Cod recovery in the north-west Atlantic. Can mother earth pay the (economic) rent? Applying ecosystem-based management.

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Abstract

The method of event ecology is applied to the socio-ecological history of Newfoundland's cod stock collapse and its subsequent failure to recover, despite a near total moratorium on fishing over the past 20 years. An examination of the collapse using event ecology and consideration of the emerging paradigm of ecosystem-based management leads to the hypothesis that ecosystem interactions, either in the form of the continuing low-level cod harvest, or of bycatch from human harvesting of the currently dominant invertebrate species of lobster, crab and shrimp, or of seal predation from the growing harp seal population, are inhibiting a recovery of cod stocks. This hypothesis is then tested through the derivation of a mathematical model incorporating both harvesting and predator-prey interactions, based on the Lotka-Volterra predation equation with logistic growth.

A triangulation of qualitative and quantitative methods is then used to set parameters and solve the equations for the equilibrium sizes of each population at various harvest levels. Potential interventions to allow cod stocks to recover are then modelled using ecological and economic tools to estimate the financial impact of cod stocks recovery. Finally, consideration is given to the governance and incentive structures necessary for sustaining a holistic ecosystem-based management approach for Newfoundland cod, and to financing the transition to a higher-value fishery where cod stocks have become more abundant.

Abbreviations

$ - Canadian Dollar

CCS – Conservation Credits Scheme

EAF – Ecosystem Approach to Fisheries

EBM – Ecosystem-based Management

EU – European Union

ICNAF – International Commission for Northwest Atlantic Fisheries

ITQ – Individual Transferable Quota

MEY – Maximum Economic Yield

MPA – Marine Protected Area

MSY – Maximum Sustainable Yield

MT – Million Tonnes

NAFO – Northwest Atlantic Fisheries Organisation

NGO – Non-governmental organisation

NPV – Net Present Value

OSY – Optimal Sustainable Yield

PV – Present Value

T – Tonne.

TAC – Total Allowable Catch

Chapter 1: Introduction

The high level requirements for sustainable fisheries management are well known, including alignment of incentives, effective governance and adequate enforcement of restrictions. Together, these measures should support long-term value creation, sustainable livelihoods and ecological health (Clover, 2004; Greenburg, 2010).

However, the history of fisheries management is replete with failures to maintain sustainable, productive wild fisheries (Pauly, 1995; Roberts, 2007). In a number of regions these failures have resulted in the collapse and commercial extinction of once productive fisheries, the most famous of which is the northwest Atlantic cod fishery collapse of the early 1990s (Kurlansky, 1997). In this and other cases, it appears that sustainable fisheries management has been prevented by inadequate human understandings of ecological and social complexity and insufficient capacity for transparent implementation of socially optimal governance solutions.

Through a case study of the Newfoundland Atlantic cod fishery, this paper examines some of the tensions and challenges in environmental management in the fisheries context by drawing on the discipline of economics to examine potential welfare gains from various policy interventions.

Conceiving of fisheries within an ecosystem context requires a rethinking of the boundaries and interactions that impact the fishery. Both human and natural factors can exert dominant influences on commercial fish stocks. Whilst the collapse of the commercial cod fishery in Newfoundland is infamous (Kurlansky, 1997; Clover, 2004; Roberts, 2007), the failure of cod stocks to recover, despite twenty years of severe fishing restrictions, has remained puzzling to researchers (Longhurst, 1998; Caddy & Cochrane, 2001; Bjorndal et.al, 2002).

Taking as its hypothesis the idea that cod's failure to recover in Newfoundland is a result of ecosystem interactions, this paper seeks to reconceive the Newfoundland fishery in its broader socio-ecological context, and examines predator-prey relationships at trophic levels both above (seals, human harvesting) and below (shrimp, crab and lobster) the cod. Having created and solved biological models for these species, interventions to support the recovery of Newfoundland cod stocks are costed, along with the potential direct economic benefits that might flow from a recovered cod fishery. The economic value of moving from a 'status quo' scenario to a 'recovered cod' future is then quantified in economic terms to allow for comparison.

Even if cod stocks can recover, and if in doing so, create significant economic value, additional costs will still be imposed as a result of such a large-scale ecological intervention. Financing such measures and ensuring they are aligned with incentives to promote stewardship and sustainable use of the natural capital of Newfoundland remains a challenge, but with some innovative potential solutions.

Report structure

This paper seeks to apply an ecosystem-based management approach to address the causes of the failure of Newfoundland cod stocks to recover.

Chapter 2 Sets out the historical development and economic rationale for various governance mechanisms used for fisheries management. It deals with the economic theory underpinning attempts at improved governance, and sets out the context and meaning of an ecosystem-based approach to fisheries management. The theory of ecological modelling of predator-prey relationships is explained as are the components required for construction of a bio-economic model to calculate the value created or destroyed by intervening in an ecosystem.

Event ecology is introduced as a methodological approach for investigating environmental changes in **Chapter 3**. The socio-ecological history of the Newfoundland cod fishery and the structure of the past and present fishing industry are set out. The paradigm of event ecology is applied to three possible causes of the failure of cod stock recovery: the ecosystem impact of the continuing cod harvest, the ecosystem impact of cod bycatch from shrimp, lobster and crab harvesting, and the ecosystem impact of predation by the growing seal population on cod.

The methodological approaches adopted for detailed analysis are set out in **Chapter 4**. The boundaries of the system studied are explicitly stated, the mathematical depiction of species interactions is devised and triangulation methods of qualitative and quantitative approaches are established, including interviews with industry participants, consultation with scientific experts and a review of scientific literature.

Chapter 4 sets out the ecological and economic results obtained from the parameters of a 'status quo' fishery in the future, as against a 'recovered cod' scenario.

The results are discussed and the challenges they pose addressed in **Chapter 5.**

Chapter 6 draws together policy insights from the analysis in concluding remarks.

Chapter 2: Literature Review

2.1 History & Governance of Fisheries

The French and Spanish languages translate 'sea food' as 'sea fruit': a natural crop that grows every year with no requirement for inputs (Greenburg, 2010). Yet human harvesting has out-stripped nature's ability to renew fish stocks in many marine ecosystems such that a majority of the world's fisheries are beset by some combination of overcapacity, habitat damage, over harvesting or poor economic returns (Clover, 2004). Whilst humans have been harvesting wild fish for millennia, the advent of the industrial revolution meant that marine species began to be affected by multiple top-down, bottom-up, and side-in impacts that did not necessarily act independently of one another (Breitburg et al, 1999). From 1750, interdependent and cumulative impacts from human activity were evident on marine species, and these impacts have dramatically accelerated since the introduction of industrial scale fishing in the twentieth century (Cloern, 2001).

Today, most fisheries are fully or over exploited (World Bank, 2010). Globally, 32% of fish stocks are now overexploited, depleted or recovering from depletion and a further 50% are being exploited at their maximum level, with many of these vulnerable to decline due to poor management (FAO, 2010). The true state of world fisheries could be even worse than depicted in these reports due to the systematic nature of over reporting landed catches from Chinese waters in recent decades (Watson & Pauly, 2001).

Canadian marine resource management regimes have wrestled with the tragedy of the commons (Hardin, 1968) since the 1970s, whereby individuals acting in their own interests over exploit common pool resources.

Globally attempts to regulate by effort, catch and gear restrictions have often proved inadequate, and the last few years have seen declining global wild fish catches (World Bank, 2010). When harvest rates of a single species, Peruvian anchoveta, (which is highly impacted by El Nino events), are excluded from world wild fish catch statistics, a steady decline in global catch from the mid 1980s is evident (Pauly et. al, 2005).

Whilst environmental change is a constant in any living system, in the last few centuries anthropogenic forces have significantly increased both the rate and scale of environmental changes in many ecosystems (Vitousek et. al, 1997). So much so that human influence is now considered the dominant driver of environmental change at a planetary scale (Steffen et. al, 2007). Despite the myriad impacts affecting marine ecosystems, it is still direct human harvesting that is the most significant factor in the collapse of fisheries (Hutchings, 2004).

In recent years, attempts to move beyond single species catch programs and integrate human and broader environmental management objectives have emerged (Roberts, 2007). At the forefront of this new thinking is the ecosystem approach to fisheries management (Pitkitch et. al, 2004) which is a management scheme placing much greater weight on integrating management across different fisheries where ecological interactions are present. Additionally an ecosystem-based approach seeks to integrate the many human factors relying on and interacting with an ecosystem. It seeks to achieve diverse habitats, replete in biodiversity, which are more resilient to shocks and more dependable for the human communities which rely on them (Pitkitch et. al, 2004). The ecosystem-approach to fisheries management will be set out in more detail following a review of historical fisheries governance theory and practice.

2.2 Traditional Management Regimes

Open Access – the economic problem

The concept of economic rent was first set out by Adam Smith (1776) and further developed by David Ricardo (1817). It may be considered to be the payment to a factor in fixed supply (Alchain, 1987). In fisheries, the quantity of supply is not fixed, and in common pool fisheries, the demand may push the supply to the point where there are no economic rents.

In an open access regime, the allowable catch (supply) is not restricted and the rents are competed away as more fish are caught. Competitive pressure and the 'race to fish' (Clover, 2004) will often lead to commercial extinction or collapse of fish stocks as harvesting exceeds the potential growth rate of the species (Schiermeier, 2002). What is rational for a fisher will be devastating to an ecological system and its capacity to produce long-term value.

Given the powerful logic of the tragedy of the commons, tenure security is hypothesized to contribute to proper stewardship of resources (Hardin 1968), investments in sustainability, and also to increased social cohesion and reduced conflicts over resources (Natcher, et.al 2009)

In economic terms, the history of fishing regulation is often portrayed as a failure to grant tenure security (Cunningham et. al, 1985). Competitive exploitation of common property fish stocks is wasteful because each fisherman or enterprise does not take into account how her fishing effort affects the size of fish stocks and thereby their growth and the unit cost of fish. Two solutions are suggested:

1) Establish property rights to fish stocks. If only the owner has the right to fish from a stock, it becomes in the owner's interest to take account of how fishing affects the growth of the stock and unit cost of fishing. However, depending on the rate of growth of the fish and its expected future value, it may be rational for the owner to extirpate the population and invest the funds so raised in another asset to earn a higher return (Grafton et. al, 2007); or

2) Control of the fish stocks by public authorities. The access to fish stocks must be limited in some way in order to avoid the overexploitation resulting from open access. In this model, stocks remain common property in that no single party has exclusive rights to harvest them unless they are granted by the public authorities.

In economic theory, an externality is a social cost not borne by the producer (Cunningham et.al, 1985). In capture fisheries, externalities have been classified into five categories (Seijo et al, 1998).

1) Stock externalities – the impacts of one fisher's activities on the availability of the target species for other fishers in the fishery;

2) Crowding externalities – the impact of vessel aggregation in a fishery on marginal catch costs in the fisheries;

3) Technological externalities – these are similar to stock externalities – but relate to fishing gear impacts on population structures of by-catch species that are targeted species for other fisheries;

4) Ecologically based externalities – this is a broadened concept of stock externalities that considers ecological interactions; and

5) Techno-ecological externalities – the impacts of fishing practices/gear on the broader ecosystem.

Where these externalities aren't managed, overcapacity, over fishing and welfare losses result, reducing the ability of fisheries to contribute to economic development, food security, poverty alleviation and the maintenance of ecosystem services including cultural, social and religious services (Lindegren et.al, 2009).

As policy makers have sought to address these externalities, the following mechanisms have been applied to attempt to reduce the damage created by an open access regime:

Technical Measures

These include indirect limits to fishing efficiency, for example horsepower restrictions, which may reduce the amount of time a vessel is able to spend fishing. Restrictions may also limit the catchability of particular fish by size or aim to eliminate bycatch through net design. Generally these standards are difficult to enforce and fishers have powerful incentives to seek to circumvent them, which frequently occurs (Cunningham et. al, 1985).

Input Controls

Often taking the form of limited licences or individual effort controls, these measures seek to limit the number or capacity of participants in a fishery. Licences may be awarded to vessels with a historic record of participation in a fishery, and will often be circumvented by technological advances which may have the effect of increasing the effective effort employed over time. Individual effort quotas will limit the amount of time a piece of gear or individual unit may spend in a fishery. In the short-run, such measures may reduce the race-to-fish which can arise from seasonal catch limits. However, if these measures are flexible in their combination, 'effort-creep' may occur whereby participants invest in particular combinations of gear to increase their effective effort (OECD, 1997).

Output Controls

Output controls such as total allowable catches (TACs) or individual transferable quotas (ITQs) seek to limit the total quantum of fish harvested. Often bycatch is a significant problem with such systems, particularly where transfer between species quotas is not allowed (Client Earth, 2011). TACs have historically resulted in a 'race to fish' (OECD, 1996). Fishers race to maximise their individual component of the TAC where a share of the TAC is not allocated to individual fishing units. This can result in serious ecological damage and encourages over-capitalisation in the fleet as vessels seek to out-compete one-another (Clover , 2004). ITQs seek to circumvent this perverse incentive by allocating the right to fish a share of the TAC to individual fishing units. ITQs may however create an elite class of producers, and lead to the elimination of small-scale producers, where ITQs are sold in a competitive auction process (OECD, 1996). They also impose a significant enforcement burden as the incentive to under-report can be substantial.

Often hailed as a panacea (Clarke, 1990), the introduction of ITQs has led to stock biomass increases in only 12 of 20 fisheries reviewed over a 30 year period (Chu, 2009). Enforcement and monitoring have been highlighted as crucial to any conservation outcome from ITQ implementation (Chu, 2009). In addition, the socio-economic impact of ITQs has often been to accelerate the consolidation of fishing rights in the hands of a few large private groups, and to push small-scale producers to exit the industry (Eythorsson, 2000).

Enforcement and control of the allowable catch level is critical to the success of any rights-based regime. Costs of monitoring can include the placement of observers on boats or installation of recording cameras (Client Earth, 2011). These costs are often ignored when rights-based regimes are established, and failure to provide adequate enforcement funding may account for some of the failure of these regimes to live up to their promise (Summalia, 2008).

Another relevant issue is the sharing of risk under ITQ type regimes. In theory an ITQ will reduce the risk for fishers, as they are guaranteed a proportional share of the annual catch, regardless of how quickly they catch it. However, if there is no financial mechanism to smooth the impact of sharp reductions in the annual catch, incentives to cheat are enhanced as the risk of adverse ecological change is borne entirely by the fishers.

The initial allocation of quotas is often controversial and past participation and catch has regularly been used as a basis for initial allocations, for example in Canada, New Zealand and Iceland (Robinson, 1986). In other jurisdictions such as Australia, a collaborative process of engagement with the industry was developed to allow a weighting for historic investment as well as catch levels, leading to very high support from the industry and the development of cultural norms opposed to cheating as the allocation was generally perceived to be fair (Robinson, 1986).

An additional issue, often ignored because of the complexity involved, is whether rights should be allocated only to boat owners or also to fishing industry workers and contractors.

Closures – temporary or permanent

An additional measure increasingly under consideration is the closure of areas to fishing, either permanently as marine protected areas (MPAs) or on a temporary basis. Research indicates that the establishment of marine reserves off the Florida coast and in the Caribbean Sea have led to increased catches in surrounding waters (Dayton, 1998). The establishment of a reserve in St Lucia has led to increases in catches by adjacent small-scale fishers of up to 90% (Schiermeier, 2002).

Single Species Measures

With the exception of protected areas, all of these regulatory measures focus on a single species, and often ignore impacts on the wider ecological system, for example predation, bycatch and destruction of benthic habitat by trawling.

Of the five externality types set out above, the first two (stock externalities and crowding externalities) fall under a narrow definition of conventional fisheries management, which seeks to overcome the problems of open access to an individual species. The remaining three externalities involve broadening the traditional fisheries management concept to apply some form of ecosystem-based approach.

2.3 Incentives

In the context of fisheries, anything affecting an individual's choice of action – from the price of inputs and final products, to fines, peer pressure and religious beliefs - may be considered an incentive. In fishery management, incentives for both producers and consumers are often not aligned with efficient, effective and equitable operations and harvesting (Clover, 2004).

Rights based incentives address the implementation of user rights within a fishery, removing open access and providing an economic incentive for long-term sustainability. With an effective mix of user rights, economic theory suggests that the remaining fishing actors may be able to maximise the net present value of resources (Clark, 1990).

Social incentives may derive from community-based institutions and social environments that create peer pressure on individuals to comply with agreed-upon community rules. These are the social mechanisms surrounding access to resources, institutional organisation and decision making, local management and power structures and attitudes and perceptions towards authorities and institutions. They may also include moral and religious codes, fishing traditions and local ecological knowledge.

Aligning incentives and managing complexity

In considering governance measures, it is important not to decide a priori on the rightness or wrongness of a particular management tool, but to develop and build upon an understanding of the biological, sociological, economic and political context in question (De Young & Charles, 2008). The ecosystem-based approach seeks to integrate the local socio-economic context into the broader ecological system sustaining human activity.

Management tools used in isolation would appear to have less chance of being successful than a mix of complimentary tools used in tandem. Managing people may be complicated, but managing fish and ecosystems with their myriad interactions may be even more complex. New tools are necessary in light of the increasing recognition of the uncertainties fisheries managers face (Pauly, 1995; Greenburg 2010). If participants in the fishery are included in management processes and understand why it is in their interest to act in a particular way, the chance of successful implementation of the resulting management intervention is increased (McCurdy, 2011). With the dawning realisation that fisheries management to date has been neither effective nor efficient, both the industry and conservation groups are advocating more participatory governance structures (McCurdy, 2011; WWF, 2011).

2.4 Ecosystem-based approaches

The Ecosystem-approach to fisheries (EAF) has been advocated on the basis of its ability to meet multiple goals including ecosystem health, sustainable resource use and the advancement of human well-being (Charles & De Young, 2008).

As EAF is primarily about managing human beings it is critical to include socio-economic and institutional arrangements in considering a management regime. A new commitment to long-term outcomes and planning may also be needed to achieve the dual objectives of socio-economic benefits and environmental sustainability (FAO, 2003).

The ecosystem approach to fisheries contains three conceptual pillars (Bianchi, 2008):

1) Management of the effects of fisheries on ecosystems (for example trawling and bycatch);

2) Management of the effects of ecosystems on fisheries (for example predation on commercial species, or climate change impacts on abundance and distribution); and

3) Attempts to manage ecosystems through manipulation and targeted interventions.

Essentially, the ecosystem approach to fisheries management seeks to combine two concepts: the conservation of biodiversity, ecosystem structure and functions, and the provision of sustainable food, incomes and livelihoods. In this sense it represents an application of the principle of sustainable development (Brundtland, 1987). The anticipated outcome of applying EAF is an integrated approach to fisheries within ecologically meaningful boundaries (FAO, 2003). To achieve this goal, EAF will need to be integrated with other cross-sectoral approaches such as Integrated Coastal Zone Management.

The key aspects of EAF for the purpose of this paper will be taken to be:

1) The integrated management of multiple fisheries and other ocean uses within a geographic context;

2) The definition of a broader set of conservation objectives to sustain target species and ecosystem structure and functioning;

3) The definition of management areas based on ecological boundaries adjusted for administrative convenience, recognising that a nested approach will be required (Greenberg, 2010);

4) The application of a precautionary approach, where target harvest levels are set at least 10% below the system's average maximum sustainable yield (Rose, 2011);

5) The application of adaptive management systems, allowing observation and experience to feed back into policy; and

6) The principle of participation, enabling stakeholders to be more closely associated with management processes, data collection, knowledge building and sharing (Greenberg, 2010).

2.5 Gap in applying an ecosystem approach to degraded fisheries

A gap in the literature regarding ecosystem-based management appears to exist in that little work has yet been completed in modelling the application of EAF to a degraded fishery. This paper seeks to contribute to this emerging area.

2.6 Governance & Ecosystem-based Management

Applying ecosystem-based management will require reform of fisheries governance, as the EAF requires democracy, transparency and a vision of fairness, equity and sustainability. Incentives for fishers will need to operate with long term and secure rights that account for interactions across stocks, and also a participatory mandate in management.

A participatory management process that supports the goals of setting exploitation levels based on the degree of ecosystem change deemed acceptable, a precautionary approach to both conservation measures and harvest activities, the allocation of rights to natural capital in ways which create incentives aligned with conservation, and transparent decision making are all integral to successful EAF (Sissenwine & Mace, 2001). Additional management resources will also likely be required to support effective enforcement, scientific research and performance evaluation (Sissenwine & Mace, 2001).

Problem of shifting baselines

Across global fisheries, the problem of shifting baselines has been highlighted as a contributor to inadequate conservation measures (Pauly, 1995). Without a baseline to measure change against, however, we might end up managing depressed populations (Myers & Worm, 2003). In addition, understanding the long term equilibrium points of an ecosystem and causes of change will assist in setting meaningful conservation targets, rather than managing what amounts to little more than ecological noise.

To know how many fish there were before fishing began would seem to be a necessary condition to predict how many fish there can be in the presence of fishing. The historical data allows a sense of what can be achieved, and identification of the level beyond which fishing is dangerous to the ecosystem.

A recent study of part of the Canadian Atlantic, involving systematic analysis of archaeological, historical, and recent data on abundance and distribution of species from all trophic levels provides some insight into potential historical baselines (Lotze & Milewski, 2004). It is clear that in Newfoundland, the ecosystem supported from 6,000,000 t to 9,000,000 t with a most likely level of 7,000,000 t of cod stock biomass, roughly 20 times today's level (Rose, 2004a).

2.7 Biological and economic Models for fisheries

The Rev. Thomas Malthus explained the biological concept of carrying capacity by noting that while living populations are capable of logarithmic growth, many of the populations and food sources on which they depend increase algorithmically (Malthus, 1798). As a result, population growth is stymied as the natural limit of food production is reached. The common *logistic equation* (Verhulst, 1838) is a mathematical model based on these insights (equation (1) below):

$$dN/dt = aN(1 - N/K), \quad (1)$$

where N is the biomass density of the population in question, a is its maximum per-capita rate of change, or the intrinsic rate of increase, and K is the equilibrium density, often called the carrying capacity of the environment.

Although this equation may be criticised for its over-simplicity, it remains a central theoretical model in ecology for depicting single-species population dynamics (Berryman, 1992).

The Lotka-Volterra model is an extension of the logistic equation. It entails the shortcomings of the logistic equation but still allows an examination of factors that determine the outcome of competitive interaction (Begon et al, 2006).

There are many models based on predator-prey relationships applied to fisheries. In these models fishing may take the role of predation, with constant values assumed for the natural mortality rate, food supply, and recruitment (Ross, 1910; Baranov, 1918; Volterra, 1926; Lotka, 1932; Schaefer, 1954). In these models the presence of the prey enhances the population of the predator, whereas the presence of the predator reduces the population of the prey. In the Lotka-Volterra model, the predator reduces the prey population, but in so doing, improves conditions for the remainder of the prey allowing for more rapid growth and reproduction.

A criticism often applied to the Lotka-Volterra predator-prey model in the marine context is that it assumes fishing effort is directly proportional to mortality. This is the case only if fish are evenly spread at each population size. Whilst no species is completely evenly spread at each population level, cod exhibit this behaviour more than most other marine species (Beverton & Holt, 1957).

An additional criticism is that growth may not be density dependent. Growth of individual fish is however, quite likely to be density-dependent, as fewer fish in a year class would mean less competition for food, which is likely to make fish grow faster. If the fishing mortality is low, old year classes will be relatively plentiful, causing more predation than otherwise on younger year classes. These growth and mortality effects imply that the yield curve will have lower values for low rates of fishing mortality (fish live longer on average but weigh less, and more of the younger fish are lost through predation.) and higher values for higher rates of fishing mortality.

Problems with the original data used to formulate the Lotka-Volterra model have been demonstrated (Nikolski, 1969). However, in seeking to examine the potential basins of attraction of predator and prey species with harvesting, it would seem to provide a parsimonious approach, in that it simplifies the system to the extent necessary to highlight key relationships, but no further. Using ordinary differential equations, it is possible to solve a coupled predator-prey model to demonstrate natural equilibrium population levels for varying degrees of harvest and predation of each species.

Such a model assumes a steady state, not usually seen in nature. However, the basins of attraction represent the most likely outcome, and provide solutions at least sufficient to test the potential ecological and consequent economic outcomes of a recovered Newfoundland cod stock.

In considering the inputs for a model of ecological interactions involving cod, both bycatch and predation are required. Despite the attraction of multi-cohort models (for cod and its predators and prey), the additional data requirements add unnecessary difficulty in compilation, and needlessly complicate the mathematics. Using the principle of parsimony as a guide, a coupled logistic predator-prey model has been constructed to test the hypothesis that reducing either direct cod harvesting, cod bycatch in the invertebrate fisheries, or predation of cod by seals will be sufficient to restore the cod population.

In a pristine fish stock, the amount of growth and decay, over time, will balance at the carrying capacity (Begon et.al, 2006). But in an exploited fishery, surplus growth is generated to compensate for the amount being fished. The growth rate increases as the stock size decreases up to a point where growth rate is maximised at the maximum sustainable yield (MSY).

As a fishery approaches the MSY, the marginal cost of catching each additional fish increases, and the additional cost of harvesting the final few fish to get to MSY is often significantly more than the marginal value of those fish. Economic theory shows that in most, but not all cases, the maximum economic yield (MEY) will be obtained at a level of harvest less than that which achieves MSY (Grafton et. al, 2007).

The MEY is the level at which the consumer's marginal willingness to pay is exactly equal to the marginal cost of fishing. MEY maximises the sustainable net returns from fishing, and society does the best it can with what nature has provided. MEY coincides with the level of harvest or effort that maximises the sustainable net returns from fishing to the theoretical position where economic rent is maximised.

However, in contrast to MEY, most fishers readily understand the biological reality of MSY, and the target of MSY is written into many fishery regimes, including that for Newfoundland marine resources (McCurdy, 2011). For the purposes of the following analysis, MSY has been used as a basis for calculating economic rents, as calculation of MEY requires knowledge of the margin cost structure of each species' fishing industry, the data for which is not publicly available.

The concept of Optimum Sustainable Yield (OSY) is also used in bioeconomics (Cunningham et. al, 1985). This concept was discarded as it is imprecise. The following analysis is not seeking to finely optimise harvesting, but rather demonstrate the range of potential welfare gains in the form of increased economic rents which may be supported by a recovered cod population.

In calculating economic value, gross profit will be used as a proxy for economic rent, whilst not an exact substitute (as rent is a return in excess of the firm's cost of capital) gross profit is more readily calculable and allows a consistent comparison of different future scenarios.

On Value

In considering the economic value of a marine resource, three components of value are generally included in the economic literature: direct use values, indirect use value and passive use or information values (Furst et. al, 2000). Direct use values include the commercial revenue from harvesting fish as a source of protein for human consumption or other economic activity. Indirect use values include the cycling of nutrients and carbon sequestration in the ocean, in part a function of biomass present (Wilson et. al, 2009). Passive use and information values include the option value of preserving fish stocks for future generations, and the continued existence value of the stock.

For simplicity of analysis, this paper will consider only direct use value as expressed in the price paid for the landed catch. Whilst this is a clearly incomplete capture of the value provided by a fishery, if anything it will understate the economic value provided by recovered cod stocks, and thus allows a conservative basis for comparison with the status quo scenario.

Discounting

Given the time-value of money, economic rents in the future are often discounted to allow comparison with values in the present moment (Stern, 2006). In fisheries economics, discounting can lead to ecologically perverse outcomes, for example the decision that it is economically preferable to extirpate a species entirely, and earn a higher rate of return through some other asset class such as stocks or bonds (Grafton et. al, 2007).

In building any economic model, the discount rate should reflect both the nature of the resource in question (for example, is it a public good?), the opportunity-cost of investment, and the relative riskiness of the project (Stern, 2006). Discount rates applied to fisheries range from 1% to 25% in published literature (OECD, 1997). The World Bank has recently adopted 3.5% as the appropriate discount rate in marine bioeconomics given the particular social, ecological and economic position of fish stocks (World Bank, 2010). A rate above this may therefore be considered conservative in its estimation of future economic gains.

Chapter 3: Event Ecology and the sustained collapse of Atlantic cod

The typical definition of species collapse is where 90% or more of the population is gone (Greenburg, 2010). The current state of Newfoundland cod, having suffered a loss of over 95% of historical population size (Rose, 2004a), is clearly a case of a collapsed population.

The paradigm of event ecology will be applied to examine possible causes of the failure of Newfoundland cod to recover. Event ecology is an approach to environmental problem solving which takes as its entry point to any situation the use of open-ended questions about why specific environmental changes occurred. Event ecology seeks to understand environmental change by making causal connections to prior events, and searching for a causal nexus or chain of events going backward in time and outward in space from effects to causes (Vayda and Walters 1999).

In the Northwest Atlantic, cod occur from inshore shallow water (about 5 m) to the edge of the continental shelf, in water as deep as 600 m (FOC, 2004).

Cod has been fished at a meaningful scale in the North Atlantic for over 1,000 years (Kurlansky, 1997:304). Across the North Atlantic, cod is a key-stone species and top level predator (Savenkoff et. al, 2004; Essington & Hansson, 2004).

Reporting on explorer John Cabot's voyage to eastern Canada, the then Milanese ambassador to London wrote: "The sea there is swarming with fish, which can be taken not only with the net, but in baskets let down with a stone, so that it sinks in the water," (di Soncino, 1497). Other historical accounts also attest to the prolific biomass of cod (Clover, 2004; Greenburg, 2010) which attracted migrants from across Europe from the 16th to 20th centuries (Kurlansky, 1998).

Cod is a slow maturing, long lived fish, with females maturing sexually at about six years of age (FOC, 2004). Spawning occurs over a wide area of the continental shelf and bottom depths from 5m. The Newfoundland cod stocks spawn from March to May along the slopes of the continental shelf in water temperatures of approximately 2.5° to 4°C (Worm, 2003). Whilst each spawner may produce several million eggs, on average only one egg per million succeeds in becoming a sexually mature adult (FOC, 2006a). At first, cod larvae rely on the yolk sac which is attached to their abdomen for nutrients. At about age two weeks they begin foraging for food. Cod diet in the initial stages of life is dominated by juvenile shrimp and gradually expands to become dominated by other fish (Worm, 2003).

European fleets began fishing the Newfoundland cod fishery shortly after the New World was discovered. Early records indicate that prior to 1550, 128 fishing vessels were operating each year in Newfoundland waters (Kurlansky, 1998). By the late 1600s the catch of cod at Newfoundland had reached almost 100,000 metric tons per year (FOC, 2004). By the late 1700s the catch had reached as high as 200,000 t annually. Cod landings during the 1800s ranged between about 150,000 and 400,000 t annually (Rose, 2004).

Following the resumption of offshore fishing with the end of World War II, effort significantly increased with larger trawlers fishing all along the continental shelf and Grand Banks. By the 1980s, devastation to the undersea landscape was widespread, with fishermen noting significant changes to the sea floor as recorded on depth sounders over the course of three decades (Bennett, 1998).

At the same time, the economic structure of the cod fishing industry was shifting, as offshore trawlers overwhelmed the smaller, artisanal fleets with giant factory ships harvesting one population after another. At the time, fishing efforts grew with the support of scientists who claimed that codfish populations couldn't be overfished (Kurlansky, 1998).

Cod's annual migratory pattern involves spending Winter in deep water, then gathering in shoals on the edge of continental shelves at the beginning of Spring before pursuing prey species inshore during Summer. This annual migratory pattern allowed offshore trawlers to continue harvesting cod as the population declined, often preventing the fish from even reaching inshore waters. It was inshore fishermen who first detected the cod population crash (Roberts, 2007, McCurdy, 2011).

Catches of cod from the northwest Atlantic were stable during the 1950s at about 900,000 t, but increased sharply during the 1960s to a peak of almost 2,000,000 t, and declined dramatically during the 1970s to below 500,000 t in 1977 (FOC, 2004). European trawlers during the 1960s carried out a great Winter and Spring fishery on the pre-spawning, spawning, and post-spawning concentrations of cod on the southern Labrador shelf (Rose, 2004a). This offshore fishery affected the inshore fisheries of Labrador and northern Newfoundland by reducing both the quantities and sizes of cod. Eventually the stocks became overfished and there was a collapse of both the inshore and offshore fisheries from their previous levels.

By the 1970s, it was apparent that input controls were not able to prevent ‘effort creep’ which is the inexorable increase in fishing effort and capacity as fishers substitute unregulated inputs for regulated inputs (Wilen, 1979).

In 1973 the major cod stocks were placed under quota regulation. The Total Allowable Catch (TAC) for each stock was based upon scientific advice presented to the International Commission for Northwest Atlantic Fisheries (ICNAF) which later became Northwest Atlantic Fisheries Organization (NAFO).

TACs at first were not effective in curbing overexploitation partly because the TACs were established at too high a level and partly because enforcement was not effective (Rose, 2004b).The major cod fisheries were placed under moratorium in 1992 (Kulansky, 1998) and with the exception of the small northern Newfoundland fishery, have remained closed ever since.

3.1 Environmental change

In seeking to tease out environmental factors contributing to the collapse of cod stocks, a recent study established temporal differentiation between fishing and climate effects on cod stocks (Rose, 2004a). Whilst environmental factors did contribute to lower reproductive rates in the 1980s and 1990s, ocean temperatures are now more favourable to cod reproduction (Bonnell, 2011). Current research does not indicate temperature or salinity are adversely impacting cod reproduction off Newfoundland (Rose, 2011). Rather, a depensatory effect is evident where the population is held below the level at which accelerating growth is possible, because of its dispersal (Rose, 2011). This depensatory effect has been magnified by philopatry, where most offspring return to their natal spawning grounds regardless of distance (Svedang et al, 2011).

Due to both the absence of cod predation and favourable environmental conditions, significant increases in abundance and distribution of concentrations of pandalid shrimp occurred from the 1990s (FOC, 2009). The relative importance of these factors and others is still not well understood (Lilly 2006; Lilly et al. 2000; Parsons and Colbourne 2006; Worm and Myers 2003).

3.2 Ecosystem structure

It is considered likely that trophic changes have led to a semi-equilibrium for Cod, with harvest and predation pressure contributing to prevention of recovery (FOC, 2009).

Following discussions with the industry (McCurdy, 2011; Goodlad, 2011) and to ensure a conservative bias in imagining future fishing scenarios, it was assumed that the invertebrate fisheries were being fished at or below MSY, and thus in reproductive terms were capable of being indefinitely sustained, on average, at current levels of exploitation. It was noted that the crab quotas are not in place as a conservation measure but rather to regulate supply. It thus seems that it is reasonable to contend these species could continue to be exploited at current levels. In addition, the off shore shrimp fishery and entire northern crab fisheries are both certified by the Marine Stewardship Council as being harvested at sustainable levels (MSC, 2011a; MSC, 2011b).

Changes in food-web structures have been implicated in suppressing cod recovery in the north-west Atlantic (Lotze and Worm 2002). For example, some studies have suggested that recovery of traditional groundfish species, including cod, may be prevented by changes in food web structure which are themselves a result of human impacts. Increased low trophic level harvesting may target essential prey species, and destroy essential habitat for both prey and predators (Collie et. al, 2001) The assumption that reduced fishing pressure will automatically lead to recovery of groundfish stocks may be too simplistic (Hutchings 2000).

However, against this view is more recent work which demonstrates that restructured food-webs off Newfoundland as a result of overfishing of cod can be expected to rebound but on a decadal scale (Frank et. al, 2011). Since cod collapsed in the 1990s, the population of forage fish has increased to over 900% of the historic level, and only after 19 years has it outstripped the available food supply of zooplankton (Frank et al, 2011). Absent other shocks or pressures, a lagged recovery in both higher and lower trophic levels can be expected (Murawski, 2010).

It appears that pressures on cod from higher trophic levels such as seals, in addition to human interference, may now be the determinative factors in holding back cod recovery.

3.3 Socio-ecological structure of the fishery

Historically the cod fishing fleet was roughly 50% inshore boats and 50% larger (greater than 65ft) offshore trawlers (McCurdy, 2011). The collapse of cod stocks had a devastating impact on livelihoods in Newfoundland with over 20,000 people losing their jobs (McCurdy, 2011). It also had a detectable impact on levels of mental well being (Bonnell, 2011). The economic void was filled to some extent by crab and shrimp population increases (McCurdy, 2011). Today the industry employs about one third less workers, many of whom have seasonal rather than full time jobs (Bonnell, 2011)

Today the fishing industry is regulated under the Core licensing policy. Of 4,000 Core licenses, 75% are under 45 feet. The crab fishery is operated entirely by owner-operator small boats, as is the lobster fishery. Up until the mid 1990s the shrimp fishery was almost entirely large, offshore factory freezer trawlers (McCurdy, 2001). However, with increasing catches, it is now approximately 40% inshore longliners and 60% offshore factory ships (Bonnell, 2011)

Fisheries and Oceans Canada reports that in the past decade, the small cod harvest allowed in one of the Newfoundland zones has declined from 30,216 t to 14,467 t. The cod price has increased from $1,425/t to $1,684/t. The lobster harvest has increased from 1,759 t to 2,499 t, with lobster prices decreasing from $10,962/t to $7,238/t. The shrimp harvest in Newfoundland has increased from 83,917 t to 89,010 t with prices declining from $2,192/t to $1,451/t. The snow crab catch has declined from 56,278 t to 53,759 t, with landed prices increasing from $4,774/t to $5,909/t (FOC, 2011b). The catch levels, revenues and costs are all set out in Appendix 1, as part of the fishery economic model.

3.4 Ecosystem impacts of cod harvest

The Newfoundland cod fishery remains closed, except for a small catch quota allowed for the northern stock (FOC, 2011b). This allowable catch has declined from 20,216 t to 14,456 t over the past ten years, with current stocks estimated at between 300,000 t and 800,000 t (FOC, 2010). This level of harvest alone, even with depensation, should not be sufficient to hold the population down (Rose, 2011). However, it may be having some impact, and was fed into the modelling work as set out below.

3.5 Ecosystem impacts of Shrimp, Lobster & Crab harvest

Shrimp harvesters are frequently criticised for wasteful fishing and disproportionately large amounts of bycatch (Greenpeace, 2011). However, the north Atlantic shrimp fishery is among the world's best in terms of minimising bycatch (MSC, 2011a). Bycatch levels have been reduced to very low amounts since the mandated introduction of the Nordmore grate in 1993 in response to concerns about the level of cod bycatch. This device allows cod and other fish to escape through an opening in the top of the net (FOC, 2004). Bycatch levels of cod are consistently estimated at less than 1% of the landed catch weight (MSC, 2011a). On an annual catch which has peaked at 130,298 t in the past decade, cod bycatch has not exceeded 1,300 t per annum (MSC, 2011a). This amount was deemed sufficient to warrant modelling work, as set out below.

The bycatch of cod from lobster and crab harvesting is even less than shrimp. Crabs are harvested using pots and lobsters using a combination of traps and long lines (FOC, 2011c). The Newfoundland lobster fishery has been estimated to have cod bycatch in the range of 0.7% to 1% of landed weight (MSC, 2011b). On an annual harvest which peaked at 2,972 t over the past decade, cod bycatch has not exceeded 30 t (MSC, 2011b). The Newfoundland crab fishery does not impact a material level of cod through bycatch due to the low impact harvest methodology (FOC, 2011c).

3.6 Ecosystem impacts of Seals

Seals have been an umbrella species of conservation efforts and remain a poster-child for many environmental NGOs (Earth Trust, 2011; IFAW, 2011; Save our Seal, 2011). Whilst predation by seals has been disputed as a cause of the failure of cod stocks to recover (Sinclair & Murawski, 1997), increasing seal predation has been implicated as a potential cause in the suppression of Newfoundland cod (Winters & Miller 2001).

In 2009, 504 active vessels and about 200 people were involved in harvest of approximately 68,000 harp seals (FOC, 2011a). The number of harvested seals has declined from almost 400,000 just five years ago (FOC, 2006b).

On average, the diet of harp seal in recent years in the Atlantic includes approximately 6% (by mass) juvenile cod (Hammill & Stenson, 2000). Studies indicate harp seals eat approximately 3.2kg per day (Harding, 1992).

In 2010, the Canadian Fisheries Department reviewed the impact of harp seals on cod and found predation was the greatest contributor to increased mortality in large southern Gulf cod (FCO, 2010).

The harp seal population has increased from approximately 5.8 million animals in 2004 - nearly triple what it was in the 1970s – to between 8.5 and 10 million in 2010 (DFO, 2010). The growth in harp seal numbers is set out below in Figure 1

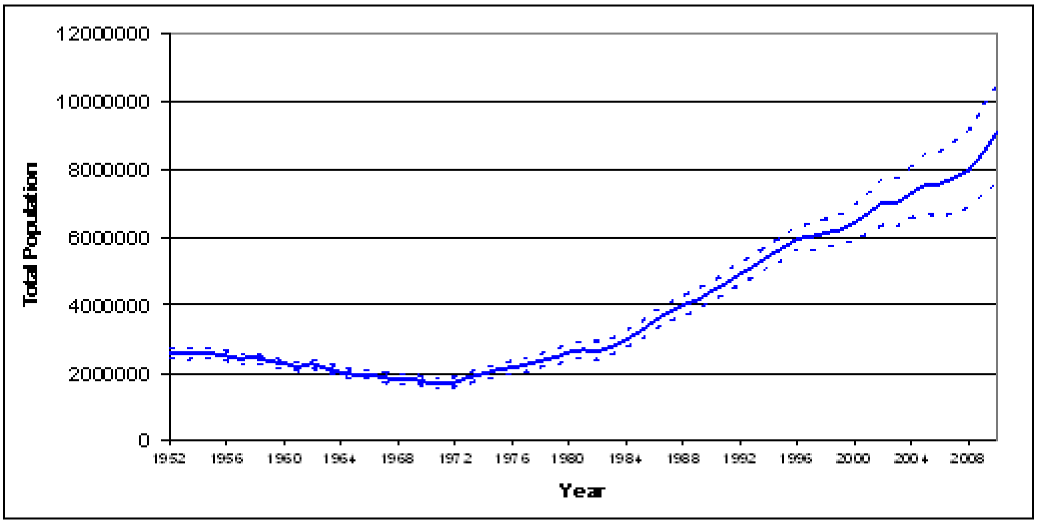

Figure 3 Estimate of total Canadian Atlantic Harp Seal population with 95% confidence limit. Source: (FOC, 2010)

The past three years have seen significant changes in the seal pelt export market, with a decrease in export values from $6.5 million in 2008 to $813,000 in 2010. As a result of a ban on commercial seal products implemented across the EU in 2010, markets in France, Germany and Estonia have closed. China remains an open seal market (FOC, 2011a).

Hutching's (1999) simulations show that the potential population rates of increase of Newfoundland Grand Banks cod are very low, and particularly sensitive to changes in either pre-adult or adult mortality rates. For example, a population of cod maturing at age 4, with an instantaneous adult mortality rate of 0.2 (˜19% per year) was calculated to increase at 15% per year. Increasing this mortality rate by 0.1 (to 26% per year) reduced the population rate of increase to 12% per year. It would appear that seals may be causing sufficient increase in juvenile mortality to hold down the cod population.

3.7 Predation by cod on invertebrates

At its current degraded level, it is unlikely cod stocks are having a major impact on any species through predation. However, a concern of participants in the invertebrate fisheries is that recovering cod could impede the populations they harvest and thereby destroy their livelihoods (McCurdy, 2011). Whilst crab and lobster do not constitute a major part of cod diet (Rose, 2011), one of the principal predators of shrimp is Atlantic cod (Savenkoff et al 2006). Any ecosystem-based approach to managing the Newfoundland fishery must consider predation by recovering cod stocks.

3.8 Management Costs

In addition to considering the direct revenues and costs of harvesting any species, an ecosystem-approach to management will entail additional system level costs. Fisheries management services usually entail the generation of scientific information, establishment and adjustment of management rules, and the enforcement of those rules. In most bioeconomic analysis, these costs are ignored (Summalia, 2008). They would appear to be relevant however and should be considered.

It is possible that in the short run, a reduction in ecological complexity may lead to reduced management costs for a fishery (Schrank et. al, 2003). In a study of the cost of fisheries management in Newfoundland between 1989 and 2000, it was found that the cost of managing had fallen from 20% to 11% of the ratio of cost to landed value. This improvement was attributed to the collapse of cod stocks, and increases in invertebrate harvests (Schrank, 2003). It is therefore reasonable to assume that recovering cod stocks, and the management systems to support them, may require an increase in system-level costs to approximately 20% of the ratio of costs to landed value.

Chapter 4: Methodology

4.1 Bounding the system

To explore the biological and economic consequences of potential strategies for restoring the northwest Atlantic cod fishery both quantitative and qualitative methodologies were employed.

Firstly, the boundaries of the socio-ecological system were defined. In any attempt to examine the workings of a system, the boundaries must be explicitly justified to allow a transparent presentation of relationships within those boundaries, and assumptions which underlie the conception of those relationships (Ulrich, 1987). The ecosystem studied includes the ecological and social interactions, and the economic outcomes which flow from those actions, of the north-western provinces of Canada and Canadian territorial waters.

Given the ecologically separate cod stocks existing in the Atlantic, the distinct Newfoundland stocks (FOC, 2009) were the focus of both modelling and qualitative investigations. The Newfoundland stocks provide over half of Canadian Atlantic crab and shrimp, and the only commercial scale operating cod fishery is in Newfoundland waters (FOC, 2009). Historically, Newfoundland cod stocks produced almost half of the annual cod harvest (FOC, 2005).

4.2 Modelling causes of the failure of cod recovery

From the literature review, it appears that both biotic and abiotic factors have impeded cod's recovery. From both interviews with fishermen and scientific literature, it also appears biological removal of cod, either through predation or harvesting, has been the predominant cause of the failure of cod stocks to recover (Rose, 2004a; Bonnell, 2011; McCurdy, 2011). Biotic factors were investigated using a triangulation of methods (Mikkelsen, 2005). Three possible causes of the failure of cod stocks to recover were investigated: the continued harvesting of cod, cod bycatch from invertebrate harvesting and the predation of cod by seals. Salinity, temperature and other abiotic environmental changes were not modelled.

The relationships modelled are set out below in Figure 2.

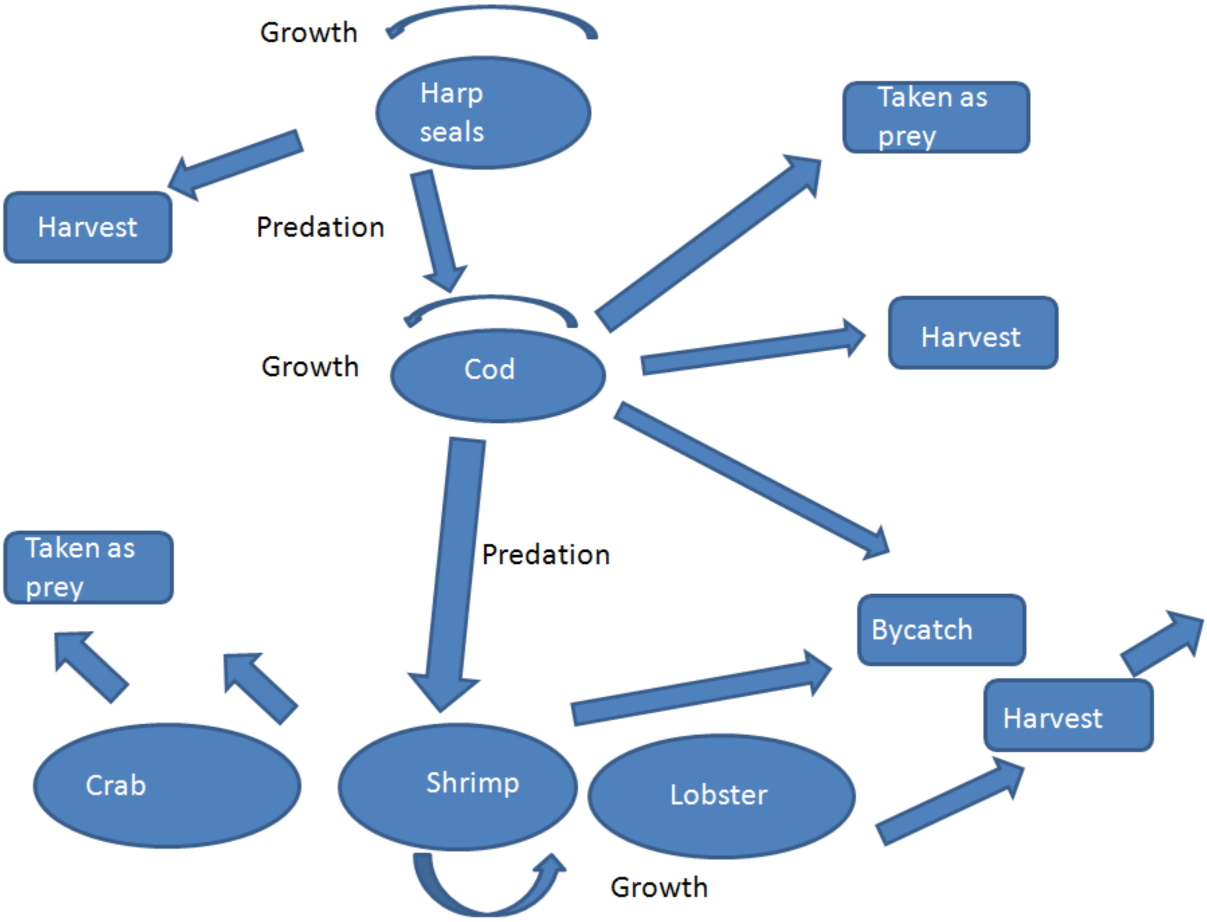

Figure 4 ecosystem relationships examined through coupled equations.

Source: Author's own

A set of biological-mathematical models of the interactions of predator and prey with harvest of both species was devised based on the logistic growth equation and Lotka-Volterra model. The following parameters were used:

N = population of Prey (lobster/crab/shrimp/cod) at time t;

P = population of Predator (Cod/Seal);
a = growth co-efficient for prey;
Kn = carrying capacity of prey;
b = Predation efficiency of Predator on Prey;
Cn = Harvest of Prey species;
c = growth rate of Predator;
f = predator's efficiency at turning food into offspring
e = bycatch factor for Cod with Lobster/Crab/Shrimp harvest
Kp = Carrying capacity of Predator
Cp = Harvest for Predator species

The first pair of equations (1) & (2) seek to establish a logistic population growth profile with constant harvesting, and no inter-species interaction, except through bycatch.

$$\frac{dN}{dt} = aN\left(1 - \frac{N}{K_N}\right) - H_N \qquad (1)$$

$$\frac{dP}{dt} = cP(1 - (P/K_p)) - eH_N - C_P \qquad (2)$$

A second pair of equations was devised in an attempt to represent the addition of a predator-prey interaction. This pair of equations (3) & (4) seek to illuminate the basins of attraction (equilibrium points) for a recovering cod population which may then have an impact on other species through its predation on them, using the Lotka-Volterra relationship.

$$\frac{dN}{dt} = \mathrm{aN}\left(1 - \frac{\mathrm{N}}{\mathrm{K_N}}\right) - \mathrm{bNP} - \mathrm{C_N} \qquad (3)$$

$$\frac{dP}{dt} = \mathrm{cP}\left(-1 - \left(\frac{\mathrm{P - fN}}{\mathrm{K_p}}\right)\right) - \mathrm{eC_N} - \mathrm{C_p} \qquad (4)$$

These equations were solved for the known parameters of the Newfoundland/Grand Banks ecosystem to determine basins of attraction which function as equilibrium points to which the system will trend (at least in this simplified form) under various harvest scenarios for cod, shrimp, lobster, crab and seals. Prior to attempting to solve these equations, expert consultations were undertaken with marine scientists in Newfoundland, which determined predation by even a fully recovered cod population would not be a material factor affecting Newfoundland lobster or crab populations. (Rose, 2011; Bonnel, 2011) The equations were thus solved only for the cod-shrimp interaction. However, as representatives of the fishing industry expressed the view that recovering cod may impact all three stocks, this more conservative level of impact by recovering cod on commercial invertebrate species was included in modelling scenarios.

The initial investigation concerned the effects of reducing bycatch from shrimp harvesting on cod. Annual bycatch of 1% of shrimp harvest by weight was used (MSC, 2009). Bycatch of cod with crab and lobster harvesting is virtually non-existent and for shrimp, the introduction of the Nordmore gate in the 1990s has reduced bycatch to levels considered negligible by world standards (MSC, 2009).

An alternative scenario of increasing seal harvest to reduce cod predation was then considered.

Once a recovered cod population had been established as a possible outcome, in this case approximately ten years after the intervention commenced, the effect of predation by cod on shrimp was considered.

Data on carrying capacity, predation effects, likely sustainable yield and other parameters was derived from the published literature (Harding, 1992; Kristjansson, 2001; Rose, 2004a) and consultations with experts from the Centre for Fisheries research at the Fisheries and Marine Institute, Memorial Univeristy in St John's Newfoundland, Canada (Rose, 2011; Bonnell, 2011), NGOs (WWF, 2011, MSC, 2011a; MSC, 2011b) and the fishing industry (Goodlad, 2011; McCurdy, 2011).

The historic population over 500 years for Newfoundland cod has been estimated at a consistent average level of approximately 7 million tonnes (Rose, 2004a). This figure was used as carrying capacity, with current stock levels of spawning biomass estimated at approximately 350,000 t (FOC, 2009).

Variational methods of solving the equations were applied in both excel and matlab, holding all parameters constant but varying the harvest size and fishing effort, to allow sensitivity analysis.

To inform and contextualise the mathematical simplification of the ecosystem, a literature review of the application of ecosystem-based management was undertaken, and interviews conducted with industry participants, fishing union representatives and marine biologists specialising in the north-western Atlantic ecosystems.

A methodological triangulation of quantitative and qualitative data analysis (Mikkelsen, 2005) to determine key parameters was complemented by consultation with ecological and socio-economic experts in cod recovery. The results for likely species interactions and biomass outcomes were then used as input data into a simple economic model seeking to quantify gross profit and economic rent produced by the fishery as a whole in either a 'status quo' scenario or a 'recovered cod' scenario.

Once the impact of recovered cod stocks on the shrimp industry had been investigated, the predation impact of seals on cod was modelled. Equation (1) was applied to cod with predation and harvesting from the seals, to find the level to which seal predation would need to be reduced, holding other variables equal, for cod to recover. The parameters were taken from published literature on cod and seal biology, and interviews with fisheries scientists (Davies & Rangeley, 2010; MacKenzie et. al, 2011; Rose, 2011; Bonnell, 2011). These results enabled calculation of a time-frame for cod to recover to MSY levels, at which the seal population would not deplete it (Rose, 2004).

4.3 Economic valuation

An economic valuation model was then constructed using price and cost data for the Newfoundland fleet. Initially cost data was sought from the industry. However much of this data is confidential and not able to be released. Published data by Canada's Department of Fisheries and Oceans was used instead (FOC, 2004).

To calculate revenues for landed catch of cod, lobster, crab and shrimp into the future, price increases of 3% per annum over the average 10 year price were assumed. This appears reasonable given anticipated global population growth with world population expected to reach 9.22 billion by 2072 (UN, 2011).

Average fixed and variable cost data per unit of catch were applied across the species of cod, lobster, snow crab and shrimp. Given the price volatility evident in the past decade (FOC, 2009) a ten-year average price and cost level were calculated for each species and then used to estimate both revenue and gross profit. These were then discounted at both 5% and 10% to provide a net-present-value (NPV) for each harvested species in 2011 and in perpetuity.

To calculate fishery management costs, recent Newfoundland studies were applied, which demonstrate that the decline of cod has led to a simpler management regime with concomitant cost savings. In moving from a 'status quo' scenario to the additional ecological complexity inherent in a 'recovered cod' scenario, costs were assumed to increase from their current 11% of gross profit to 20% of gross profit (Shrank, 2003). This estimate was used to calculate management costs to be deducted from the fishery NPV under each scenario.

Given the significant required increase in seal harvesting suggested by the coupled predator-prey model, and the recent closure of European markets to Canadian seal products, it was considered appropriate to use a zero value for revenue from any increase in seal harvesting, and to maintain costs as a proportion of harvest – thus ignoring any possible economies of scale – given the increase in seal harvest would be a one-off cull rather than a sustained increase over time, thus limiting the capacity for reduced costs through learning efficiencies.

Modelling an increase in seal cull was taken to be purely a financial cost, with no additional revenue to offset it.

The total value of seal pelts and other products in 2005-06 was CAD$34m (FOC, 2011c). Assuming a 10% net profit for the sealers would imply the cost of harvest was CAD$30.6m. This implies a cost per seal of $83.84.

Assuming costs were actually \$100 per seal, the cost of culling 2.5m seals would be \$250 million.

To compare the status quo future scenario against a possible recovered cod future, a perpetuity value was calculated for each of the four commercial food species – lobster, cod, shrimp and crab, using an average value over the past ten years, with sensitivity analysis conducted using a variety of discount rates from 5% to 10%.

The perpetuity value is the sum of all revenues or gross profits, discounted at the requisite discount rate. It is calculated using the perpetuity formula (PV = D/r-g) to give a value of the perpetuity in 2021 using an annual growth rate of 3%. In the perpetuity formula, D = the cash flow, r = the discount rate and g = the rate of growth of the cash flow.

To make explicit the trade-off between present and future rents, the annual profit of the marine harvest and a perpetuity value of each species was discounted by between 5% and 10% per annum and these discounting factors were applied to all future revenues and profits.

Discount rates of 5% and 10% were considered appropriate given the nature of fishery resources. A discount rate higher than that applied to climate change action would seem appropriate, since there is no alternative to climate, but there is an alternative to fish (at least for the primary financial value, which is nutrition). The discount rate applied in the Stern Report is less than 2% (Stern, 2006). Yet fisheries are finite renewable resources, with additional, substantial non-financial value. In addition, discount rates above natural reproductive rates can lead to the economically rational decision of extirpating a species (Grafton et. al, 2007). It is likely that a discount rate of approximately 5% would be appropriate. A 10% discount rate was also employed to provide a lower valuation bound.

Chapter 5: Results

The initial hypothesis was that an ecosystem approach required reducing the catch of invertebrates to reduce bycatch of cod. As set out above, lobster, crab and shrimp harvesting are all managed sufficiently well to effectively reduce bycatch to non-significant levels. The model bore out no material impact on cod from a reduction of bycatch to zero, which could only be achieved through reducing commercial shrimp catches to near-zero levels. Bycatch has no material impact on cod at less than 2.5% of current allowable cod catch of ~20,000 t, and less than 0.25% of the estimated annual seal predation level of 200,000 t. However, predation by seals does appear to be having a meaningful impact.

5.1 Predation by seals

Seals currently consume approximately 200,000 t per annum of juvenile cod. With a current estimated biomass of approximately 800,000 t, the cod population is prevented from growing at an average rate greater than harvest and mortality rates with a total predation and harvest take of this magnitude (almost 220,000 t). The average equilibrium harvest rate for this population size is approximately 240,000 t and is at risk of being exceeded. This is congruent with a population failing to grow and recover.

Assuming a present day harp seal population of approximately 10 million, two seal culling levels were examined. A 50% reduction of the seal population and cod consumption, and a 25% reduction of seal population and cod consumption.

Reducing the seal predation rate on cod by 50% to 100,000 t per annum, leads to full recovery in 11 years and population at the size where growth is maximised (MSY) in 8 years. Reducing the seal predation rate by 25%, perhaps more realistic, leads to recovery at MSY level after 10 years and carrying capacity after 13 years. A cod harvest at a precautionary level of less than 10% carrying capacity, for example 550,000 t, could commence sustainably after ten years.

5.2 Close down the remaining cod fishery?

Solving the equations demonstrated that the current cod harvest (<20,000 t) is not the primary cause of the failure of cod to recover. If the seal population (and consequent predation) is reduced by 25%, and the operating cod fishery is also closed, the year of recovery of cod stocks only comes forward by a single year. There is therefore no meaningful gain from closing the remaining cod fishery.

5.3 Impact of recovered cod on shrimp harvest

The basins of attraction solved from equations (3) and (4) as applied to cod and shrimp stocks off Newfoundland are set out below in Figure 3

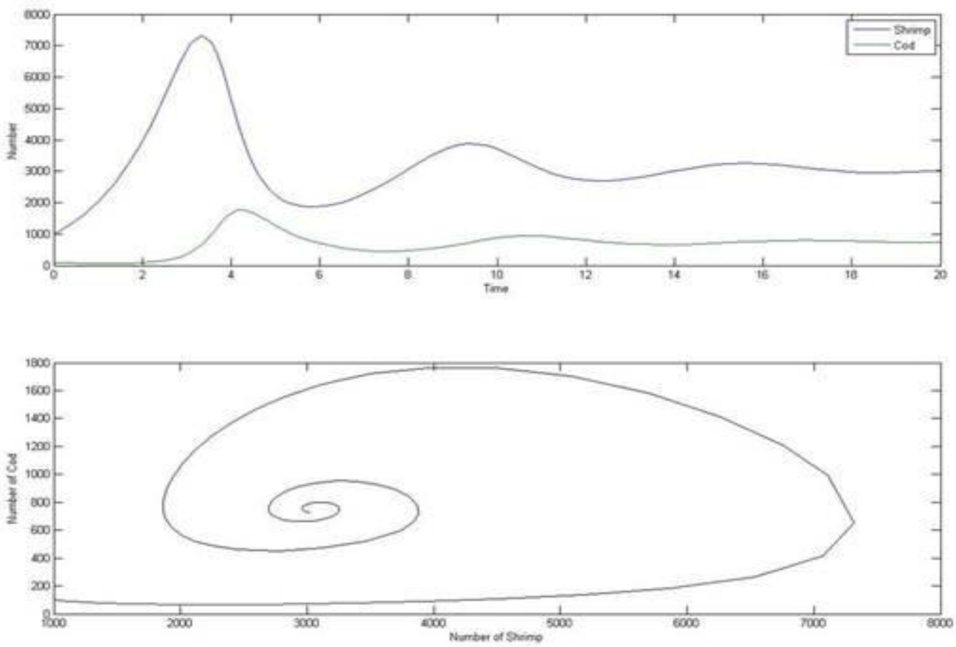

Figure 5 Equilibrium solution for Equations (3) & (4) applied to cod and shrimp

Source: Author's own

Using a simulation of population dynamics over discrete time periods, the shrimp population collapsed at a harvest rate above 35,000 t with cod predation, and the equilibrium population level tended towards just over 3,000,000 t of shrimp spawning stock biomass. This implies a maximum sustainable harvest level of 35,000 t per annum for shrimp, once cod stocks recover. The predation by cod effectively forces the current harvest of ~115,000 t to be reduced by almost 70%. This is not dis-similar to the result expected by the fishing industry, of at least a 50% reduction in shrimp harvest with any meaningful cod recovery. Citing the experience of the interplay between shrimp and Pollack populations in waters off Alaska, the fishing industry suspects a decrease of 50% of shrimp with a recovering cod population (McCurdy, 2011).

Colonial fishery accounts record impressive lobster, crab and cod harvests co-existing in North America (Ames, 2004). Today's published literature and interviews with marine scientists demonstrate that a recovered cod population is unlikely to have a material impact on lobster and crab populations, due to negligible predation (Rose, 2011). However fishermen hold the view that a 10% reduction in catch is a likely scenario as cod recovers (McCauley, 2011). This impact was included in the economic model to ensure a conservative bias, and in an attempt to integrate local ecological knowledge. The MSY harvest levels produced by the model and these sources under a 'status quo' and 'recovered cod' scenario are set out below in Table 1

5.4 Ecosystem results

With long term average carrying capacity of 7 million tonnes, the model demonstrates that cod can support a harvest of 550,000 tonnes per annum when the population is held at or above the MSY level of 3.5 million tonnes. The expected decreases in available harvest for invertebrate species under such a 'recovered cod' scenario are set out below in Table 1. This result is consistent with Canadian government modelling which suggests a sustainable Newfoundland cod harvest of 600,000T per annum should be achievable if cod were to recover (McCurdy, 2011).

Species	Annual Catch (MT) 2009	Average Annual Catch (MT) 2021 (Status Quo)	Average Annual Catch (MT) 2021 (Recovered Cod)
Lobster	2,499	2,659	2,393
Cod	14,467	16,009	550,000
Crab	53,759	50,129	45,116

Shrimp	98,010	114,879	35,000

Table 1: 2009 and projected 2021 catches under 'status quo' and 'recovered cod' scenarios

Source: Author's own

5.5 Economic results

In the year 2021 (once cod has recovered) the total revenue generated for each scenario is set out below in Table 2:

Species	Revenue (in 2021 $) (Status Quo)	Revenue (in 2021 $) (Recovered Cod)
Lobster	$37.1m	$33.4m
Cod	$29.9m	$1,027m
Crab	$246.7m	$218.8m
Shrimp	$225.7m	$68.9m

Table 2: Projected 2021 nominal revenue under 'status quo' and 'recovered cod' scenarios

Source: Author's own

Profit and Perpetuity values in 2011 dollars for the Status Quo scenario are set out below in Table 3:

Species	Annual Catch	Perp. Revenue @ 10% DR	Perp. Revenue @ 5% DR	Perp. Profit @ 10% DR	Perp. Profit @ 5% DR
Lobster	2,659	$395m	$1,381m	$120m	$421m
Cod	16,009	$427m	$1,495m	$103m	$359m
Crab	50,129	$2,622m	$9,179m	$393m	$1,377m
Shrimp	114,879	$2,399m	$8,398m	$360m	$1,260m
Mgmt	11% of gross			$107m	$375m

Costs	profit				
Total NPV				**$869m**	**$3,041m**

Table 3: Projected revenues and profits in perpetuity at 5% and 10% discount rates for 'status quo' scenario.

Source: Author's own

Perpetuity values for the Recovered Cod scenario are as set out below in Table 4:

Species	Annual Catch	Perp. Revenue @ 10% DR	Perp. Revenue @ 5% DR	Perp. Profit @ 10% DR	Perp. Profit @ 5% DR
Lobster	2,393	$355m	$1,243m	$108m	$379m
Cod	550,000	$5,658m	$31,535m	$1,358m	$7,568m
Crab	45,116	$2,326	$8,141m	$349m	$1,221m
Shrimp	35,000	$732m	$2,562m	$110m	$384m
Seal Cull	~2,500,000	-$250m	-$250m	-$250m	-$250m
Mgmt Costs	20% of gross profit			$385m	$1,911m
Total NPV				**$1,290m**	**$7,392m**

Table 4: Projected revenues and profits in perpetuity at 5% and 10% discount rates for 'recovered-cod' scenario.

Source: Author's own

The cod value only commences ten years after the intervention, so the perpetuity value calculation is completed for 2021 and then discounted back to 2011 at both 5% and 10% discount rates. The seal culling costs are incurred over the first 12 months of the intervention and thus not discounted.

Management costs are calculated as 11% of gross profit under the 'status quo' scenario and 20% of gross profit under the 'recovered cod' scenario. Gross profit in the 'recovered cod' scenario excludes the cost of the seal harvest, as to net-off the seal cull would reduce management costs.

Chapter 6: Discussion

6.1 Economic Gains

The size of potential economic gains from supporting cod recovery is truly staggering. In 2021, revenue from cod harvesting could be over $ 1 billion, as against $29 million today. Given predation by the recovered cod, shrimp revenue would drop from $226 million to $69 million, with crab and lobster revenues also declining to $219 million and $33 million respectively. Nonetheless, in 2021 the Newfoundland fishery could be producing revenues of over $1.3 billion as against $540 million under a status quo scenario.

It must also be noted that in the absence of a keystone predator (cod) the probability of maintaining the current elevated harvest levels of invertebrate species of shrimp, lobster and crab is less than it would be for any of these species in a recovered system (Rose, 2011).

It is also important to note that income from cod harvesting is regular, and year-round, as is the harvest, whereas much of the crustacean harvest activity is seasonal (McCurdy, 2011). In addition, cod harvesting and processing are both labour-intensive compared to crustacean species (Bonnell, 2011). It would appear that employment generated from a recovered cod fishery would be both more reliable and would employ more people than today's invertebrate fisheries.

The most important result is that the net value of production has more than doubled in the recovered cod future. In 2021, the recovered cod fishery looks to be a great asset to Newfoundland. To consider its value in 2011, revenues and profits must be discounted to account for the time value of money.

The model demonstrates that at a 5% discount rate, which seems appropriate for the fishing industry, the value of the average harvest under a 'recovered cod' scenario into perpetuity is $7.39 billion. This compares to a value of the 'status quo' scenario into perpetuity of $3.04 billion. The gap between the two values remains significant even when the discount rate is increased to 10%. In this case the 'recovered cod' scenario yields a perpetuity value of $1.29 billion as against the 'status quo' perpetuity value of $869 million.

These differences are meaningful, and occur after the deduction of a one-off cost of $250 million in 2011 to finance the seal cull, and also include an increase in system management costs from 11% of gross profit margin to 20%. The economic case for applying ecosystem-based management to support cod recovery appears clear.

6.2 Ecological benefits

Whilst increasing and diversifying sustainable seafood production is a sound ecological and economic goal, culling seals is likely to be a controversial process, given the strong opposition to seal harvesting evident today (Earth Trust, 2011; IFAW, 2011; Save our Seal, 2011). However, it makes compelling ecological sense. It is entirely consistent to conceive of a non-governmental organisation (NGO) such as the Marine Stewardship Council certifying sustainably harvested seal products. Unfortunately it appears that as charismatic mammals with evocative faces, seals are afforded a special place in human concepts of wildlife. The recent EU decision to ban the import of commercial seal products may be a product of this effect, and would need to be overcome if revenues were to be generated from the seal cull. Nonetheless, and even in the absence of any revenue from a seal cull, the costs of an intervention of the magnitude envisaged would be more than compensated for by the first cod harvest, ten years after the cull.

6.3 Integrating inshore fishermen and local ecological knowledge

A simple demonstration of potential welfare gains will not of itself achieve reform in governance and the buy-in of the industry is critical. If the offshore cod fisheries of Newfoundland are restored, only to suffer the vagaries of industrial fishing again the mistakes of the past will haunt the future of the people and ecology of the west Atlantic.

The migratory patterns of cod, forming offshore in shoals before moving to inshore waters in pursuit of prey during Summer mean that inshore fishermen will likely detect population shifts first. Inshore fishermen need to be included in fishery management decision making, as they are uniquely placed to feed-in local ecological knowledge. Their inclusion would be a new development in cod stock management and could substantially improve the ability of regulators to detect emerging trends in population dynamics.

One of the causes of past overharvesting was scientific advice that did not integrate feedback and information gathered by local inshore fishermen (McCurdy, 2011). Despite close and productive relationships between the fishing industry in Newfoundland and fisheries scientists and regulators in the crab and lobster industries, no such relationship exists in respect of cod (Bonnell, 2011). In fact, relations between the industry and scientists at the Canadian Fisheries and Oceans Department have been characterised as hostile both before and after the collapse of cod (McCurdy, 2011). Even today, the perception from the Fish and Allied Workers' Union is that cod scientists do not take input from local fishermen seriously (McCurdy, 2011).

Despite this tension, local inshore cod fishers in Newfoundland have been collecting systematic local ecological knowledge for the past 15 years in the form of the Cod Sentinel Fishery. This program involves 74 sites around Newfoundland where six nets are cast at each site. In total a few hundred hooks are set in a pre-determined fashion and three additional nets are cast to suit the conditions (McCurdy, 2011). The fish captured are examined for biological fitness, age and fecundity. This program has now amassed a 15 year dataset which would be a useful input into the cod recovery strategy.

A more sustainable governance regime would give a voice to the small scale artisanal fishermen who are deeply invested in the area and fish they catch, and hold both formal and informal local ecological knowledge of sub populations. A nested system of scientific and industry input into management decisions has been advanced, based on the federal government structure (Ames, 2004).

Including inshore fishers in the management process for cod would appear particularly important given the migratory patterns of cod mean that inshore fishers are the first to detect negative trends.

Such a model operates today in the Maine lobster fishery. 7,000 fishers are divided into community unity, each one having intimate knowledge over a local ocean area. To gain a licence, fishers must be local residents, with evidence of long-term commitment to their particular stock (Greenburg, 2010).

Another useful precedent is offered by an initiative from Scotland. The Responsible Fishing Conference involves fishers giving professional input to scientific studies in an attempt to build trust between the two groups and informal connections. Fisher participants indicated overwhelmingly that they are now less suspicious of fishing scientists (Schiermeier, 2002).

6.4 Sustaining Ecosystem Based Management

Even in healthy gladiform populations, the fecundity of cohorts can vary by as much as 50% in a year (Rose, 2004a). Additional tools will be required to ensure cod stocks are managed through this variability in a sustainable fashion. In Scotland, a very recent initiative is the Conservation Credits Scheme (CCS). The CCS uses temporary closures of areas where spawning cod congregate. It requires fishers to provide real time data regarding catch rates. When catch rates indicate the stock is in a vulnerable phase or level, a real time closure is triggered (Goodlad, 2011). Such a scheme may become more important as climate change brings more rapid environmental changes.

6.5 Integrating licensing

In ownership of boats and participation in the fishery, there is some overlap between the shrimp fishermen and the former cod trawl fishermen (McCurdy, 2011). The integration of fishing rights for these two species may help to mitigate the costs borne by individual fishers as cod stocks increase and shrimp stocks decline, and also better align incentives.

6.6 Marine Protected Areas

One of the problems with individual transferable or non-transferable catch quota shares (ITQs) is that even when they are successful at reducing unsustainable harvesting of the target species, they do not explicitly include consideration of broader ecosystem effects. Bycatch and benthic habitat destruction are two impacts which are absent in ITQ regimes, and yet both have critical ecological importance (Smith et al, 2009).

It seems clear that ITQs are insufficient to achieve effective ecosystem-based management. Protection of habitat through marine zoning and protected areas, as well as a precautionary approach to benthic disturbance through a requirement that operators demonstrate gear is not damaging to the benthos would provide a valuable additional management tool (Dayton et al. 1995, Fogarty 1999, Palumbi 2001). Reversing the burden of proof so that an offshore trawler wishing to use bottom-disturbing gear must first demonstrate that it would not affect the breeding habit of important species (either commercial species or their prey) could help to provide the additional protection the Newfoundland ecosystem will require to reach and retain its peak productivity.

If overexploited fish populations are to rebuild to harvestable numbers, it is likely that fish breeding grounds and nursery habitat must be reserved as safe havens. Today 6.8% of terrestrial Canada is set aside in protected reserves, but less than 1% of Canadian territorial waters are so protected (Earth Trends, 2003). Establishing a similar marine reserve structure to that existing on land would support breeding habitat for stock recovery and provide an additional buffer against the inherent uncertainty catch and effort control systems.

6.7 Financing ecosystem-based management

Moving to a recovered cod state will impose significant up-front costs, not least being approximately $250 million in the first year of the intervention to fund a seal cull. In an ideal world this kind of investment in natural capital would be available from the public purse. However, in an age of austerity, additional public funding of this magnitude may not be forthcoming. If this and other costs are viewed as an investment, the returns from increased cod catches will provide a very significant payback, though only after approximately ten years.

This reality encapsulates the difficulty in attracting private finance to fisheries restoration. Investors typically require credit histories, reasonably predictable cash-flows and an understanding of the risks involved in any investment (Kiernan, 2009). Where financial instruments have succeeded in attracting private capital to fishery restoration, the projects have been small scale (Manta Consulting, 2011). Any large scale private-sector investment will necessarily require the appropriate institutional and governance structures as a pre-condition (Rands et.al, 2010).

One possible solution is a kind of public-private financing model, where loans are provided, secured against a levy on future catches. This form of instrument has been proposed by conservationists and aims to replicate the success of student loan programs where debts are only repayable once income exceeds a certain level (Rangely & Davies, 2011). Under this model, loaned funds would pay for adjustment costs, such as reduction in shrimp catch, the seal cull and increased management costs if required, while ensuring financial security for fishers during the transition. The resultant sustainable fishery with much higher economic value is then able to generate profits, a portion of which are set aside to repay the transitional finance costs (Rangeley & Davies, 2011).

6.8 Economics of the model

Use of MSY

The model consciously adopted maximum sustainable yield (MSY) rather than maximum economic yield (MEY) as a basis for calculation of the revenue and gross profit figures in each scenario. Whilst MEY provides a theoretically optimal economic solution, in practice, biological models need to use the threshold of MSY to ensure the species in question can rebound from the harvest level applied. There is so much uncertainty in ascertaining MSY that to seek further optimisation in the form of MEY seemed inappropriate.

The other assumption of the model that should be made explicit, is that MSY harvests should also be avoided. The harvest is set at least 10% below MSY in performing economic calculations, to reduce the risk of exceeding the MSY in practice, given the variation inherent in a marine ecosystem. A degree of additional precaution, as evidenced by this approach, would seem to be a necessary component of ecosystem-based management.

Discount Rate

As fish are a social resource, a lower than commercial discount rate would seem appropriate (Stern, 2006). A range of discount rates from 5% to 10% was applied to test the sensitivity of the economic results and serve as a proxy for the inherent risks in any biological recovery venture. The higher the discount rate applied to the model, the smaller the gap between the expected value of future revenues from a 'recovered cod' scenario as compared to a 'status quo' scenario. This is because the 'recovered cod' scenario does not generate any revenue or rent until ten years have passed from the imposition of the intervention costs of the seal cull. In contrast, the 'status quo' scenario is generating substantial rents today.

In modelling fisheries, the World Bank uses a discount rate of 3.5% (World Bank, 2010). In this analysis, a 5% rate was used as a proxy for the uncertainty inherent in the recovery action. If the rate is deemed too high, and the same rate as the World Bank uses is applied, the gains from intervention are only magnified.

6.9 Climate change

The model and analysis used has assumed a relatively stable ocean environment in the northwest Atlantic. However, it is clear that as a result of increasing atmospheric greenhouse gas concentrations, the global ocean is freshening (Antonov et al., 2002) and warming (IPCC, 2007).

Reduced salinity and accelerating temperature change may adversely impact the fecundity of cod, along with many other marine species (MacKenzie, 2007; Koster et. al, 2009)

As a result of increasing greenhouse gas emissions, the ocean is also acidifying, which will further adversely impact cod reproductive success (Rogers & Laffoley, 2011).

All of these impacts are expected to worsen in coming decades due to atmospheric forcing (IPCC, 2007). However, a recovered cod biomass stock may be a valuable means of increasing the ability of the ocean to act as a carbon sink (Wilson et.al, 2009). Fish produce calcium carbonate, a substance which reduces the acidity of sea water, and are responsible for up to 15% of total calcium carbonate found in the ocean (Wilson et. al, 2009). Increasing the biomass present in the ocean by supporting an increase in cod stocks may thus assist in producing additional capacity climate change mitigation.

6.10 Future research

Despite consistent mathematical attempts to quantify ecological interactions over the past two centuries, many of the causes of variation in fish and crustacean growth rates and abundance remain poorly understood. Further research on the system-level interactions of species in the northwest Atlantic would provide powerful validation to the apparent value of restoring a healthy cod fishery. Fisheries scientists are currently working on models of this form, but to date, none have been published (Bonnell, 2011).

Chapter 7: Conclusion

The degraded state of the Newfoundland cod fishery represents a good opportunity for the application of ecosystem-based management. Intervening in the Newfoundland marine ecosystem to cull 2.5 million seals and thereby support recovery of a viable cod stock towards its historic biomass of seven million tonnes is consistent with an ecosystem-based management approach and would generate substantial economic rents.

An integrated assessment of predator-prey interactions with harvesting demonstrates that the cod population is being held down due to predation from a seal population now four times its size of 50 years ago. Culling the seal population to support cod recovery will also impact cod prey species, in particular, shrimp.

When the expected costs and benefits of an intervention to allow the cod to recover are quantified and compared to a 'status quo' future they range from $4.3 billion to $421 million (at a discount rate of between 5% and 10%.) in additional economic rent for the Newfoundland fishery.

Intervening in this way requires a reconceptualisation of conservation and ecosystem management goals to prioritise the production of target species harvests and maintenance of ecosystem structure and functioning. Adaptive, participatory management systems and a catch set below the estimated MSY level will help to ensure that incentives in the restored fishery are aligned with long term sustainable use and local ecological knowledge is harnessed to support good management.

The collapse of Newfoundland cod is justly infamous as an example of fisheries management failures. Its recovery would provide a powerful example of the efficacy of ecosystem-based management.

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Appendix 1

Discounted cash flow model