a; Campbell, pers. comm. 2006; Campbell et al. 1999
BCGD	No	Stock(s) 'sustainable'	Approximately 75% of pre-fishing biomass	DFO 2006b; Hand, pers. comm. 2006
VAB	Yes	Stock(s) 'stable' and 'fully-fished'	23-41% of pre-fishing biomass	DNR 2002; Dept. of Env. and Heritage 2003

	Historical fishery status	Current fishery status		
Fishery	**Previously documented overfishing?**	**Qualitative fishery status**	**Quantitative fishery status**	**Source**
CL	Yes	Stocks recovering since overfishing during late 1980s	Unknown. Catches stabilized in 1993, lowest level in 1982	Castilla and Defeo 2001

*Fishery acronyms are defined in Table 1 and the MSC refers to the Marine Stewardship Council.

Table 3: Summary of the major regulatory strategies employed in each model fishery.

Fishery	Major regulatory strategies: Harvest strategy and method of implementation	Reference points	Quota allocation scheme/access structure of dominant user-group	Source
SBPS	Constant fishing mortality to specify total allowable catch	Lower reference point on stock size	Limited entry	Jackson et al. 2005
WAD	Effort limit, based on 10-year mean catch level	Lower reference point on stock size	Limited entry	St. John et al. 2004; St. John, pers. comm. 2006
DSR	Constant fishing mortality to specify total allowable catch	Lower reference point on stock size	Limited entry	GOA FMP 2005; O'Connell et al. 2006
GMGG	Constant fishing mortality based on maximum yield	None	None	SAFMC 1983; SEDAR 2004; Atran, pers. comm. 2006
CBC	Constant fishing mortality based on maximum yield	None	None	Butler et al. 1999; Butler et al. 2003; PFMC 2004
IR	Constant fishing mortality to specify total allowable catch	None	Commercial individual vessel quotas and limited recreational entry	Yamanaka and Lacko 2001; Yamanaka et al. 2004; DFO 2006c
NZRL	Area-specific harvest control rules to specify total allowable catch	Lower reference point and target on stock size	Individual transferable quotas	Bentley et al. 2005; NRLMG 2005
WARL	Constant total allowable effort to specify individual effort levels.	Lower reference point on stock size	Individual transferable effort quotas	RLIAC 1999; de Lestang and Melville-Smith 2004
VRL	Decision rule to 'review', 'reduce', or 'hold' total allowable catch	Lower reference point and target on stock size	Individual transferable quotas	Anon 2003
BCSU	Constant fishing mortality to specify total allowable catch	Lower reference point on stock size	Individual quotas	Campbell et al. 1999; DFO 2006a
BCGD	Constant fishing mortality to specify total allowable catch	Lower reference point on stock size	Individual quotas	DFO 2006b
VAB	Maximum constant yield to specify total allowable catch	Trigger and target	Individual transferable quotas	DNR 2002; Zhang and Hand 2006
CL	Constant fishing mortality to specify total allowable catch	None	Limited entry	Castilla and Defeo 2001; Gelcich et al. 2007

*Fishery acronyms are defined in Table 1. Dominant user-group = the user-group with the highest annual catch.

Table 4: Overview of each model fishery in the context of criteria one to three of fishery management success.

Fishery	Critical components for fishery management success (Hilborn et al. 2003a, 2005) (Criteria 1-3): 1	2	3	Source
SBPS	Unknown	TAC for both commercial and recreational sectors and limited entry for all user-groups	Community working group sets fishery objectives	Jackson et al. 2005
WAD	Unknown	No dominant user-group, harvest limited by overall ceiling on fishing boat licenses	Co-management began in 2003	St. John et al. 2004; St. John, pers. comm. 2006
DSR	Unknown	No dominant user-group, open-access recreational sector, and no commercial IQs	Hearings held regarding management plan amendments and levels of optimum yield	GOA FMP 2005; O'Connell et al. 2006
GMGG	Unknown	No dominant user-group, open-access recreational sector, and no commercial IQs	No documented co-management	SAFMC 1983; SEDAR 2004; Atran, pers. comm. 2006
CBC	Unknown	No dominant user-group, open-access recreational sector, and no commercial IQs	No documented co-management	Butler et al. 1999; Butler et al. 2003; PFMC 2004
IR	Unknown	Commercial IQs, yet no dominant user-group and an open access recreational sector	Stakeholder groups have management input	Yamanaka et al. 2004; DFO 2006c
NZRL	Unknown	Commercial ITQs, yet no dominant user-group and an open-access recreational sector	Managed by group representing all stakeholders	Bentley et al. 2005; NRLMG 2005
WARL	Unknown	Dominant commercial user-group with individual effort allocations and a limit on total fishing effort	Management input from Council and sub-committees of stakeholders	RLIAC 1999; de Lestang and Melville-Smith 2004
VRL	Unknown	TAC for both commercial and recreational sectors, and commercial ITQs	Explicit co-management arrangement mandated (1995 Fisheries Act)	Anon 2003
BCSU	Unknown	Dominant commercial user-group with IQs	Explicit co-management arrangement for decision-making, responsibilities, costs, and benefits	Campbell et al. 1999; DFO 2006a
BCGD	Unknown	Dominant commercial user-group with IQs	Consultative management process with a sectoral committee of stakeholders	DFO 2006b

Fishery	Critical components for fishery management success (Hilborn et al. 2003a, 2005) (Criteria 1-3) 1	2	3	Source
VAB	Yes, nearly each bed managed	Dominant commercial user-group with IQs	Explicit co-management arrangement mandated (1995 Fisheries Act)	DNR 2002; Zhang and Hand 2006
CL	Yes, nearly each bed managed	Dominant artisanal user-group with multiple TACs and limited entry	Co-management under Chilean Fisheries and Aquaculture Law since 1991	Castilla and Defeo 2001; Gelcich et al. 2007

*Fishery acronyms are defined in Table 1. TAC = total allowable catch, MP = management procedure, DR = decision rule, LRP = lower reference point, dominant user-group = a user-group which accounts for greater than 90% of total landings, IQ = individual quota, M = natural mortality, MSY = maximum sustainable yield, ITQ = individual transferable quota, TAE = total allowable effort, and B_0 = an estimate of adult pre-fishing biomass.

Table 5: Overview of each model fishery in the context of criteria four to seven of fishery management success.

Fishery	Elements of a Management Oriented Paradigm (MOP) (de la Mare 1998) (Criteria 4-7): 4	5	6	7	Source
SBPS	Management objective clear and measurable	No MP. DR (constant catch, 3 year duration) based on a biomass rebuilding trajectory; LRP present	Annual	Prospective evaluation of harvest strategies	Jackson et al. 2005
WAD	Management objectives not present	No MP. DR based on past catch levels; LRPs present	Irregular (2002)	No prospective evaluation	St. John et al. 2004; St. John, pers. comm. 2006
DSR	Management objectives present but not clear or measurable and not specific to the fishery	No MP. DR present (dynamic TAC set as the product of a fraction of *M* and current biomass); LRP present	Annual	No prospective evaluation	GOA FMP 2005; O'Connell et al. 2006
GMGG	Management objectives present but not clear or measurable and not specific to the fishery	No MP. DR based on maximum yield used as a proxy for MSY; No LRP	2003	No prospective evaluation	SAFMC 1983; SEDAR 2004; Atran, pers. comm. 2006
CBC	Management objectives present but not clear or measurable and not specific to the fishery	No MP. DR from proxy-MSY; No LRP	1999	No prospective evaluation	Butler et al. 1999; Butler et al. 2003; PFMC 2004
IR	Management objectives present but not clear or measurable	No MP. DR (dynamic TAC set as the product of a fraction of *M* and current biomass); No LRP	(Irregular 2001)	No prospective evaluation	Yamanaka and Lacko 2001; Yamanaka et al. 2004; DFO 2006c
NZRL	Management objectives use clear and measurable performance indicators to determine harvest control rules	MP based on area-specific DRs to determine TACs; LRPs present	Annual	Prospective evaluation of management procedure using performance measures	Bentley et al. 2005; NRLMG 2005
WARL	Revised 1999 Primary Management Objective clear and measurable	No MP. DR (based on pot numbers and usage rate) to determine TAE; LRP present	Annual	Prospective evaluation of harvest strategies	RLIAC 1999; de Lestang and Melville-Smith 2004
VRL	Clear and measurable 'strategies' to achieve non-measurable objectives	No MP. DR (based on B_0) and a 'TAC-forum' used to determine TAC; LRP present	Annual	Prospective evaluation of harvest strategies	Anon 2003
BCSU	Management objectives present but not clear or measurable	No MP. DR (MSY-based) to determine TAC every two years; LRP present	Every two years	No prospective evaluation	Campbell et al. 1999; DFO 2006a

Elements of a Management Oriented Paradigm (MOP) (de la Mare 1998) (Criteria 4-7)

Fishery	4	5	6	7	Source
BCGD	Clear and measurable 'management objectives' to attain non-measurable 'biological objectives'	No MP. DR (constant catch based on B_0) to determine annual TAC; LRP present	Annual	Prospective evaluation of harvest strategies	DFO 2006b
VAB	Clear and measurable reference points to achieve non-measurable objectives	No MP. DR to determine TAC (maximum constant yield); LRP present	Annual	Prospective evaluation of harvest strategies	DNR 2002; Zhang and Hand 2006
CL	Management objectives present but not measurable and not specific to each managed area	Management plan for each area. 10% to 25% of exploitable stock harvested in each area; No LRP present	Annual	Prospective evaluation of harvest strategies	Castilla and Defeo 2001; Gelcich et al. 2007

*Fishery acronyms are presented in Table 1. TAC = total allowable catch, MP = management procedure, DR = decision rule, LRP = lower reference point, dominant user-group = a user-group which accounts for greater than 90% of total landings, IQ = individual quota, M = natural mortality, MSY = maximum sustainable yield, ITQ = individual transferable quota, TAE = total allowable effort, and B_0 = an estimate of adult pre-fishing biomass.

Table 6: A scoring rubric depicting the number of model fisheries that were successful/sustainable or not and whether each fishery met the seven sustainability criteria or not. Each table may be read horizontally and vertically.

Criterion 1:

	Fishery met criterion	Fishery did not meet criterion
Fishery successful	2	7
Fishery not successful	0	4

Criterion 2:

	Fishery met criterion	Fishery did not meet criterion
Fishery successful	7	2
Fishery not successful	0	4

Criterion 3:

	Fishery met criterion	Fishery did not meet criterion
Fishery successful	9	0
Fishery not successful	2	2

Criterion 4:

	Fishery met criterion	Fishery did not meet criterion
Fishery successful	7	2
Fishery not successful	3	1

Criterion 5:

	Fishery met criterion	Fishery did not meet criterion
Fishery successful	8	1
Fishery not successful	1	3

Criterion 6:

	Fishery met criterion	Fishery did not meet criterion
Fishery successful	8	1
Fishery not successful	0	4

Criterion 7:

	Fishery met criterion	Fishery did not meet criterion
Fishery successful	7	2
Fishery not successful	0	4

*Each criterion is defined in the text, as are each of the 13 ‘model’ fisheries. Each table sums to the total number of model fisheries (13).

Table 7: **Summary of literature review and synthesis of KS^n fisheries regarding how each criteria of fishery management success may permit or impede the applicability of relatively successful/sustainable management tactics for the IR fishery.**

Criteria 1-7 of fisheries management success identified in Hilborn et al. (2003a, 2005), and de la Mare (1998)	**Management structure or conditions presumably leading to fishery sustainability in KS^n fisheries, corresponding to each criterion**	**Apparently sustainable in KS^n fisheries?**	**'Conditions' applicable in the current IR fishery?**
1	Management at the sub-population scale	Unknown	Challenging due to the presence of multiple user-groups, data gaps, and high in-situ monitoring costs
2	Single dominant user-group with a form of fishing or effort quota	Yes *(WARL, BCSU, BCGD, VAB, and CL)*	Challenging due to the current large open-access recreational and Aboriginal fisheries, along with commercial fishery and high incidental catch of IR in several other fisheries
	Multiple user-groups, each with a form of fishing or effort quota	Yes *(SBPS, DSR, and VRL)*	
3	Co-management	Yes *(all fisheries with the exception of GMGG and CBC)*	Yes, since stakeholder groups (e.g., Groundfish Hook and Line Advisory Committee) currently provide management input
4	Management objective(s) present, clear, and measurable	Yes *(SBPS, NZRL, WARL, VRL, BCGD, VAB, CL)*	Measurability of objectives is possible but challenging due to data limitation
5	Management procedure	Yes (*NZRL*)	Applicability depends on the presence of other criteria (e.g., criterion 4) and components of criterion 5 (e.g., decision rules). Precautionary stock reference points could be employed with limited data
	Decision rules (e.g., harvest control rules)	Yes (all fisheries employed a form of	
	Biological reference points (specifically lower stock size reference points)	Yes *(lower reference points absent only in GMGG, CBC, IR, and CL)*	
6	Regular assessment of stock condition	Yes *(regular assessments absent in*	Yes, but limited due to issues of feasibility and data-limitation
7	Prospective evaluation of management procedure using performance measures	Yes *(NZRL)*	Effectiveness based on the presence of criteria 4,5, and 6
	Prospective evaluation of harvest strategy	Yes *(SBPS, WARL, VRL, BCGD, VAB, and CL)*	Applicability limited based on the presence of criterion 4 and potentially limited due to data-limitation

*Fishery acronyms are defined in Table 1. 'Conditions' = biological (e.g., life-history) or spatial (e.g., organism movement) characteristics of the managed species, 'dominant user-group' = a user-group with estimated landings totalling over 90% of annual landings, and the VRL, BCGD, VAB, and CL fisheries employed clearly stated and measurable means to work toward clearly stated, yet non-measurable, objectives (regarding Criterion 4).