

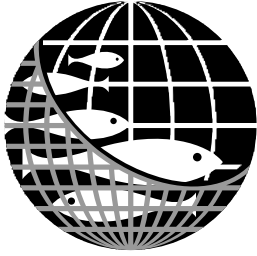
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of BC**

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Northern and Central Coast
of BC**

Fisheries Centre, University of British Columbia, Canada

A synthesis of research activities at the Fisheries Centre on ecosystem-based fisheries modelling and assessment with emphasis on the Northern and Central Coast of BC

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A SYNTHESIS OF RESEARCH ACTIVITIES AT THE FISHERIES CENTRE ON ECOSYSTEM-BASED FISHERIES MODELLING AND ASSESSMENT WITH EMPHASIS ON THE NORTHERN AND CENTRAL COAST OF BC

by

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David Preikshot, Daniel Pauly

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Director's foreword

This report reviews ecosystem modelling and related work performed in the last decade on marine ecosystems of British Columbia. While many of our activities are global in scope and entail field work on other continents, we are equally committed to addressing the marine challenges of our home province. In fact, as the numerous references in this report indicate, the Fisheries Centre has worked extensively on British Columbia freshwater and marine ecosystems, which is especially evident from the number of theses of our graduate students. This research, however, has been under-utilised by outside institutions.

We therefore embraced the opportunity when Jack Mathias, as Senior Policy Advisor, Department of Fisheries and Oceans of Canada (DFO) Oceans Planning, gave us a small grant to review the work of the Fisheries Centre on ecosystem-based modelling and management within BC. Upon his agreement to disseminate this study beyond DFO, we edited this *Fisheries Centre Research Report*. This report synthesises past work and also scopes out future work to stimulate the transition, in BC and elsewhere, of fisheries management to ecosystem perspective.

We are thrilled to share this report on BC marine ecosystems with other institutions who may benefit by connecting it to their own activities, e.g., setting up marine protected areas along parts of the BC coast. The bulk of the work cited herein and the report itself are available from our website (www.fisheries.ubc.ca). We encourage those interested to contact the authors of this report and other Fisheries Centre researchers to clarify any issues related to the marine and freshwater ecosystems of BC and their modelling.

It is in this collaborative spirit that I invite you to read this report, which highlights the areas that we know well, so that together we can fill in the gaps with effective solutions that track and maintain marine ecosystem health to keep British Columbia beautiful!

Executive summary

Worldwide, fisheries are perceived as facing serious threats, and the need to turn towards ecosystem-based management is widely recognised. The transition towards ecosystem-based management makes it increasingly important to incorporate strategic considerations into the management process. Ecosystem modelling that includes ecological and socio-economic considerations is a tool to address different questions than single species assessment by placing the entire catch of fisheries into an ecosystem context. The present report addresses this by first outlining the methods developed at the Fisheries Centre to reconstruct (approximate) time series of illegal, unreported or unregulated (IUU) catches. We then give an account of work on marine ecosystem modelling performed at the Fisheries Centre, using Ecopath with Ecosim (EwE). Through EwE, it has been possible to investigate the impact of fisheries, climate and nutrient loading on ecosystems. Fisheries management scenarios such as various gear restrictions and marine reserves have also been evaluated using EwE or with Ecospace, a spatially explicit extension of EwE.

Numerous ecosystem models have been created for the coast of BC and the northeast Pacific. The construction of each of these models has served as an opportunity to compile and synthesise a huge amount of disparate and dispersed data, taking into account oceanographic and geographic features of the coast to delineate ecosystems, and in the process making gaps in our knowledge much more explicit. These models have then been refined through several iterations and workshops that included researchers from Fisheries and Oceans Canada, the NOAA Alaska Fisheries Science Center, the University of Washington and the University of British Columbia. The model created for the Strait of Georgia has been modified, updated and built for different time periods and fitted to time series for the period 1950-2000. The Hecate Strait model has been refined with several versions and has been recently modified to cover Northern BC.

The Back To The Future technique has been developed to be able, based on models of past and present ecosystems, to identify states of the system at different points in time. The models are then used to simulate various scenarios of exploitation with the objective of identifying viable fisheries, aimed at creating a sustainable food and wealth source, from capture fisheries and aquatic ecosystems. This technique has been used with models of the Hecate Strait, Northern BC and the Strait of Georgia as well as with other ecosystems in the temperate regions of the world.

The policy search procedure of EwE has been used in several studies to evaluate the benefits and constraints of various policies. The outcomes have been evaluated using economic and biodiversity indices. Additional synthetic methods include the use of fuzzy logic to assess species vulnerability to exploitation.

Much work has been made on ecosystems valuation. Previous methods of economic assessment relied heavily on the cost to the user in present terms, usually net present value. While this proved effective for quantifying the cost to current generations, future generations are completely ignored. Our socio-economic models of resource use have included intergenerational discounting as a means to describe human behaviour in the face of exploitable natural resources. Intergenerational approaches to evaluating future benefits of a rebuilt ecosystem can be applied both to single-species fisheries problems, and to entire ecosystems, such as the EwE model of Northern BC.

1. Introduction

Worldwide, fisheries are perceived as facing serious threats, manifest in stagnating or declining global catches (FAO/FISHCODE 2001; Watson and Pauly 2001), the collapse or decline of iconic species such as northern cod (Walters and Maguire 1996), and the persistent overcapacity of fishing fleets (Mace 1997), widely associated with inappropriate incentive structures (Clark et al. 2003; Hilborn et al. 2004; Hilborn et al. 2005; Clark et al. in press). This is a major reason for the worldwide call for a transition toward ecosystem-based management of living resources, expressed by bodies and conventions such as UNCED, the Convention on Biological Diversity, the Kyoto Conference Plans of Action, the UN Code of Conduct for Responsible Fisheries, the Reykjavik Declaration on Sustainable Fisheries, and the Johannesburg Declaration.

As well, fisheries management is becoming increasingly complex as more stakeholders get involved, and as more objectives have to be considered as part of the management process. As part of the move toward ecosystem-based management, there is, however, a series of research-related issues that have to be explored, notably those related to techniques, indicators and information needs that should be considered as part of the management process.

The trend toward ecosystem-based management of fisheries (or EAF, Ecosystem Approaches to Fisheries, as it is termed by FAO) makes it increasingly important to incorporate strategic considerations into the management process (Christensen and Walters 2005). Fisheries management has been concentrated on tactical questions, which indeed is where the immediate management concerns are, while strategic considerations have been virtually absent from the management procedures. Incorporating multiple objectives into the management calls for ecosystem modelling to be included in the procedures. This topic will form the centrepiece of this report.

Ecosystem modelling as we define it (modelling which includes all trophic levels as well as economic and social factors that may influence the ecosystem) indeed has its major strength when viewed as part of a strategic management process. Ecosystem modelling is not designed to decide what fishing mortality we should be allowing for a given species in a given inlet in the summer of this year. It is rather to address questions of how we would prefer the ecosystem to function some years from now. It should be seen as a complement to the traditional management process, used to address different questions. Indeed, there needs to be a strong interaction between single-species models and ecosystem models: one where the single species models supply information about the major species to the ecosystem models, which in turn analyse these in the ecosystem context and provides reference values (e.g., natural mortality rates), and back to the single-species models as part of an iterative process.

The present report focuses on ecosystem studies that have been carried out at the Fisheries Centre and which can be used to inform and support the ecosystem management process (for obvious reason, we focus on studies performed in or which have relevance to BC). Fisheries Centre researchers and students have worked on various aspects of ecosystem modelling and food web description, generally employing Ecopath with Ecosim (EwE), a software for ecosystem modelling that is being developed at the Centre, mainly by Villy Christensen and Carl Walters (Walters et al. 1999; Christensen and Walters 2004b). Using EwE, several case studies have been developed to investigate the usefulness of marine reserves, and the effect of fishing and climate, and fishing management scenarios, on ecosystems. Ecological, social and economic indices available within EwE capabilities can be used to compare potential ecosystem mechanisms and search for policies to optimise all, or any subset, of those indices.

Researchers at the UBC Fisheries Centre have developed methodologies for estimating unreported catches. Local and traditional knowledge of aboriginal people and their possible contribution to ecosystem modelling have also been explored. Socio-economic models have also been developed and used to address the value of management scenarios using social indices and market and non-market values, sometimes using an intergenerational perspective.

This report first addresses techniques developed to estimate unreported catches and catch rates for fisheries, and continues on to ecosystem modelling, its characteristics and the types of problems EwE has been applied to. The Back To The Future (BTF) approach, also developed at the Fisheries Centre, is described in a separate section because of the series of techniques it developed for ecosystem reconstruction, policy searches and ecosystem valuations. Then, tools developed to assess the status of

ecosystems based on economic, social and ecological characteristics are described. We conclude with economic models that have been developed to be used for single species and adapted to ecosystem modelling.

2. Comprehensive catch as a pre-requisite to ecosystem-based management

2.1 Problem statement, with emphasis on BC

The transition to ecosystem-based fishery management requires comprehensive time series of total extractions, going as far back as possible. The importance of this cannot be overemphasized. Estimates of total extractions include the illegal, unreported (including discards) and unregulated catches (IUU), of which the last two can be understood as ‘unmandated catch’, i.e., catches which regulatory agencies are not tasked (‘mandated’) to report (Pitcher and Watson 2000; Pitcher et al. 2002d).

Ecosystem-based management of fisheries requires comprehensive catch estimates because all withdrawals affect ecosystems and the species embedded therein, be it directly (because catching a species reduces its biomass in the system), and indirectly (because catching, e.g., the common prey species of a marine mammal, will force it to turn to other prey and/or decline). Also, catching or not catching species is our main ‘handle’ for affecting an ecosystem, and the history of our ecosystem impacts is thus reflected in time series of withdrawals. This history must go as far back as possible because earlier periods, generally with lower withdrawals, provide a contrast through which the impact of present rates of extractions can be evaluated. This is the reason why we devote an entire section (below) to reconstructing catches.

2.2 Methods for estimating unreported catches

Precise data are difficult to obtain on IUU catches since few records are kept and most of this activity is (by its very definition) unreported. Since assuming that there are no IUU catches is very unrealistic, we have at the Fisheries Centre developed two methods to estimate the amount of IUU catches. These two related and partly overlapping methods emphasise different aspects of IUU time series, and are best presented separately. The first of these methods, which goes back to a paper by Pauly (1998), and emphasizes unmandated catches, was applied by D. Zeller and collaborators to various, data-sparse tropical countries and territories (Zeller et al. 2005; 2006; Zeller et al. 2007). It can be summarized by six steps:

1. Identification and sourcing of existing, reported catches (R_{ij}) for each reported year i by taxon j ;
2. Identification of sectors, time periods, species, gears, etc. not covered by (1) e.g., unreported ‘missing’ catch data via extensive literature searches and consultations with local experts;
3. Sourcing of available alternative information sources dealing with unreported ‘missing’ data identified in (2), via literature searches and consultations with experts;
4. Development of data ‘anchor’ points in time for unreported data items, and their expansion to country-wide catch estimates by taxon;
5. Interpolation for time periods between country-wide expanded data ‘anchor’ points, generally via *per capita* catch rates, deriving estimated unreported catch (U_{ij}) for each year i by taxon j ; and
6. Estimation of final total catch (C_i) for year i , combining reported catches (R_{ij}) in year i by taxon j from (1) and interpolated, country-expanded unreported catch estimates (U_{ij}) in year i by taxon j from (5), i.e., $C_i = \sum_j R_{ij} + U_{ij}$.

The other method, developed by T. Pitcher and collaborators (Pitcher and Watson 2000; Pitcher et al. 2002d), and emphasising illegal catches, is as follows:

1. Divide the time span for which IUU catches are to be estimated into decades (or more appropriate time periods);
2. Define factors influencing (on a qualitative scale) fishers' behaviours, i.e., the incentives and disincentives to misreport, corresponding to changes in regulatory regimes;
3. Assign quantitative values, best estimate and confidence intervals (called anchor points), to influence factors based on available data by gear;
4. Interpolate values for missing periods based on the incentives;
5. Provide an estimate of total IUU catch weighted by gear; and
6. Use Monte Carlo resampling to estimate the mean value and confidence interval of the total misreported extraction.

Pitcher et al. (2002d) estimated the total extractions for Iceland and Morocco using this methodology. Ainsworth (2006) refined this methodology by evaluating each of the categories of IUU separately under

the assumption that each category would be influenced differently by changes in regulatory regimes. The study, as articulated in this way, should be useful to managers who wish to identify the major sources of unreported catch and the most pressing actions to take. Ainsworth (2006) has used this methodology to examine the major fisheries of BC: salmon and groundfish, including all gear types and recreational fishery. The results from this study show that between 10 and 20 thousand tonnes of catch were unrecorded every year for BC salmon and groundfish fisheries between the 1950s and the 1980s. This estimate subsequently increased to 30 thousand tonnes per year (18% of the catch) in 1990. The methodology proposed in these studies has the advantage of being transparent with regard to assumptions and methods, making it straightforward to evaluate. Ainsworth (2006) suggests additional approaches to improve and refine the estimation of unreported catches for British Columbia.

2.3 Native fishery and First Nations footprint at European contact

In the Pacific Northwest bioregion of North America, complex pre-contact indigenous societies fostered ecosystem and socio-cultural resilience and sustainability (Trosper 2002; 2003; Lam and Gonzalez-Plaza 2006), with millennial local adaptations to, and modifications of their environments (Suttles 1987; Ames and Maschner 1999). Dynamic, apparently sustainable, yet intense interactions between the indigenous people, organisms, and landscape demonstrated respectful adaptive management (Berkes 1999). The indigenous inhabitants of the Salish Sea or Georgia Basin/Puget Sound bioregion extensively intervened and manipulated the regional landscape in environmental practices, e.g. directed fire use, cultivation, and fishing, without obvious signs of overexploitation (Boyd 1999). Sustainable cultivation of prairies of camas lilies, a staple regional food, and harvesting of salmon, an ecological and cultural keystone species (Garibaldi and Turner 2004) ensured not only their survival, but also enhanced the viability of the social contract between humans and their ecosystem 'partners'. Following contact with Europeans, however, indigenous communities suffered cultural loss, when reservations disrupted their local relationships with terrestrial and marine resources and traditional ways of life.

The significance of aboriginal pre-contact subsistence fishery in the Pacific Northwest has been analysed and aboriginal salmon consumption estimated, as a measure of the use of fishery resources in aboriginal pre-contact and transition to commercial fishing (Hewes 1973). For example, for the northwest coast native groups at time of contact, the total aboriginal salmon consumption was estimated at roughly 35 million pounds of fresh fish per year (16 million kg) for a population of about 75,000, which gives a per capita estimated consumption of 485 pounds/year (220 kg/year). Decline of the native fishery as a result of European contact has been proposed to have led to a temporary existence of a fish surplus and hence early high productivity of the commercial fisheries. Along the Fraser River, with an estimated aboriginal population of 60,000, the pre-contact annual catch has been estimated at "12 million sockeye equivalents" in all salmon species, comparable to the post-contact actual sockeye catch plus spawning escapements of between 3.3 and 11.2 million fish during 1894-1986 (Trosper 2002). With this historical consumption data and estimates of the range of aboriginal catches, an aboriginal 'footprint analysis' on the marine environment at the time of contact may be conducted. Historical and present aboriginal and industrial fisheries can then be compared, to quantify anecdotal claims of 'light' aboriginal footprints (Chief Blaney, Homalco First Nation, pers. comm. to M. Lam) and our suggestion of relatively light historical fishprints due to the more limited ranges of fishing technology and trade in the past.

In the Pacific Northwest, First Nations have practised adaptive resource management and subsistence harvest technologies for at least twelve thousand years (Haggan et al. 2004). They contributed to spreading salmon, increasing habitat complexity, stabilizing food supplies, and advancing their socio-cultural development. Such integrated aboriginal ecosystem management maintained productive marine resources at the time of European contact. How this millennial, fine-scale, integrated aboriginal ecosystem knowledge can be linked with current ecosystem-based science through computer modelling and marine resource valuation is being explored to encompass the economic, ecological, social, and cultural variables in predicting future ecosystem states and their human consequences (Haggan et al. 2005).

3. Ecosystem modelling

3.1 Ecopath with Ecosim

Ecosystem modelling has a clear contribution to make to resources management. Once we start involving multiple resources and multiple objectives into management it becomes necessary to manage trade-offs in an ecosystem setting (Christensen and Walters 2004a; Walters and Coleman 2004). The most widely applied tool for ecosystem modelling with a fisheries perspective is the Ecopath with Ecosim software

(EwE) that is being developed at the Fisheries Centre mainly by Villy Christensen and Carl Walters (Walters et al. 1999; Christensen and Walters 2004b). More than half of all primary publications in the field of ecosystem-based fisheries management apply EwE (Christensen and Walters 2005) and it is currently being reviewed by NOAA as an additional tool for fisheries management (Boldt et al. 2005).

Ecopath models are a description of food web interactions of functional groups (composed of one or a group of species), which constitute a 'snapshot' image of the ecosystem for the period considered (Christensen and Pauly 1992). The models account for the biomass (usually in $t \cdot km^{-2}$) of each functional group, their diet composition, consumption per unit of biomass, natural and fishing mortality, accumulation of biomass and net migration. The principle behind this modelling approach is that, on an annual basis, biomass and energy in an ecosystem are conserved (Walters et al. 1997).

ECOSIM: TIME-DYNAMIC SIMULATIONS

Ecosim is a tool for dynamic simulations based on the Ecopath model, allowing projections forward in time. Ecosim uses a system of differential equations to describe the changes in biomass and flow within the system over time, by accounting for change in predation, consumption rate, emigration and fishing (Walters et al. 1997; Christensen and Walters 2004b). Over time, the software was endowed with additional functionalities that allow explicit consideration of life history changes, population age-structure, predator-prey relationships, fishing and fleet dynamics, climate and nutrient loading anomalies, and trophic mediation, which we describe further in the following sections. We concentrate on these features because of their relevance for the work done in various temperate ecosystems notably in BC. Further details can be found in the voluminous literature published on this software (Pauly 1998; Walters 1998; Walters et al. 1998; Walters et al. 1999; Pauly et al. 2000; Christensen and Walters 2004a; b).

There are, of course, and will always be additional features necessary to address specific problems that will eventually be encountered. For example, it is difficult at the present to account for hatcheries production and wild fish of the same species in an Ecopath model (e.g., Cox and Kitchell 2004). However, a current project that aims at evaluating the impact of salmon hatcheries in the northeast Pacific may provide to one of us (D. Preikshot) the opportunity to work on this modelling difficulty. Thus, a feature for explicitly modelling the impact of hatchery production has recently been added to the EwE system.

STANZAS

Ecosim is also able to incorporate multiple stanzas representing life history stages for species with complex life histories. The stanzas are linked and their respective production per unit of biomass ($P/B \text{ year}^{-1}$), consumption per unit of biomass ($Q/B \text{ year}^{-1}$), and growth is calculated from a baseline estimate for a leading group (the adults in most cases). Growth for each stanza is inferred from the von Bertalanffy growth curve and assumes stable survivorship within age groups (Christensen and Walters 2004b).

PREDATOR-PREY RELATIONSHIP

The functional predator-prey relationship is based on the foraging arena theory, dividing the prey biomass into vulnerable and invulnerable pools (Walters and Kitchell 2001). The transfer rate between these two pools (also called 'vulnerability') can range from one to infinity, with higher rates implying that the behaviour of both the prey and the predator have weaker effects on limiting predation rates. A large vulnerability value also means that the initial biomass of the predator is low compared to its carrying capacity. There is thus little compensation due to density-dependence, and an increase in its biomass will cause a corresponding increase in mortality rate in its prey. In contrast, a low vulnerability means that an increase in predator abundance will not result in a large change in the mortality of its prey, the predator is close to its carrying capacity, and there is a strong density-dependence. This functional response equation predicts changes in diet composition due to changes in relative availability of prey and alternative prey, but it does not allow switching effects to new prey, i.e., prey that were not consumed initially. Model fitting to time series reference data (notably from assessment and surveys) is achieved by evaluating vulnerabilities that minimise the sum of squares of differences between the data and the model predictions. The model also allows the time spent foraging by a predator to vary to respond to diminishing prey availability.

TROPHIC MEDIATION

It is possible to account for species that may modify the trophic relationship between two other species. For example, tuna predation may drive small pelagics closer to the surface, thus facilitating predation by seabirds. Biogenic bottom structure, e.g., made up of corals and other macrobenthic organisms, can decrease the availability of prey to a certain type of predators by providing hiding places. The user can

determine the form of the relationship between the impacts of a third group as a function of its biomass. The function is then applied to modify the vulnerability of the prey to the predator.

ACCOUNTING FOR CLIMATE VARIATIONS

In addition to using EwE models to emulate changes in reference data of fishing and total mortality over time, it is also possible to have Ecosim simulate potential time series of climate anomalies such as the Pacific Decadal Oscillation. These climate anomalies are linked to primary production in EwE. By changing phytoplankton biomasses over the time period modelled, different hypotheses for bottom-up mechanisms can be examined. Changes in the biomass of primary producer groups in a model can be varied on an annual or interannual basis to simulate bottom-up changes in the energy available to the ecosystem. By varying the production of primary producers, and depending upon the vulnerability settings of herbivores and their predators, different biomass trajectories can be created for groups in the trophic web in response to changes in total available ecosystem energy. By incorporating this type of bottom-up mechanism it is possible to examine the regime concept of decadal scale periods of increased or decreased production in an ecosystem (e.g., Francis et al. 1998; Hare and Mantua 2000).

NUTRIENT LOADING

The nutrients in the system are divided into two pools: free and fixed (in functional groups). By default, the proportion of free nutrients is assumed to be very high and the inflow and outflow of nutrients are assumed to be at equilibrium. Nutrients can be forced to vary through time by adding a time series in Ecosim input data. The primary production is then linked to the free nutrient pool at each iteration (details in Christensen et al. 2005). This feature has not been used much and is not always warranted in fisheries management scenarios (Walters and Martell 2004, p. 271-275). However, in coastal areas where nutrient loading has varied considerably, accounting for nutrients may be crucial to explain functional group variations. Recent case studies, e.g., the Chesapeake Bay (Christensen et al. unpublished data) and some Florida bays (Walters et al. unpublished data), have warranted the addition of a module to calculate the variation in nutrients and their circulation in the bay based on rainfall, runoffs, wind, and loading of nutrients. The Ecospace models are still under construction, but they have already shown an ability to explain the observed changes in plankton and fish through time. The most important finding from the studies so far is that there are strong linkages between productivity changes (be they caused by environmental changes or nutrient runoff patterns) and fish production, and that both such productivity changes and fishing patterns need to be considered explicitly when attempting to explain what has happened in a given ecosystem over time (Christensen and Walters 2005).

FISHING AND FLEET DYNAMICS

Fisheries dynamics are included in the model through fishing mortality or effort time series applied directly as input to drive the model and generate estimates of catches and biomasses over time. Using effort time series for each fleet allows the estimation of catches of target species and the associated bycatches to be compared to time series reference data. In addition, temporal changes in fleet size and fishing effort can be modelled to explore various management scenarios. It is also possible to explore the impact of unregulated economic response dynamics by modelling fleet/effort dynamics. Then effort becomes the result of fishers' investment and operating decisions according to bioeconomic dynamics, assuming that fishers are behaving like predators. Most modelling simulations that we are aware of have used the first option.

Ecosim has been widely used for forward projections to explore management scenarios of disturbances and fishing policies (e.g., Mackinson et al. 1997; Trites et al. 1999a; Mackinson et al. 2003; Christensen and Maclean 2004), the keystone role of apex predators (Kitchell et al. 1999; Libralato et al. in press), the effect of model structure and complexity on the simulated responses to disturbances (e.g., Vasconcellos et al. 1997; Fulton et al. 2004; Pinnegar et al. 2005) and the impact of alternative trophic functional relationships on model behaviour (Mackinson et al. 2003). Also, the comparison of ecosystem models of areas submitted to different management measures can allow identification of key mechanisms explaining the differences in observed biomass. For example, the relationship between trawling, seagrass and the biomass of fish was explored by comparison of models of trawled and untrawled seagrass habitat in a study of the Mediterranean Sea (Rodriguez-Ruiz 2001 in Walters and Martell 2004).

In more recent applications of Ecosim, the simulations in time are typically driven by fishing effort and the results are compared to available time series reference data of for example, abundance, catches, and mortality. Model fitting is achieved by evaluating vulnerabilities that will minimise the sum of squares of

differences between the data and the model predictions. Experience shows that aiming at replicating observed behaviour through time often results in appreciably modified parameters and influences the dynamics behind the simulations. In spite of the large number of parameters involved, accounting for what is known about exploited species and putting it in an ecosystem context quantitatively and explicitly adds constraints to the model (Christensen and Walters 2005).

For the purpose of screening management policies, using Ecosim without the benefit of comparing with data time series is problematic. Mainly, without time series data, it is difficult to differentiate between high growth rate/low initial biomass and low growth rate/high initial biomass scenarios. However, as in any modelling exercise, the fact that the model is well-fitted to the observed time series does not guarantee correct predictions about future policy impacts (Essington 2004; Walters and Martell 2004).

POLICY SEARCH

Ecosim can be used to explore the ecosystem effects of management policies by simulating various scenarios of fishing effort directly by changing the time series and comparing results with previous simulation or the baseline situation. An additional tool included in the software allows a search for the mix of fishing rates that would maximize objective functions composed of

1. fisheries rent;
2. social benefits (employment);
3. rebuilding of species above a threshold biomass; and
4. 'health' of the ecosystem structure (as defined by the user).

The policy search module was developed by Carl Walters and colleagues (Walters et al. 2002) for a FAO/UBC workshop held at the Fisheries Centre in July 2000, which involved application to 18 different ecosystem models (Pitcher and Cochrane 2002). The module has developed further since then and has been tested recently (Christensen and Walters 2004a). Several studies of the Back To The Future Project have included policy searches at different levels of development and are discussed in the BTF section.

ECOSPACE: INCLUDING THE SPATIAL STRUCTURE

Ecospace was developed to account for spatially structured behaviour and processes in addition to the key elements of time dynamics of the Ecosim approach (Walters et al. 1999). Ecospace allocates the biomass across a grid map covering the area modelled, and accounts for:

1. symmetrical movements from a cell to its four adjacent cells unless a preferred habitat has been defined;
2. user-defined predation risks and feeding rate in preferred habitats; and
3. a level of fishing effort proportional, in each cell, to the overall profitability of fishing in that cell and sensitive to costs (e.g., distance from port).

Ecospace now allows for seasonal migrations (although still limited) by letting the user define the path of migration and the target location of the centre of gravity of the population at each month. Cells can be further characterised by being assigned to a protected area or other management measures, such as closed seasons. Simulation studies show that trophic interactions are not necessarily modelled better in Ecospace (Christensen et al. 2005). For example, a predator that does not overlap much in habitat with a group of prey species is already taken into account in Ecopath (and Ecosim) by the low proportion these species contribute to the predator's diet. As advection processes are important in determining sites of high productivity and of accumulation, the spatial variation in production is included by using data of current velocity, provided by the user based on oceanographic models, and Ecospace accounts for local shoreline, bottom topography and the Coriolis force in calculating equilibrium upwelling/downwelling velocities (Christensen et al. 2005).

Ecospace is still limited in the representation of trophic interactions and details of species population dynamics (Martell et al. 2005). The main problem in the development of spatial simulations is the lack of spatially-explicit data on species abundances that would help to test the model predictions (Walters and Martell 2004, p.282). Also, it is difficult to model real migrations as we rarely know the incentives for the fish to disperse the way they do. The evaluation of fisheries management policies and their potential effect on the ecosystem trophic and spatial structure makes it dangerous to fix migrations to historical average migration trajectories. Finally, the impact of seasonal migrations on predation and fishing mortalities can be quite important in some cases (e.g., localised bottleneck caused by fisheries during salmon migrations) and is not well accounted for in Ecospace (Walters and Martell 2004).

3.2 Ecosystem delineation

In order for a model to serve a useful purpose, it must allow the researcher to explore different mechanisms, which may affect the system being modelled. This means that the system being modelled must have its dynamics primarily depend on internal processes. If most of the change is generated outside the modelled system, any model of it will fail to capture known changes or provide a robust platform to explore hypotheses of why known changes occur. In the case of managed aquatic ecosystems, this type of exploration involves simulating how various natural and anthropogenic mechanisms could influence the dynamics of the populations of organisms in which humans have an interest. Very often, if these species supply a commercial or recreational service to humans, there may even be some idea of how the managed populations have changed, how productive they are, and how they interact with their predators, competitors and prey. Therefore, to better interpret how fisheries, climate and species interactions may change the populations of marine organisms, the ecosystem (and model) should have more internal exchanges of energy than what is either imported or exported (Christensen and Walters 2004b). We note, incidentally, that this is part of an ecosystem's definition (Pauly and Christensen 2002).

Thus, this logic in many ways parallels work that has been done to delimit the existence of natural boundaries in the world's oceans that divides them into ecosystems. As will be discussed, the boundaries that separate marine ecosystems can be quite dynamic in time and space, depending on the dimensions used to define them. For example, one obvious boundary in the ocean is provided by strong temperature gradients. Most marine organisms prefer a rather limited range of temperature. Temperature boundaries occur at many obvious and well-recognised features of the seas, e.g., the thermocline, current fronts, and eddies. One well-established characteristic of such physical processes, though, is their tendency to change on time scales ranging from days to decades. One excellent example of this is the changes in latitudinal position of the divergence of the California and Alaskan currents off the west coast of North America, generally off the west coast of Vancouver Island. The divergence tends to be further north in the summer, but the absolute latitude to which it moves in either summer or winter and the duration and intensity of the current can also change on annual and decadal scales. Therefore, the northward extension of the California current can impart very different physical effects upon the BC coast depending on its timing, duration, latitudinal position and intensity.

It is within this physical oceanographic context that much of the work dividing the oceans into ecosystems has been conducted. In the northeast Pacific, Ware and McFarlane (1989) used oceanographic features and distribution of fish and fisheries to help delimit three production domains: central subarctic, coastal upwelling and coastal downwelling. Similar work by Longhurst (1995), using oceanography and primary production characteristics divides the world's oceans into about 50 biogeochemical provinces, three of which correspond to the northeast Pacific 'production domains' of Ware and McFarlane (1989). Such work suggests that these three large regions have sufficient internal exchanges and coherence to qualify as ecosystems and to be meaningfully adapted to an ecosystem model.

However, there are smaller regions within these larger zones which, though not described in the above works as 'domains' or 'provinces', have distinct characteristics that separate them from surrounding ecosystems. Two obvious examples are the Strait of Georgia and Puget Sound. Though influenced by currents, these areas qualify as ecosystems because the geographic constraints of the land masses that contain them limit the flow of biota across their aquatic boundaries. Note, however, that the large migrations of salmonids, sea lions, and marine birds through such small ecosystems make any model's ability to resolve population dynamics of the migrating species very problematic. Thus in EwE models of Puget Sound (Preikshot and Beattie 2001) and the Strait of Georgia (Martell et al. 2001), populations of resident salmonids (coho and chinook salmon) were addressed explicitly, while salmonids moving through these ecosystems (chum, pink, and sockeye salmon) were dealt with in a more perfunctory manner.

As an example of the importance of defining the areal scale of a model, Preikshot (2005) showed that for groundfish such as walleye pollock and arrowtooth flounder, biomass dynamics were better explained in the context of a larger model of the northeast Pacific, including the Bering Sea, Gulf of Alaska, and BC coast, than was possible in a model of the BC coast alone. Work that will be available later this year (Preikshot in prep.) shows that the Strait of Georgia and BC coast Ecosim models were better able to explain the biomass dynamics of coho and chinook (which tend to be coastal) than the northeast Pacific model. In the cases of pink, sockeye, and chum salmon (which move throughout the northeast Pacific and into the Alaska Gyre), on the other hand, the northeast Pacific model was better able to capture the

changes in catch than the BC coast model. In the end, researchers can build whatever model they want. If, however, they wish to maximise the value of the work invested in the model, it is crucial that the behaviour of the species modelled and the physical boundaries within which they move are compatible.

3.3 Available models of the northeast Pacific

Several ecosystem models have been created for the coast of BC and the northeast Pacific (Table 1) and are available. The initial mass-balanced models created for the Strait of Georgia (e.g., Pauly and Christensen 1996), have been modified, updated and built for different time periods and fitted to time series for the period 1950–2000. The Hecate Strait model has been refined with several versions and has been enlarged recently to become the Northern BC model (Ainsworth 2006). There is also a model of the western coast of Vancouver Island that has been used to place the shrimp fishery into an ecosystem context (Martell 2002).

Several workshops refined models and explored Ecosim possibilities and limitations. In September 2003 and May 2004, for example, workshops in Nanaimo and Seattle aimed at evaluating the relative roles of fisheries and environmental factors for North Pacific ecosystems from the Bering Sea to California. The workshop brought together researchers from the Pacific Biological Station of the Department of Fisheries and Oceans, the NOAA Alaska Fisheries Science Center, the University of Washington and the University of British Columbia. Some of the models presented and used at these workshops have been published in a Fisheries Centre Research Report (Guénette and Christensen 2005).

3.4 Questions that have been addressed

STATUS OF THE ECOSYSTEM

A large number of mass-balanced models have been published over the last 20 years (Christensen and Pauly 1993, Table 1; Christensen and Maclean 2004). They have been used to synthesise information on various components of the ecosystems and their relationships and to calculate various ecosystem indices familiar in network analysis, such as the connectance index, resilience and maturity (Christensen 1995). Lately the network analysis indices have also been incorporated into Ecosim and have been used to look at the emergent properties of the ecosystems over time. Specifically, in the Gulf of Alaska, the network analysis indices showed that the decline in Steller sea lions, sharks, arrowtooth flounder, Atka mackerel and pollock in the Aleutians had a noticeable effect on the recycling and general resilience of the system, while the increase in sea lions, herring, Pacific cod and halibut in southeast Alaska did not have the expected effect on the recycling, flow dynamics and resilience of the system (Heymans et al. 2007).

Ecopath models have also been used extensively to explore the impact of various hypotheses about ecosystem properties (Aydin et al. 2002; Aydin et al. 2003). Several Back To The Future studies (see next section and Table 1) have compared ecosystems at different periods. Christensen et al. (2003b) have illustrated the dramatic decline in large predatory fish in the North Atlantic as a result of overfishing since 1950 and even 1880. They used a multiple regression combining 23 ecosystem (Ecopath) models of different periods with spatialised data on fishing effort and environmental variables, an approach later reproduced in Southeast Asia (Christensen et al. 2003a) and in West Africa (Christensen et al. 2004).

EFFECTS OF FISHING AND CLIMATE

A number of studies have evaluated the effects of fishing using EwE models and time series data aiming to explain the recent history of these systems. In most cases, it was possible to attribute the ecosystem changes over time to fisheries with or without environmental changes. For example, the Gulf of Thailand model, depicting a tropical system, showed that fisheries alone were responsible for the decline of exploited species (Christensen 1998; FAO/FISHCODE 2001). However, we found that in subtropical and temperate regions, although fishing explains a large part of the trends of exploited species, it is as a rule necessary to invoke climate factors as well in order to explain the changes that have occurred in ecosystems (Trites et al. 1999b; Christensen and Walters 2005).

One early example was the model of the French Frigate Shoals in which lobster and monk seals biomass dynamics were better explained when a decline in primary production was combined with known fishing and mortality data (Polovina 2002). Such a decline in primary production was indeed supported by parallel oceanographic research in the area. Similar primary production forcing was used in the Strait of Georgia by Martell et al. (2002) to improve predicted time series of biomass for a range of species, including seals, herring, salmon, and hake. In larger-scale EwE models of the BC coast and the Northeast Pacific, Preikshot (2005) demonstrated that the primary production anomalies producing the best fit of

Table 1. Models of British Columbia and the Northeast Pacific, the years they represent, their type, and their use. An asterisk indicates that the model was constructed within the Back To The Future project. The 'fitted' column indicates whether the model EwE has been fitted to times series.

Model area	Period	Type of model	Fitted	Use and remarks
Models of British Columbia				
Southern BC Shelf (Pauly and Christensen 1996)	late 1980s	Ecopath	No	First attempts at building Ecopath models of this area.
Strait of Georgia (Pauly and Christensen 1996)	late 1980s	Ecopath	No	First attempts at building Ecopath models of this area.
Strait of Georgia (Pauly et al. 1998a)	1990s, 1890s, 1490s	EwE *	No	Build Ecopath models of various periods and compare them on the basis of selected attributes from the results of Ecosim simulations of hypothesis about the 1490s model.
Strait of Georgia (Beamish et al. 2001)	1999 - 2001	Ecopath	No	Comparison of the two years to try to understand the abrupt change in productivity in 2000.
Strait of Georgia (Martell et al. 2001; Martell et al. 2002)	1950	EwE	Yes	Model with life history details included for some species (stanzas). Fitting using fishing fleet effort and climate anomalies. Uses optimal search procedure in Martell et al. (2002). Shows the importance of climate anomalies to fit the model.
Burnaby Narrows (portion of Gwaii Hanaas, Hecate Strait (Salomon et al. 2002))	1990s	Ecospace	No	Impact of size, location and intensity of fishing restriction, within a 277 km ² MPA, on the biomass of various trophic groups.
Hecate Strait (Beattie 1999)	early 1990s	Ecopath *	No	Basic Ecopath model (area 5C, 5D).
Hecate Strait (Haggan and Beattie 1999)	1890s	Ecopath *	No	Basic Ecopath model (area 5C, 5D).
Hecate Strait (Beattie 2001)	early 1990s	Ecospace	No	Improves and expands Ecopath model built in 1999 (Beattie 1999). Develops the Ecosseed procedure to determine the optimal size and placement of reserves based on ecological and economic indices.
Hecate Strait (A. Sinclair, DFO, Nanaimo, unpublished)	1990s	EwE	Yes	Detailed model and times series for the 1990s.
Northern BC (Ainsworth et al. 2002a)	2000, 1950, 1900, 1750	Ecopath *	No	Comparison of the time period ecosystems.
Northern BC (Ainsworth 2006)	1950 - 2000	EwE *	Yes	Improved version of the previous model (Ainsworth et al. 2002a). Includes climate, and a forcing function for herring egg production.
Northern BC (Ainsworth 2004 in Ainsworth 2006)	2000	Ecospace	No	Area similar to PNCIMA. Evaluate different scenarios of fishery closures for the Gwaii Hanaas NMCA.
West Coast of Vancouver Island (Martell 2002)	1950-2000	EwE	Yes	Basic model focusing on the main trophic interactions. Examine the shrimp population dynamics in an ecosystem context.
Coastal BC (Preikshot 2005)	1950-2002	EwE	Yes	These two nested models were built to test the impact of scale in dealing with climate forcing. Simulations results also show the relative importance of climate and fishing to explain trends in fish species. A third Strait of Georgia model is in preparation to continue the modelling of climate and fish at different areal scales.
Northeast Pacific (Bering Sea to BC coast) (Preikshot 2005)	1950-2002	EwE	Yes	These two nested models were built to test the impact of scale in dealing with climate forcing. Simulations results also show the relative importance of climate and fishing to explain trends in fish species. A third Strait of Georgia model is in preparation to continue the modelling of climate and fish at different areal scales.

Model area	Period	Type of model	Fitted	Use and remarks
Bowie Seamount (Beamish and Neville in prep. in Canessa et al. 2003)	unknown	Ecopath	No	First attempt at describing the seamount's food web.
Models of the northeast Pacific				
Prince William Sound, Alaska (Okey and Pauly 1999)	1994-1996	Ecopath model for 1994-1996	No	Ecosim and Ecospace simulations to explore various fisheries management and catastrophic disturbances scenarios, and the usefulness of marine reserves.
Southeast Alaska (Guénette 2005) and Western and Central Aleutian Islands (Heymans 2005a)	1963-2002	EwE	Yes	Models used to investigate the causes of the decline of Steller sea lions in western Alaska, considering climate, fisheries, and trophic relationships simultaneously (Guénette et al. 2006). Effects of fishing and environmental variations are presented in Heymans et al. (2005).
Eastern Aleutians and Central Gulf of Alaska (Heymans 2005b)	1963-2002	Ecopath	No	Preliminary model.
Northern California Current (Field 2004)	1960s-1990s	EwE	Yes	Explore the role of fisheries and climate, using time series.
Puget Sound (Preikshot and Beattie 2001)	1970-1999	EwE	Yes	Elucidate the factors influencing coho early marine survival rates and determine crucial trophic relationships among the organisms of the study area.
Eastern Bering Sea (Trites et al. 1999a)	1950s-1980s	EwE	No	Compare both periods and explore the role of primary production, fisheries and mammals.
Central Pacific (Kitchell et al. 1999)	circa 1988	EwE	No	Evaluate the role of apex predators as keystone species by simulating a sudden increase of mortality for one large predator at a time.
Central Pacific (0-40°N and 150°W-130°E) (Kitchell et al. 2002)	1988		No	Modified from Kitchell et al. (1999). Examine the impact of longline fishing on sharks, their competitors and their food base. Also examine the impact of various level of cannibalism for sharks.
Central Pacific (Cox et al. 2002)	1950-1998	EwE	Yes	Test Ecosim model against time series data and the effects of fisheries on the food web structure and on tuna recruitment.
Central Pacific (Martell et al. 2005)	1950-1998	Ecospace	Yes	Test usefulness of marine reserves as a management tool assuming a continuous increase in fishing effort and under various scenarios of oceanographic processes and fish dispersal.
Central Pacific (Kitchell et al. 2004)	1952-1998	EwE	Yes	Used the model by Cox et al. (2002). Explore the ecosystem effects of marlin bycatch and the possible effect of conservation policies. Scenarios were evaluated on the basis of ecological and economic indices.
Alaska Gyre (Pauly and Christensen 1996)	late 1980s	Ecopath	No	First attempts at building Ecopath models of this area.
Eastern and Western Basins of the subarctic Pacific (Aydin et al. 2003)	late 1980s-early 1990s	EwE	No	Used to synthesise information, compare both ecosystems, and with results of NEMURO model for the lower levels of the food web. Evaluated the response to major perturbations of climate and food web. Simulation results compared using salmon abundance time series.

modelled to reference data were related to climate indices measured over similar areal scales. The BC coast model, about 100,000 km², suggested a time series of phytoplankton biomass from 1950 to 2000, which were similar to Bakun upwelling indices (Bakun 1973; Schwing et al. 1996) from 51°N and 54°N, which are measured over tens of thousands of km². At a larger scale, the Northeast Pacific model, about 1.5 million km², suggested that changes in phytoplankton biomass were more similar to the Pacific Decadal Oscillation, which is measured over millions of km². The same has been demonstrated for the west coast of Vancouver Island (Martell 2002), the Gulf of Alaska (Heymans et al. 2005), the Baltic Sea (Harvey et al. 2003), the Black Sea (Daskalov 2002), and the English Channel (Stanford and Pitcher 2004).

Gu nette et al. (2006) have compared two ecosystems of the Gulf of Alaska fitted with their respective time series (Table 1). The goal was to understand the dynamics of Steller sea lions by comparing an ecosystem in which Steller sea lions have declined since the late 1970s (the Aleutians Islands) to an ecosystem in which Steller sea lion populations have increased during the same time period (Southeast Alaska). Fishing, climate, predation by killer whales and competition with flatfish were considered simultaneously and are likely to have contributed to the Steller sea lion population dynamics. Changes in productivity, conveyed by the Pacific Decadal Oscillation, had inverse effects in each ecosystem, causing downward trajectories in several fish of the Aleutians in contrast to the upward trend observed in Southeast Alaska.

Cox and Kitchell (2004) used EwE to explore the possible trophic interactions that would explain why lake herring (*Coregonus artedii*) still do not show signs of recovery from overexploitation in Lake Superior. This ecosystem is interesting because of the long series of disturbances, invasion of lamprey (*Petromyzon marinus*) and rainbow smelt (*Osmerus mordax*), overexploitation, salmon introduction, and stock enhancement. The authors found that assuming a strong predation of rainbow smelt on lake herring juveniles provides the best fit to historical data. This study underlined the importance of a triangle between lake trout, rainbow smelt and lake herring in which the recovering population of lake trout may have facilitated a few good years of recruitment for lake herring by suppressing predation by rainbow smelt. Results from this preliminary modelling study, as the authors informed the readers, provided insights for management issues and pointed out possible problems with the lake trout fishery.

FISHERIES MANAGEMENT SCENARIOS AND POLICY SEARCH

Several studies have looked at management scenarios to assess the impact of fisheries policy on the ecosystem. For example, Kitchell et al. (2004) explored management policies for tuna fisheries based on ecosystem effects and economic gains if the conservation policies were implemented in 1998 for the next 30 years. The authors suggest that better management policies exploration would be achieved by taking into account the spatial distribution of fish and fisheries, which will be key for this specific question.

Christensen and Walters (2004a) have examined the behaviour of the policy search and explored the trade-offs involved using the Gulf of Thailand model. The objectives were to maximize the economic profit, the landed value of the catch and the ecosystem structure. The ecosystem structure was based on two indices: 1. the longevity-weighted biomass summed over ecosystem groupings (except invertebrates) as a measure of ecosystem maturity obtained from the biomass/production ratio for each vertebrate group in the system; and 2. a biodiversity index based on Kempton's Q75 index (Kempton 2002) that gives an idea of the number of species that compose the biomass of the ecosystem. For convenience, this index was expressed relatively to the results of the Ecosim baseline scenario. The results were consistent through various weighting methods and suggested a basis for true compromises between ecological considerations and profit, but not with landed value (Christensen and Walters 2004a). See also the work done in the context of the Back To The Future projects in both fisheries policy exploration and policy search.

USEFULNESS OF MARINE RESERVES

Three Ecospace simulations have been built for a portion of the coast of BC to assess the usefulness of marine reserves. Beattie (2001) used a model of the shelf and slope waters of Hecate Strait and the Queen Charlotte Sound, encompassing a total area of more than 70,000 km², to determine the optimum size and configuration of MPAs in an ecosystem context. This was based on a module (Ecosed) which uses a recursive algorithm to progressively increase the size of the closed area by testing and adding adjacent cells at each step until the increase in benefit has reached its maximum. The procedure includes managers' choices of cells to close and the subsequent response of the fishing fleets, which redistribute their effort to maximise economic gains. Simulation results were compared based on bio-economic indicators, including market and non-market values. The same approach was later applied to the overexploited North Sea and it was found that closing off 40% of its area to fishing would optimise economic rent (Beattie et al. 2002).

Salomon et al. (2002) modelled a square area (40 km by 40 km) at the Burnaby Narrows, in the Gwaii Hanaas proposed NMCA (National Marine Conservation Area, Parks Canada). Simulations show that the reserve would cause the prey to decrease within the reserve as the predators' biomass increased (trophic cascade) and that the small size of the reserve, compared to the home range of the species modelled, would result in density gradients within the reserve, as illustrated in Walters et al. (1999); but see Martell et al. (2005) for different conclusions using a model of the central Pacific. Salomon et al. (2002) explored the impact of the intensity of fishing restrictions and various zoning options within the MPA. Also, based on a gravity model included in Ecospace (Walters et al. 1998), the fishing effort was concentrated around the edges of the MPA, which would hinder the recovery of low-dispersing top predators. They discussed the possible short-term effect on biodiversity and the importance of designating MPAs based on ecosystem criteria.

Ainsworth (2006) modelled the Northern Coast of BC, an area similar to that of the Pacific North Coast Integrated Management Area (PNCIMA), which includes the proposed Gwaii Hanaas NMCA. The Ecospace model included current circulation, tidal speed, primary production per cell and nine habitat types. The author intended to assess the possible effects of closing the NMCA to fishing under several scenarios: 1. five seasonal closures of increasing length (0-12 months) with a maximal effort allowed during open seasons; 2. trawl ban; and 3. no commercial fishery. Although the modelling exercise was worthwhile, the author identified several improvements that are necessary to pursue these efforts.

Martell et al. (2005) used Ecospace to evaluate the usefulness of large marine reserves as a management tool for apex predators (tuna, sharks, billfish) and explore the sensitivity of the policy decisions to alternate hypotheses of fish dispersal and oceanographic processes (static or monthly dynamic). All scenarios assumed that fishing effort would increase in the next 50 years. They show that, given the sometimes drastic changes in fish distribution due to the high variability in oceanographic processes, it was difficult to draw robust conclusions about the efficacy of reserves. On the other hand, assumptions about dispersal rates and movements did not have as much influence as expected. The study brought insights on the ecological studies necessary to evaluate the potential of marine reserves. For example, the authors suggest that instead of focussing on migration behaviour in relation to productivity gradients, the closed areas should be treated as an adaptive management experiment to learn about responses in behaviour and the efficacy of reserves.

Zeller and Reinert (2004) have explored the usefulness of adding reserves in the Faroe Islands in the Northeast Atlantic to supplement the present management regime. The Ecospace model they built included an advection field to account for the retention of plankton production around the islands. Under simulation conditions, offshore closures were found to be beneficial to some species. However, the present regime, which includes catch and effort quotas, spawning season closures and a trawl ban within 12 nm of the islands, seemed quite effective for demersals.

Pitcher et al. (2002a; 2002b) used Ecospace to evaluate the usefulness of artificial reefs combined with marine protected areas and fisheries exclusion zones in Hong Kong. The model was not fitted to time series data and did not include climate forcing, but it accounts for non-feeding association between corals and other living bottom structures and the fish that are attracted to them. They compared various scenarios based on the catch level, biomass of targeted fish and species as an ecological indicator, and economic benefits.

4. Rebuilding value and Back To The Future approach

The Back To The Future approach (BTF) is used to construct models of past and present ecosystems to identify states of the system at different points in time. The models are then used to simulate various scenarios of fisheries exploitation with the objective of identifying viable fisheries, aimed at creating a truly sustainable food and wealth source from capture fisheries and aquatic ecosystems (Pitcher 1999; Pitcher 2005). Undepleted past ecosystems are often referred to as a 'Lost Valley' (LV) state, and these previous states of the ecosystem can be used as a goal for future ecosystem status (Pitcher 2004), as they support a larger biomass, and have the capability of providing greater services to humans if managed properly. Hence the BTF approach sets the goal to re-create ecosystems as they existed at some time in the past. These past ecosystems or LVs are obtained by building models of a past time, and attempting to alter present day fisheries in order to allow species abundances to return to or toward those of the LV states (Pitcher 2001). Pitcher (1998) outlines the basic BTF procedure in seven stages:

1. Ecopath model construction of present and alternative ecosystems. This gives a snapshot in time of the ecosystem concerning the species' present and fishing levels for that particular time. Several Ecopath models can be built for a specific location to identify the state of the system at different periods in time. The alternative ecosystems refer to past models of the same area or Lost Valley, as this may be used as a restoration goal;
2. Ecosim and Ecospace exploration of the limits to fishing (sector by sector) for each alternate model. Ecosim utilizes the snapshot in time (a mass-balance model) and projects it forward in time to see how systems will change under different fishing pressures to explore how this will impact the systems; Ecospace can add a spatial component to this analysis, allowing the user to view the changes that will occur spatially within the ecosystem;
3. Evaluation of economic and social benefits for each system. All scenarios can be evaluated and compared to each other based on job diversity, net present value of the system, or other benefits;
4. Choice of policy goal as the ecosystem that maximizes benefits to society. The policy search function in EwE allows the user to determine the priorities of ecosystem function, whether social, economic, ecological, or a combination of all benefits;
5. Design of instruments to achieve this policy goal. New routines to examine the costs of achieving restoration goals in relation to benefits have been used for the Northern BC model;
6. Evaluation of costs of these management measures. This mostly applies to economic valuation and if restoration goals are feasible; and
7. Adaptive implementation and monitoring of management measures. This task has yet to be attempted or completed for the BTF process.

At the Fisheries Centre, these methods have been applied to a variety of locations and ecosystems. Most relevant in the context of this report is the work done in the Hecate Strait and in the the Strait of Georgia, and the more recent Northern BC ecosystem model, which expands the Hecate Strait model geographically. The Northern BC model is a comprehensive model, which coupled with Ecosim simulations has applied the most recent techniques for assessing how to return this ecosystem to a rebuilt status without sacrificing social or economic benefits, insofar as this is possible.

4.1 Local and traditional environmental knowledge

Past ecosystems are commonly assembled using scientific archival data (published and unpublished reports), archaeological data and historical information (books, letters, trade accounts, and historical documents), and Local/Traditional Environmental Knowledge (oral sources, interviews, and discussions). Haggan et al. (1998a) outline the ideas of Traditional and Local Environmental Knowledge (TEK and LEK) and define them as myths and stories illustrating the relationship between people and the rest of creation, information from First Nations Elders, commercial fishers, sport fishers, human artifacts and fish remains in the archaeological records, archival sources and popular literature, and information on past, present and future trends in climate. They then go on to incorporate interviews and questionnaires into data suitable for modelling, by organizing resources by presence/ absence and use by different aboriginal groups (Haggan et al. 1998b). Results show which species were used in different areas and help to determine abundance (or absence) of species regionally for historical time periods. These results can then be used directly to determine which species should be included in past models, and their relative abundances. Other methods used to incorporate LEK and TEK into BTF include estimating past catches of First Nations in Labrador by reconstructing aboriginal diet from archeological remains, and estimating the amount of food needed per person per day to get an annual estimate of each food type consumed for pre-contact times (Heymans 2003). This was then used to reconstruct an estimate of fish species present, and in relative abundances in the area, again to ultimately be used as part of an Ecopath model for parameter estimates, and more generally to identify which species to include in each model.

More recently, members from First Nations and other coastal communities have been included in estimating biomass of local species, and general trends in species abundance. Local information is mostly gained through interviews or workshops. For the Strait of Georgia and Hecate Strait areas, various sources of LEK were incorporated into the models (Salas et al. 1998; Wallace 1998; Beattie et al. 1999; Jones 1999). Interviews for the Northern BC model consisted of 48 community members from the Prince Rupert region and Haida Gwaii. Flashcards with pictures of selected species were shown to interviewees, and comments

were recorded. Community interview processes for the Northern BC model are presented with results from interviews in Erfan (in press). Historical knowledge used for the Northern BC model is available at the BTF website¹, and is organized by species type or time period, which is readily accessible to the general public.

Ainsworth (2006) created a system for prioritising comments taken in the community interviews. He ranks interviews and comments by a pedigree system with more experienced fishers having higher importance than non-experienced fishers, and with each fisher more reliable in relation to the species they target, as opposed to their non-target species. Correlation of trends for each species between LEK (taken as interviews) and fisheries stock assessments (DFO 2004) was poor (37%). Fishers having more than forty years experience provided a significantly higher fraction of comments agreeing with the stock assessments (Ainsworth 2006). Ainsworth suggests more effort could be directed to this area of research as only a small fraction of the BTF historical data and interview database was used in this study. Data collected as LEK was applied to the BC model for data-poor groups, when scientific information was lacking, although this cannot be compared to catch statistics to give a greater confidence (Ainsworth 2006). Methods incorporating LEK into ecosystem models might not be as sensitive to shorter-term changes in the ecosystem, but they do prove crucial when attempting to show general trends or abundances, especially for past ecosystem models.

4.2 Models developed

Several Ecopath models have been constructed for the east and west coasts of Canada over the last 10 years at the UBC Fisheries Centre. Several models for the Strait of Georgia and Hecate Strait regions were constructed explicitly for use with the BTF method. The Strait of Georgia model was created to gain information on the present ecosystem as well as on what the ecosystem looked like 100 and 500 years ago. Community interviews of different First Nations groups were used to incorporate LEK and TEK into the past models, while the present day model was assembled with contributions from students, researchers, and DFO staff providing information on different functional groups.

Past and present ecosystem models, for their construction, rely heavily on the literature for parameter estimates. However, reliable data for each species in a specific area are not always available. Literature values used for estimating parameters for Hecate Strait and Strait of Georgia models are based on contributions from numerous researchers and students (Pauly et al. 1998a). Methods have been devised to accommodate for the lack of data by using other modelling techniques as well as the use of LEK to approximate parameters (Martell and Wallace 1998; Martell 1999). While this proves to be a time-intensive effort, it can be invaluable when no other parameter estimates are available. Also, the 'pedigree' of each input into Ecopath indicates its associated uncertainty.

The Strait of Georgia model was created by various researchers contributing knowledge of different functional groups (Table 1). The model in its completed form appears in Dalsgaard et al. (1998), which integrates all functional groups for all time periods, collectively gathered by others listed in Table 1. An Ecosim model was created for this area (Martell et al. 2002), as discussed in the policy search section.

Mass-balanced models for the Hecate Strait were created (Pauly and Christensen 1996; Beattie 1999) for DFO statistical areas 5C and 5D, using literature values for parameter inputs. Parameters and functional groups, many of which have been carried over into the Northern BC model, are available in Haggan and Beattie (1999). The Hecate Strait model relied on a workshop to expand expertise of different functional groups (Haggan and Beattie 1999). This workshop in Prince Rupert provided a forum for participants to work together to build a model for the area as well as input their ideas on different functional groups. The Hecate Strait model was expanded and provided some of the parameter estimates for the Northern BC model.

NORTHERN BC MODEL

The Northern BC is the most complete BTF ecosystem model to date by the Fisheries Centre for this region. Particularly, it includes all stages of the BTF process for the areas encompassing the Queen Charlotte Sound, Hecate Strait, and Dixon entrance (DFO areas 1-10). Martell (2004) described the two main phases of the BTF process:

1. reconstruct several ecosystem models, with several states between pristine and present; and

¹ Website < <http://www.fisheries.ubc.ca/projects/btf/> > Click on Case Studies (Northern BC), and then on Chronology to access data by species or time period.

2. simulate how to optimally utilise the resources of an unfished ecosystem and compare the results to a present day state.

While the previously mentioned models have been used to accomplish the first of these tasks, the Northern BC model allows tackling the second process of simulating different fishing pressures on past and present ecosystems in order to quantify restoration goals. In order to construct the ecosystem model, it is advisable to choose times before and after major changes in gear or in exploitations levels, in order to see how the system has changed as a result of gear or exploitation changes (Heymans and Pitcher 2004). Vasconcellos and Pitcher (2002) led to the conclusion of three time periods to model for the BC coast: the present, the early 1900s, and the early 1700s. The early 1900s was chosen because of extinctions of sea otters, increases in Steller sea lions, increases in salmon and halibut fishing; also this time was before whaling resumed and before trawls were used. The 1700s was chosen because it was before European contact, and could potentially be the most pristine system which we aim to recreate through BTF techniques. The 2002 BTF workshop on reconstructing past ecosystem models of Northern British Columbia (Ainsworth et al. 2002b), which included students, researchers, DFO members and independent consultants, provided information for ecosystem functional groups, and estimated parameters needed for Ecopath. Functional groups discussed were sea otters, baleen whales, toothed whales, seals and sea lions, seabirds, dogfish, Pacific salmon, halibut groundfish species, Pacific cod, Pacific herring, forage fish, eulachon, and invertebrates, and this knowledge was documented in Pitcher et al. (2002c). Complete models for the BC coast including functional groups, landings, and diet matrices for the 1750, 1900, and present-day, plus an additional model for 1950, are available in Ainsworth et al. (2006). The 1950 model was added to demonstrate what the ecosystem looked like during the heyday of the Pacific salmon fisheries, and before major depletions of commercial fish populations. Thus, once an Ecopath model is created for a time period, Ecosim can drive the system forward in time, and allows the user to identify trends in biomass for each functional group, as well as fit models to data.

4.3 Including long-term climate

In order to accurately project the ecosystem through time, a variety of methods for controlling the patterns of different functional groups have been devised. The first is a forcing function, which can be applied to individual groups as well as different fisheries to increase or decrease a particular functional group or fishery. While this method can be useful when the temporal pattern of different functional groups is known, more general methods can be applied to drive the ecosystem as a whole. This is done by driving primary production, usually with some form of climate data. To quantify past climate scenarios, a variety of methods have been employed to drive primary production of the system, from using tree ring data in Northern BC (Szeicz and MacDonald 1995), growth ring of bamboo coral in the deep sea (Koslow and Thresher 1996), and sea surface temperature to drive biomass models of Japanese sardines (Noto and Yasuda 2003). While these methods can show changes from year to year, forcing functions can be used to emulate longer-term cycles in climate such as El Niño cycles (Pitcher and Forrest 2004). This is done by creating a trend for the forcing function and allowing the user to select the functional groups impacted.

For the Northern BC model, Ainsworth (2006) uses the Pacific Decadal Oscillation index to drive primary production. Once the model is fitted to data and realistic trends in functional groups are obtained, modelling the capacity of past and present fisheries can be done to see how changing the current fisheries will alter the system. Therefore, having an accurate representation of current fishing levels is essential to evaluation.

4.4 Fleet dynamics

The fleet structure created for the Northern BC model in 2000 contains 17 gear types (Ainsworth 2006), with gear structure based on Beattie (2001). The fleet structure chosen by Ainsworth was created to accurately represent the present-day fisheries in the area. However, changes can be made according to the objectives of management (or other users) for the different fisheries. One possibility would be to split fisheries by license type rather than gear type to determine management decisions regarding number of licenses, and how this affects the fishery, rather than lumping all gear types together (Ainsworth 2002). This will add to the number of parameters needed, but may prove crucial when attempting to make management decisions regarding recreational versus commercial limits of each gear type. In addition to quantifying the effects the current fisheries have on the ecosystem, creating alternative fisheries presents the user with insight as to how fishing levels affect the ecosystem status.

Once present fisheries have been examined, the idea of the Lost Valley fisheries and how to examine them becomes important. The LV fleet is a sustainable fleet structure (determined by the user) that will, with present-day use, allow the system to return to the desired LV state. By looking back to the LV, a set of objectives to design an ideal fishery can be applied, with the assumption that, by reducing or altering current fisheries activities, the ecosystem will be allowed to restore itself to a prior state. By looking back 50-100 years, or longer, an ideal ecosystem state can be identified. Then, from that state, different fishing scenarios and goals can be met using an EwE policy optimization method (Pitcher 2004). Each LV fishery has a determined target species and a minimal level of bycatch and discards, assuming that the successful application of technology improvements to this end can be realistically foreseen. LV fleets can be operated with or without recreational fleets, or trawlers, and with a variety of harvest objectives described below. The next step is to determine the level of effort each LV fishery is allowed to impose whilst remaining sustainable and meeting the goals described above (Pitcher 2002). The LV fleet constructed for Northern BC has 12 fisheries (with fisheries set to 2.5% of total biomass of their target groups annually), including groundfish trawl, shrimp trawl, shrimp trap, herring seine, halibut longline, salmon freezer troll, salmon wheel, live rockfish, crab trap, clam dredge, and aboriginal and recreational fisheries (Ainsworth et al. 2004). The next section deals with how fisheries can be quantified in terms of ecosystem value. As humans have a diverse set of values it remains up to the user to identify values important in the goal of ecosystem function.

4.5 Policy search

A policy search can be applied to both the present-day fisheries or the LV fleet in order to assess the economic, social, and ecological values of long-term fishing activities. This policy search is applied in EwE, with the goals of ecosystem function determined by the user, whether social, economic, ecological, or a combination of all three.

These searches determine the fishing effort for each gear type that will maximize the objective function over a fixed-time simulation. Five objectives are used in Ecosim policy searches: ecological, economic, social, mixed objective, and a portfolio log-utility function. The ecological function aims to increase the abundance of slower-growing and long-lived groups, while the economic function maximizes rent from the system, and the social function maximizes fisheries employment. A mixed objective combines the ecological, social, and economic priorities, while the portfolio log-utility adds an additional risk-aversion algorithm, to re-establish the status of a depleted stock (Ainsworth et al. 2004). In a mixed-objective search, different weighting can be given to each function, depending on the desired end-state; for example, if ecological value is more important than social value, the user can allocate a higher weighting to ecological value.

Each objective is then run for a harvest regime under each fleet option, for example, 100-year harvest with 50 years dynamic and 50 years equilibrium, and evaluated. Policy searches for the Strait of Georgia showed maximising economic values led to an increase in all fisheries, with the exception of herring. Maximising both economic and social values reduced ecosystem stability. Maximising ecosystem stability led to an increase in long-lived groups, with slow turnover, but essentially shut down all the fisheries. When all objectives were maximised, economic and social gains were made up through increases in krill, hake, and groundfish fisheries (all fisheries are assumed to have the same value in this case) (Martell et al. 2002). For the Northern BC model, the policy search ranked the 1750 model as having the greatest equilibrium benefits, as the pre-contact system was able to sustain large catch rates, generate wealth and jobs, while preserving the biodiversity at a higher level than other periods (Ainsworth 2006). Although we can evaluate which ecosystems perform better under different policy functions, measuring all values of an ecosystem can prove much more trying.

VALUATION

Assessing biodiversity can be complicated when using ecosystem models, which do not allow for extinctions. Therefore, Ainsworth (Ainsworth 2002; 2006; Ainsworth and Pitcher in press) created a Q-90 statistic, based on Kempton's Q-75 index (Kempton 2002), to evaluate biodiversity of different models using different fishing conditions. The method of assessing diversity compensates for EwE's inability to allow for extinctions. This method allows for a quasi-extinct situation in the biodiversity analysis, even though all groups are still present in some number in the Ecosim model, by calculating how many functional groups fall below a certain predetermined (extinction) level. Extinctions can be an important

factor when assessing the ecological status of an ecosystem and, while this does not place a value on ecological conditions, it does bring modellers closer to assessing it.

Valuation of the ecosystem includes assessing the social, economic, and ecological benefits of each management possibility. Economic assessment has mostly been dealt with in monetary terms, usually as net present value (NPV), and social assessment frequently assesses the number or type of jobs that can be created under different scenarios. These methods, which will be described in more detail later, offer specific numbers that can be compared to assess the value of the ecosystem, and the benefits they will provide. Ecological valuation does not always present itself as a number that can be compared or evaluated. Basic assessments for ecological status, such as number of species, total biomass, or the presence of particular keystone species, can be used as a general indicator, but do not provide an overall assessment of the ecosystem.

Methods (independent of EwE) for addressing the ecological vulnerability include assessing the vulnerability of the species within the system. This has been done by estimating the risk of extinction with a fuzzy logic system, using life history and ecological characteristics to determine intrinsic vulnerability (Cheung et al. 2005). This method was applied to coral reef fishes, although the life history parameters available were not sufficient to reliably predict intrinsic vulnerabilities. However, once supplemental information was added, a significant relationship with observed rates of decline under exploitation could be identified. When the same method was applied to seamount fishes, it also showed that intrinsic vulnerabilities were significantly related to the simulated population declines of marine fishes caused by fishing (Morato et al. 2006). This may prove useful in the future when applying vulnerabilities to Ecosim models or simply evaluating the vulnerability of the species contained in the ecosystem.

Another approach is that developed by Pitcher and colleagues (Pitcher 1999; Pitcher and Preikshot 2001), who used a rapid appraisal system (Rapfish) to assess the status of current fish stocks and fisheries. Rapfish can assess systems on a regional or national scale to identify how countries are performing in terms of sustainability and fisheries management. Rapfish ranks each fishery based on a variety of criteria (ecological, social, ethical, economic, and technological) and then evaluates the level of compliance with the FAO code of Conduct for Responsible Fisheries. This method helps to determine the level of sustainability at which the current fishing operations are performing. Currently at the Fisheries Centre the Rapfish method is being used to assess fisheries sustainability and compliance on a global scale (Pitcher et al. 2006).

In an attempt to analyse ecosystems as a whole, Pitcher et al. (2005) used a Monte Carlo simulation technique to evaluate risk of extirpation, depletion and biodiversity for ecosystem models. Results for the Northern BC model indicated that the risks from present day fisheries are considerable. Christensen (1995) also measured whole ecosystems by ranking 41 ecosystems based on the level of maturity using several of Odum's (1969) concepts of ecosystem maturity. The author compared ecosystems for a variety of factors and acknowledged that findings may be related to measures of stability or maturity but, until we have correctly defined stability and maturity, such findings will remain inconclusive.

Pauly and Christensen (1995) evaluated the amount of primary production required to sustain fisheries globally and estimated that 8% of the aquatic primary production is appropriated by fisheries. This, however, increased to 30-35% when only shelves were considered, an area that provides about 90% of the world catch. This estimate suggests that fisheries may be having a huge impact on coastal ecosystems, a suggestion also made by Wallace (1999), who estimated the primary production required by the BC fisheries. Primary production is important as it forms the basis of all food webs, and simply calculating the amount needed to sustain current fishing proves the need for an ecosystem-based management approach.

These approaches to assess the value of the ecosystems only matter if ecological integrity is the goal of management. These goals must be addressed at the beginning of the policy search, and before evaluating any ecosystem function. It is critical in deciding how to go about the policy search and the weighting given to each objective.

Currently, the Northern BC model encompasses the area of the PNCIMA project, although it lacks watersheds and land ecosystems. In terms of marine ecosystem evaluation, this model offers a perspective on how fisheries could potentially be managed to meet the goals of restoration. Although a model representing 1750 offers the most pristine ecosystem, as well as advantageous social and economic

situations relative to today, the benefits may be unrealistic when costs are considered. The economic section of this report deals with how to make restoration goals feasible and profitable, and potentially offers insight as to how to set realistic goals when comparing benefits of an ecosystem.

Perhaps the most important contribution to these methods would be a way to evaluate ecosystems on a variety of scales and compare them using a common metric. Although we are coming closer to this assessment, mostly in the economic field with intergenerational discounting (see below), we have yet to create a method of evaluation incorporating social, economic, and ecologic values of society into one evaluation system. The policy search routine does allow weighting to be assigned, but this is left to the user of the system and does not necessarily account for a diverse range of values. Incorporating, for example, ecosystem use by First Nations will prove challenging, but insightful, when addressing how the ecosystem is valued. While we have methods for assessing individual values, our methods for considering how to combine values into one assessment still need further development.

5. Ecosystem status

Profound changes in ecosystem structure can be evaluated through several indicators that track the status of species. Indices that would describe the ecosystem are only addressing a part of the ecosystem complexity and are constrained, for the time being, by the data available. We present a few indicators that have been recently chosen by a BC working group (Ministry of Environment of BC in prep.)².

As a result of fishing pressure, exploited species decrease in various proportions depending on their life history, the severity of the exploitation and the state of their habitat. The status of BC salmon populations and stocks has been assessed for their risk of extinction, based on stock assessments and habitat quality for the exploited stocks and the number of spawners that return for the others (Ministry of Environment of BC in prep.). Other exploited species are subjected to stock assessments that vary in scope and in depth, which makes it difficult to draw conclusions about their status. As fish populations become overexploited, the preferred species become depleted and fishers turn to smaller species of lower trophic level. This phenomenon is often called 'fishing down the food web' (Pauly et al. 1998b) and would indicate that the fishery is not sustainable. Its extent is derived from the mean of the trophic level of each exploited species, weighted by its catch. For Canadian waters, Pauly et al. (2001) reported a general downward trend in mean trophic level on both east and west Coasts (Pauly et al. 2001). A recent study, on the other hand, suggests that for BC, mean trophic level is stable (Ministry of Environment of BC in prep.). Although other factors can explain the decline in the mean trophic level of the catch, this index can be used as a first-line indicator, and has been adopted as such by the Convention on Biological Diversity (Pauly and Watson 2005). Secondary indicators proposed are linked to the quantity of unreported catch and of bycatch in the fishery as a measure of success of the management regimes. These indices are linked to the at-sea observer programs and to indirect methods of estimating unreported catches (see earlier sections).

Recent work has attempted to assess the success of marine protected areas by using qualitative indicators of their representation and of the stressors affecting them. With regard to protected areas in BC, ecosystems are not equally represented, and critics are concerned that more economically valuable systems are under-represented (Ministry of Environment of BC in prep.). As far as marine reserves are concerned, 90% of protected areas are for waters less than 200 m in depth. While these areas are important habitats for nurseries, the deeper habitats have been much neglected (Ministry of Environment of BC in prep.). Based on a Conservation Risk Assessment (CRA), the Ministry of Environment (Ministry of Environment of BC in prep.) named harvesting or gathering (fishing) the most common internal stressor to MPAs. When this information is taken into account, only 0.4% of marine areas are larger than 2000 ha and considered protected from gathering or harvesting materials. This suggests that terrestrial ecosystems are better protected (11.7% of land) compared to marine ecosystems (0.4%), although five times more coastal areas are protected today than were in the 1970s (Ministry of Environment of BC in prep.).

Future ecosystem states will be greatly affected by climate, not only with regard to ecological status, but also to the socio-economic status of the people and industries the changes will affect. Trends in temperatures and precipitation established since the 1950s show that, in BC, the average temperature and precipitation have increased. Sea surface temperatures have also shown an increase in half of the stations along the BC coast. One of the stations showing a significant change in mean annual temperature is at Langara Island, at the northern tip of Haida Gwaii. Temperatures in the deep waters off southern BC have

² This report is due to be published in late 2006 and posted on the web

also been shown to have increased over the last 50 years. Models to predict the effects of climate change in the BC area are under development, and the UBC Fisheries Centre is presently applying for funding for a project linking climate change and ecosystem models, with the aim of describing how climate change may impact fisheries in BC.

A potential ecological indicator of ecosystem status is the historical nutrient flow between marine and terrestrial ecosystems, known to enhance community structure and biodiversity in transitional habitats (Hocking and Reimchen 2002). As a complement to catch statistics, a specific indicator for salmon that escape catch and survive to spawn or to be eaten by predators, is the ratio of marine-derived to terrestrial nitrogen (measured as $^{15}\text{N}/^{14}\text{N}$) in plant, vertebrate and invertebrate biomass of riparian forests (Wilkinson et al. 2005). Transfer of ^{15}N from salmon to terrestrial habitats is mediated primarily by bears, through their transport of salmon carcasses and their excretions. The ecosystem flow of marine-derived nitrogen within a BC watershed has been tracked by EwE (including Ecospace) (Watkinson 2001). Nutrient flow may be an important non-market-value (see discussion on ecosystem valuations) indicator of marine ecosystem status.

In recognition that every human population imposes an ecological 'load' or 'footprint' on earth, the 'ecological footprint' concept (Wackernagel and Rees 1996) was introduced as a measure of a population's appropriated carrying capacity or sustained demand for bio-productivity. The ecological footprint of a population is defined as "the area of land and water ecosystems required, on a continuous basis, to produce the resources that the population consumes, and to assimilate (some of) the wastes that the population produces, wherever on Earth the relevant land/water is located" (Rees 2006a). D. Pauly has extended this concept to marine environments to compute the footprint of fisheries or 'fishprints' (www.seaaroundus.org/FootPrintMethod.htm) as the relative primary productivity required for fisheries catches by trophic levels (Pauly and Christensen 1995). This was used to compute the ecological fishprints of nations, which gave an ecological overshoot (fishprint minus biological capacity) of 157% for 2003 (Talberth et al. 2006). Lam and Pauly (unpublished) are developing the 'relative fishprint', expressed in terms of the flow of renewable resources, as an integrated indicator.

Socio-economic indicators of marine ecosystem status measure the current and future net benefits that can be derived from marine ecosystem resources, including their sustainability, distribution, and viability of the fisheries supported. FAO (2003) documents economic indicators of the net profits for the fishery (to the sector and export value) and social indicators, e.g., fish consumption per capita, employment by fleet, lifestyle and cultural values, and numbers of food compliance reports, indigenous fishers, and fisheries management plans with indicators and reference points. The socio-economic indicators of Charles et al. (2001) stress community variables: economic valuations of fishery resources and marine environments, social fisheries resilience, distributions of indices (access to catch within a fleet, sector, catch among fishers within a fishery, and landed value by vessel length), aquaculture (production value and employment), and workplace safety (accident claims registered and compensated per thousand fishers). Their economic indicators value marine ecosystem services and fish stocks (natural capital plus deappreciation and appreciation) along with total landed and fishery export values, market price, contribution to gross domestic product, and employment (by landed weight and landed value). Social fisheries resilience can be tracked by fishers' debt levels, age distributions, proportion with multiple licenses, bankruptcy reports and liabilities, distribution of landed value across species, and diversification of employment sources. The diversity of socio-economic indicators highlights the need for qualitative and quantitative standards of reference.

No indicator selection rules exist for policy-makers, but how well they perform the following criteria should be considered: 1. discriminate among competing hypotheses or alternative policies, either for specific decisions or general policy directions; 2. structure understanding of issues and conceptualise solutions; 3. track performance as determined by results-based management; and 4. inform different resource users (Hezri 2005). More research is needed to establish standard socio-economic indicators and reference values to assess marine ecosystem status. Sumaila (2005b) has suggested a relative quantitative socio-economic indicator for monitoring fishing pressure on marine ecosystems that is based on the discount rates (see section 6.2) fishers apply in their catching decisions. High poverty among coastal fishing communities and corporate debt among commercial fishing companies result in high private discount rates of subsistence and commercial fishers, respectively. The resultant fishing pressure on sustainable management of fisheries and ecosystem status can be monitored by the subsistence (small-scale fisheries) and commercial (large-scale fisheries) fishers' poverty and debt, respectively, relative to national levels:

Poverty index = Income (fishing community) / Income (national poverty line); and

Debt index = Debt (commercial fishery) / Debt (national business entities).

This work can be extended (Lam, unpublished) by defining socio-economic indicators relative to the historically averaged incomes and debts of the subsistence and commercial fishers, over the relevant time scales. Human behaviour, perceptions, and attitudes are influenced by experiences accumulated over a lifetime spent within a particular culture. In the case of fishers, a natural gauge for the individual perception of the size or value of the catch would be the historically-averaged income, debt, or largest catch, as a personal indicator of the deappreciation/appreciation of the economic activity associated with resource exploitation of the marine ecosystem. The relevant time scale depends on context, but one-, five-/ten-, and twenty-year time horizons might be appropriate for calculating 'entrepreneurial', 'established', and 'career' individual averages. While this discussion is framed in the context of the individual subsistence fishers' or commercial fishery's perceptions motivating behaviour, methodologically it is unnecessary to calculate the statistics for the individual. Instead, the average over all individuals within the group of interest can be calculated over the relevant time period. From a policy perspective of assessing ecosystem status and trends, longer time horizons, e.g., twenty, fifty, and one hundred years, might be appropriate. As indicators of the effectiveness of an implemented policy, shorter time horizons, e.g., one, two, and four years, given the period of governmental mandates, would be useful to track and report progress to interested voters and lobby groups. This analysis is related to the shifting baselines of environmental resources (Pauly 1995) and intergenerational discounting rates (Sumaila and Walters 2005), both of which implicitly account for self-referencing and individual perceptions.

6. Socio-economic models

6.1 Valuations of 'goods and services' from the marine environment

Economic theory of valuations is based on people's preferences, which depend on cultural belief and value systems, and are captured in behavioural choices as the trade-offs of various alternatives are weighed, given particular resource and time constraints (Sumaila 2005a). A comprehensive economic valuation of ecosystem goods and services includes both market and non-market valuations: market values are traded in the market (e.g., the commercial value of fish harvested and sold), while non-market values are not (e.g., ecosystem services, such as water or nutrient cycling and waste assimilation). They may be alternatively categorised as present and future consumptive and non-consumptive use values: direct and indirect use, option, existence, and bequest values. Direct use values are directly used for consumptive purposes. Indirect use values are used as intermediate inputs to production. Option values are currently unspecified, potentially valuable ecosystem goods and services. Existence values are non-use values, such as the 'biophilic' (Kellert 1993) aesthetic, ethical, moral, and religious values. Bequest values are resource benefits to future generations. Market and non-market values, and their weighting, will vary with country and resource user. Thus, when the marine environment is shared, a full accounting (i.e., a comprehensive compilation) of all ecosystem values of the various resource users is imperative to capture the diverse preferences that may influence the decision-making process and should be considered in shaping policy.

“All things are connected, like the blood that runs in our family...the water's murmur is the voice of my father's father.” (Chief Seattle 1854)

As captured in the above quote, aboriginal values (Lucas 2004), i.e., the historical cultural values of First Nations communities to specific fisheries and marine resources, have been difficult to include within socio-economic models of resource use. Not surprisingly, First Nations people view negatively attempts to attach monetary values to their cultural values, so Haggan (2004) suggested 'social and cultural capital', similar to natural and economic capital, as an alternative valuation method of ecosystem resources integral to First Nations culture and subsistence. How to do this in practice, within socio-economic modelling frameworks and policy negotiations involving diverse resource users, however, is unclear in the absence of universal standards for assessing social and cultural capital. Lack of cross-cultural understanding and recognition of cultural differences in how people conceptualise nature and interact with their environments (Nisbett and Masuda 2003; Atran et al. 2005) constrains progress toward defining such capital. Also, the oral and qualitative nature of traditional ecological knowledge compared with the documented, quantitative,

disciplinary nature of science have made incorporation of First Nation perceptions within ecological models difficult (Cajete 2000). Encouragingly, local and scientific knowledge have been integrated within an adaptive, fuzzy-logic expert-system approach to predict spatial dynamics of shoals of migratory herring (Mackinson 2000; 2001). Both ecosystem-based science and socio-economic models of resource use increasingly overlap with the integrated management perspectives of traditional ecological knowledge, and recognition of non-market values such as aboriginal values, will influence the social dynamics of environmental decision-making.

First Nations requirements were imposed as an extra constraint within the stock dynamics of a recent single-species, multi-species, or ecosystem model fisheries management approach (Sumaila 2004b). First Nations values from fishery resources were incorporated after fisheries resource conservation, but before commercial fishing allocation of harvestable biomass, as legally constrained by the Canadian Fisheries Act. The First Nations values or catch allocations are not specified by the model, but rather are imposed as an external constraint on the fish population dynamics that must be determined by treaty or other negotiations. Non-market environmental resource valuation is difficult without universal social standards, but is increasingly important in assessing human impacts on ecosystems, exacerbated by global trade. This was patent in the wake of the Exxon Valdez oil spill, when standard market valuations of the damage to local marine resources were insufficient to account for loss of 'cultural property' (Kirsch 2001), the traditional ways of living and knowing of indigenous communities (Snyder et al. 2003). A broader understanding of socio-economic valuations of marine ecosystem goods and services by First Nations and coastal communities, including resident, touristic, recreational, and commercial users, would facilitate negotiations for policy implementation of catch levels, as well as integrated management of marine protected areas (Mascia 2004).

6.2 Socio-economic models of resource use

Improved socio-economic models capture the economic and social perspectives of different resource users in, for example:

1. the net price per unit of market goods and services;
2. the unit non-market value derived from the ecosystem; and
3. the discount rates applied to ecosystem goods and services (Sumaila 2005a).

Humans tend to discount future benefits relative to present values by applying a discount rate to flows of net benefits over time. Applied to marine ecosystems, a high discount rate favours present gain over future benefits, as witnessed by the dramatic declines in fish biomass at the local, regional, and global levels. A low discount rate would preserve fish biomass at a higher rate for the benefit of future generations and a higher (non-market) value in the marine ecosystem (Sumaila 2004a). Thus, human behaviour can be modelled analytically through the discount rate, which reflects the risk aversion, impatience or conservatism of resource users, i.e., their differing valuations of marine ecosystem goods and services in decision-making. The discount rate (in % per unit time) prevailing in a given country may be expressed mathematically as a function of different indices of human behaviour at the level of individual resource users: the degree of uncertainty of future outcomes; the risk aversion prevailing in that country; and the levels of poverty, debt, and capital non-malleability of the fishing gear (Sumaila 2005a). Hence, high discounting correlates with high uncertainty of future outcomes, low risk aversion, high poverty, high debt and low capital malleability.

Such a socio-economic perspective, for which modelling by evolutionary game theory is well-suited, makes explicit the marine resource policy conflict that managers face of whether to accept short-term costs to accrue long-term benefits. The human tendency to discount future flows of benefits (from marine and other natural resources) predisposes them to unsustainable management practices and policies. Marine ecosystem restoration is strongly favoured, however, in an intergenerational cost-benefit analysis (Sumaila 2004a): the time perspective ('discounting clock') is adjusted so that future benefits are valued by future generations, who will benefit from restoration efforts and thus value them fully. A social welfare function ('intergenerational equity') has been derived (Sumaila and Walters 2005) which balances empirical or 'personal discounting' with ethical or 'social discounting' in inter-temporal preferences. The perspectives of both current and future generations are explicitly incorporated, as required in many management jurisdictions by sustainability mandates, to calculate net discounted benefits from natural capital and environmental resources use.

The current economic value is widely assessed by the net present value (NPV) of a flow of net benefits, defined as the sum over all time of the product of the net benefit in a given period and the weight used to discount the benefit. In this method, the (present) cost of restoration will in many instances not be economically beneficial due to discounting of (future) benefits. Restoration becomes economically viable for current and future generations in the near future with an intergenerational weight or discount factor (Sumaila 2004a). By summing intergenerational NPVs to calculate the flows of benefits that accrue to future generations from the time perspectives of those generations receiving the benefits, future flows are discounted less and restoration efforts are valued more.

An ecosystem-based management approach may be impeded by the diverse social and economic (compounded by cultural and political) objectives of resource users, which lead to different 1. market and non-market valuations and 2. time preferences or discount rates, in valuations of flows of benefits (Sumaila 2005a). Policy solutions include: 1. 'side payments' or 'carrot' incentives to alter the high-discount-rate entity's discounting; 2. 'stick and carrot' sanctions and incentives by the low- to the high-discount-rate entity to lower its discount rate and weight non-market values more; and 3. raising awareness of non-market values by environmental non-government and civil-society organisations (Sumaila 2005a). A greater understanding of resource user's time preferences and relative valuations allows the majority to offer economic, social, moral, and ethical incentives and impose sanctions by appealing to the interests (values) of a user group to change so that those sharing the resources benefit. Successful cooperative bargaining requires both sides to make compromises and trade-offs.

Berman and Sumaila (2006) compared the problems of discounting benefits of ecosystem restoration to valuing the amenities that the restored ecosystem could produce. They applied these methods, using a 7% discount rate (comparable to present day), to two scenarios for an Icelandic marine ecosystem: status quo and restoration. The status quo scenario predicted high initial total average catches, which then declined steadily in the 100-year simulation; the restoration model produced lower average catches for roughly 25 years, followed by sustained higher levels of catch through year 100. The restoration of depleted or degraded ecosystems can be seen as a reinvestment in natural capital, with high short-term costs for benefits in the distant future (Sumaila 2004c). The ultimate environmental consequences of human use of marine resources will be determined by the proximate will of individuals to trade consumerist and societal values for common future benefits (Ashley 2006). Rees (2006b) advocates for a socio-economic worldview that "fosters social equity, cultural dynamism, and human development, all without material economic growth". Such a steady-state socio-economic model of resource use requires concerted individual and political will, which may only emerge with the realization that the strong sustainability criterion applies not only to marine resources, but to us.

6.3 BC case studies of valuation

Economic assessments using EwE with NPV and standard rates of discounting do not favour restoration. For example, the Northern BC model for 1750 (Ainsworth 2006) offered the most economic benefits, if it were available for use today, with a sustainable annual production of \$1073 million, while the 2000 model produced the lowest economic value, demonstrating the reduced production experienced today as a result of depleted ecosystems. Ainsworth and Sumaila (2004a) applied an intergenerational discounting term to the Lost Valley fleet policy search in EwE to allow the future generations to be more heavily weighted when considering the benefit of returns. Ainsworth (2006) shows that restoration scenarios aimed at recreating the 1950 model for Northern BC are able to deliver a greater NPV than the current exploitation levels, using either the LV or current BC fleet. As an investment in natural capital, restoration plans involving both fleets are able to outperform bank interest even after 30 years of reduced fishing, therefore making it profitable in present terms. Using methods such as intergenerational discounting helps to put a longer-term economic vision into perspective when assessing present and future levels of the ecosystem.

New applications to EwE enable users to assess the social and economic values of different fisheries at different ecosystem levels. They extend policy searches to either present-day or LV fleets and quantify the socio-economic implications of altering catch limits or gear types. Interpreting relative policy impacts on social values is more complex than economic impact, which has a monetary standard. Social values may be quantified by diversity (e.g., an employment diversity index) for optimal policy investigations in EwE (Ainsworth and Sumaila 2004b), based on Shannon's entropy function. Total value per gear type can be related to relative number of jobs and employment per sector. When applied to Northern BC, this method identifies the 1750 model with the best employment (total, potential, and diversity), followed by the 1950,

1900, and 2000 models, respectively (Ainsworth 2006). Social values not related to employment are less readily evaluated. For example, First Nations use of an ecosystem cannot be quantified by catches or diversity of jobs created. Quantifying values of different groups to the ecosystem is plagued by the often trumping role of economics in defining social values.

Another consideration relevant to the ecosystems off the coast of BC is aquaculture. Sumaila et al. (in press) have analysed the potential net economic benefits (added value) from the incipient sablefish aquaculture industry on the current capture fishery in BC. They estimated two externalities: ecological impacts on the wild sablefish sector, caused by farm-induced disease spread and parasites and genetic interactions and habitat competition by farmed escapees; and market price effects on both sablefish fisheries induced by the increased sablefish supply. Net social benefits to BC were modelled under two policy scenarios: 1. sablefish farming allowed globally and 2. sablefish farming allowed globally, but banned in BC. They concluded that potential costs to the sablefish industry (both wild and farmed) incurred by the increased sablefish supply would export benefits to the primary sablefish consumers in Japan without gains to BC's and Canada's GDP, export earnings, or number of people employed. A sablefish aquaculture ban in the face of a global industry could benefit BC if the wild sablefish is marketed for a price premium of 20-25% over the farmed product. This analysis demonstrates the socio-economic trade-offs that must be considered before policy decisions are made in fisheries management, as the non-intuitive result above emphasizes.

7. Conclusion

The research in ecosystems at the UBC Fisheries Centre has been centred around modelling and the development of tools to address fisheries management issues, including marine reserves and the interactions of aquaculture and wild fisheries. Several case studies of BC, the Northeast Pacific and many other ecosystems elsewhere in the world have allowed us to refine the modelling tools and our understanding of ecosystems' functions.

Ecosystem models of the coast of BC at various scales (from the Bowie Seamount to the Northeast Pacific (Table 1) have been built by researchers and students of the Fisheries Centre and researchers at the Department of Fisheries and Oceans. The construction of each of these models has served as an opportunity to compile and synthesise a huge amount of disparate data, taking into account oceanographic and geographic features of the coast to delineate ecosystems and, in the process, making gaps in our knowledge much more explicit. The compilation of information alone, for any one area modelled, is valuable in itself as an opportunity for a thorough search of literature and an effort to synthesise knowledge.

Several of the models for the BC area have been refined considerably over the years, and some have been fitted with times series and climate anomalies to improve our understanding of the dynamics involved in the ecosystem. The work accomplished in the PNCIMA region has been important and includes ecosystem modelling to address fisheries management issues, the potential impact of closing the Gwaii Hanaas NMCA, and the state of ecosystems in the past. The several successive versions of each model have involved a team of researchers and graduate students and each iteration has improved the model. The last version of the Northern BC model, published by Ainsworth (2006), may constitute a good basis for the DFO effort to build on.

Past and present ecosystem models have been built for the Strait of Georgia, Hecate Strait, and Northern BC, using Back To The Future techniques. While these models represent the closest we have come in attempting to capture as many functions as possible of the ecosystem in one model, the possibility of using these techniques for policy-making decisions has yet to be explored. Models depicting the situation before European contact are especially difficult in this context, as the earliest period for which it is possible to fit the data to a time series start in 1950. Time periods before this rely heavily on general trends, where a starting and ending biomass are assumed, and there is no method to fit the model to data.

Several indices for evaluating the ecological status have been developed, such as the evaluation of iconic (salmon) or keystone species based on spawning stocks and habitat quality, and the mean trophic level of the catch. The comparison of management scenarios in ecosystem modelling has been realised on the basis of a biodiversity index (e.g., Q-90 statistic), and socio-economic indices. Additional synthetic methods include the use of fuzzy logic to assess species vulnerability to exploitation (Cheung et al. 2005).

Much work has been made in valuing ecosystems. Previous methods of economic assessment rely heavily on the cost to the user in present terms, usually net present value. While this proves effective for quantifying the cost to the current generation, the future generations are completely ignored. Intergenerational approaches to evaluating future benefits of a rebuilt ecosystem do become viable for single-species fisheries, and even when applied to the entire ecosystem, such as the EwE model of Northern BC (Ainsworth 2006).

In the future, it may be useful to include watersheds, land use, and freshwater ecosystems with respect to the current PNCIMA project. This may be difficult in EwE, although some work has been done to incorporate riparian and marine systems for the purposes of modelling migratory species (Stanford 2002). However, use of EwE models to account for the marine ecosystems could potentially be incorporated in combination with land-based models to give a comprehensive overview of the PNCIMA area. While much of this work has been accomplished for the marine environment, knowledge of the watershed systems could help to improve existing marine models and better capture the interactions between the two systems, such as in an ecological footprint/fishprint analysis.

A comprehensive economic theory of market and non-market valuation for marine ecosystem goods and services is being developed. Socio-economic models of resource use have included intergenerational discounting, to describe human behaviour in the face of exploitable natural resources, but seldom aboriginal values, and only in rather *ad hoc* fashion. Both may be explicitly modelled by evolutionary game theory of market-driven and indigenous fishers' strategies with differing discount rates and time perspectives (e.g., 7% discount rate vs. a 7-generational perspective) and valuations (economic vs. ecological and cultural), to predict what mixture of present strategies by different resource users (e.g., commercial and First Nations) can sustainably manage marine resources. By exploring ecosystem-based governance (Lam 2005) - conceptually, in modelling convergence of aboriginal and ecosystem knowledge and values, and empirically, in ongoing conversations with First Nations, government, and academic partners - DFO's objectives for PNCIMA may be approached, including its goal of sustaining marine ecosystem values for present and future generations.

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